DISSERTATION

WATER QUALITY AND FISHERY IMPACTS

Submitted by

Michelan Wilson

Department of Economics

In partial fulfillment of the requirements

For the Degree of Doctor of Philosophy

Colorado State University

Fort Collins, Colorado

Summer 2023

Doctoral Committee:

Advisor: Edward Barbier

Anita Pena
Daniele Tavani
Dale Manning

ABSTRACT

WATER QUALITY AND FISHERY IMPACTS

Fishery production is an important source of income for many people across the world. Marine and freshwater fisheries provide jobs in local communities and is also important for trade in both developing and developed economies. Climate change and human activities pose a threat to the fish production by changing habitat quality. Changes in fish production have direct impacts on economic welfare (consumer and producer surplus) and the goal of fishery management is to improve both biological and economic fishery outcomes. This dissertation explores the relationship between fish production and various water quality indicators in the Estuary and Gulf of St. Lawrence (EGSL) and examines the economic impact of changes in water quality in fisheries. It examines the pathways through which water quality impacts fish stocks and subsequent impacts on harvest and economic welfare, and suggests policies for fishery management. Chapter one examines the impact of water quality in a marine fishery under regulated access. It particularly focuses on how hypoxia impacts welfare in a large fishery that has different degrees of hypoxia. Chapter two examines water quality in a freshwater fishery and focuses on the direct and indirect mechanisms through which nutrient pollution impacts fisheries. Finally, chapter three acts as an extension to chapter one, examining the impact of hypoxia on a marine species with an additional assumption that the impact of this water quality indicator may be different at different life stages, which may change the possible welfare impacts of variations in water quality.

ACKNOWLEDGEMENTS

I would like to express my deepest gratitude to my thesis and research advisor, Professor Edward Barbier, for his patience and guidance throughout this process. He has inspired me to become an independent researcher and has taught me the value patience and consistency throughout the scientific process. He has also demonstrated how staying true to your passion can inspire creativity and a sense of accomplishment.

My sincere thanks must also go to the members of my thesis committee: Professor Anita Pena, Professor Daniele Tavani and Dr. Dale Manning. They have generously given their time when called upon and valuable comments that have been used to improve my work. In particular, Professor Pena was quick to give suggestions for improving my econometric analysis and even took the time to proof-read my work and point out inconsistencies. With Professor Tavani's assistance, I was able to approach numerical analysis using some of the more widely used programs, even though this was a daunting prospect for someone with no prior experience. This forced me to step outside of the box and has enabled me to become proficient at multiple programming languages, some of which I have utilized for my numerical analyses. Finally, Dr. Manning has provided constant reminders that, while my work is interdisciplinary, I am an economist and my work should be grounded in the discipline.

I am grateful to the collaborators for lending their expertise and intuition to the my economic problems: Dr. Philippe Archambault and Charlotte Carrier-Belleau from the Department of Biology, Laval University, Québec, Canada and Marc Mingelbier from the Ministère des Forêts, de la Faune et des Parcs, Québec, Canada.

Special thanks to the administrative staff in the Department of Economics at Colorado State University for always trying to find the answers to my questions. Thank you to Rosanna and Alison for always making the time to provide a listening ear in those moments when I just needed to talk, even though I know for sure that you had better things to do. Your contribution to my success is invaluable. In the same breath, I would like to thank my colleagues and particularly, the members

of my PhD cohort who were stood with me as we struggled (not so silently) together. That includes you, Lackson. Thank you to my friend Sachintha Mendis for agreeing with me that the sunk costs were too great for us to quit after four years. As I got to celebrate her accomplishment, I hope that we will get to celebrate each other's accomplishments soon.

I deeply thank my family for their confidence in me. During the times of my greatest struggles, it was my refusal to disappoint you that kept me going. Though I know that you would never be disappointed in me, the expectations that I thought you had of me, led me to have high expectations of myself.

Finally, I would like to thank me - for believing in me, for doing the hard work and for never quitting (Snoop Dogg, 2018).

DEDICATION

I would like to dedicate this thesis to my mother.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Chapter 1

Estimating water quality impacts in marine fisheries: The case of the Northern Shrimp in the Estuary and Gulf of St. Lawrence

1.1 Introduction

There are numerous environmental disturbances that can affect commercially harvested marine species and their habitat. For the Northern Shrimp (*Pandalus borealis*), these environmental impacts include variations in temperature and oxygen concentration (Daoud et al., 2007, 2008, 2010; Dupont-Prinet et al., 2013). While fluctuations in temperature are mostly attributed to climate change, variations in oxygen concentration may be caused by both climate change and human activities, and this is often used as an indicator of water quality (Rounds et al., 2013). In the Estuary and the Gulf of St. Lawrence (EGSL), Canada, oxygen concentration has declined for seventy-two years since 1930, with a drastic decline of 50% between 1930 and 1980 (Dupont-Prinet et al., 2013; Gilbert et al., 2005). This decline in the EGSL has led to many areas being characterized as hypoxic. Specifically, hypoxia can defined as water having dissolved oxygen concentration ranging from 18 -40%, with severe hypoxia being oxygen concentration below 22% (2 mg L^{-1}) (DFO, 2012). As hypoxia is most common in deep waters, species that inhabit deep waters are likely to be dramatically affected by this decline in oxygen concentration (Gilbert et al., 2005). Its impact, however, depends on temperature, salinity, the species overall tolerance and life history stage tolerance to hypoxia (Roman et al., 2019; Dupont-Prinet et al., 2013; DFO, 2012; Claret et al., 2018; Huang et al., 2010; Roman et al., 2019). For example, while species like the Atlantic Cod die within days of being exposed to severely hypoxic conditions, other species like the Northern Shrimp may be more tolerant, given they routinely inhabit depths with lower dissolved oxygen (DFO, 2012; Dupont-Prinet et al., 2013).

While numerous studies have examined the impact of hypoxia on fish harvest (Zimmerman & Nance, 2001; Craig & Crowder, 2005; Massey et al., 2006; O'Connor & Whitall, 2007; Craig & Bosman, 2013; Smith et al., 2014; Purcell et al., 2017), only a few economic studies have analyzed the direct effect of hypoxia on fish stocks (Huang et al., 2010; Smith & Crowder, 2011; Massey et al., 2006; Diaz & Solow, 1999). Analyzing the impact on stock is important because while shifts in harvest from a particular fishery may be an indicator of fluctuations in stock, there are other random or unrelated factors that may affect stocks and harvest (Vert-Pre et al., 2013). For instance, if fluctuations in harvest are actually caused by harvest regimes instead of environmental variation, policy targeting environmental conditions would be unsuccessful at improving management of the fishery (Vert-Pre et al., 2013). But in some fisheries, changing environmental conditions, such as the shift to hypoxic conditions, are important. One difficulty encountered is that quantifying the impact of hypoxia on fish stock entails isolating its effects from the effects of other covarying environmental factors (Huang et al., 2010; Diaz & Solow, 1999). Huang et al. (2010) included the effects of temperature and salinity, identifying effects that varied with hypoxia and controlling for these. The inclusion of these covarying factors allowed the authors to make conclusions about the impact of hypoxia on fish stock that could not have been drawn in previous studies (Diaz & Solow, 1999).

The EGSL is an example of a marine ecosystem that has been severely impacted by declining oxygen concentration and even hypoxia, as well as other environmental factors. The Northern Shrimp fishery in Atlantic Canada, including the EGSL, began in the 1960s and 1970s. The EGSL shrimp fishery supports the livelihood of many workers, including First Nations and demand for Northern Shrimp for commercial harvesting rose in the mid-1970s with the collapse of cod and redfish stocks. Given the commercial importance of Northern Shrimp in the EGSL, the effects of changes in hypoxia will have welfare impacts. Studies on the impact of hypoxia in North Carolina and areas of the Gulf of Mexico show that hypoxia reduces harvest and a reduction in the direct causes of hypoxia would increase fishery rents (consumer surplus and profits) (Smith & Crowder, 2011; Huang et al., 2010, 2012). However, the welfare impacts of hypoxia depends on fishery management. Under open access, while there may be marginal gains from improved environmental conditions, economic rents from the fishery will dissipate in the short and long run, regardless of environmental conditions, and the fishery will be over-exploited (Smith & Crowder, 2011). However, regulated restricted access fisheries often generate profits (Deacon et al., 2011; Reimer & Wilen, 2013), at least in the short run, and improving environmental conditions serves to benefit stakeholders while maintaining biological health of the stock.

Given the role of hypoxia in determining shrimp stock, it is important to analyze how changes in hypoxic conditions impact fishery outcomes. The aim of this paper is to develop a bioeconomic model under regulated access that will be used to quantify the economic impact of variations in oxygen saturation, by analyzing the impact on the growth rate of shrimp stock, and resulting harvest, and then estimating the welfare impacts of this effect on the commercial Northern Shrimp fishery of the EGSL. Depending on whether regulatory conditions lead to rent dissipation or not, the welfare effect of reducing hypoxic pressure in the fishery are measured in terms of changes in consumer and/or producer surplus (profits). The bioeconomic model is used to inform a fixed effects panel specification, which is used to empirically estimate the impact of oxygen saturation on the Northern Shrimp fishery in the EGSL, using biological and environmental data from each of the four Northern Shrimp fishing zones[1] (Zone 8, 9, 10 and 12 shown in Figure 1.1) from 1990 - 2018. The estimated parameters are used in the bioeconomic model to simulate the welfare effects of hypothetical policy changes that lead to a 5-percentage point increase in oxygen saturation (reduced hypoxia).

The empirical results reveal an overall positive and significant relationship between oxygen saturation and Northern Shrimp growth rate for all zones. Welfare estimates suggest large gains for modest increases in oxygen saturation, with Zone 10 having the largest overall welfare gain and Zone 12 having the smallest loss in profits. Under a 5-percentage point increase in average oxygen saturation in each zone, overall welfare would have increased by CAN$2.2 million[2] annually from 1991 - 2018. These figures give an indication of the benefits of water quality improvement in terms of the consumer and producer surplus associated with the Northern shrimp fishery. This contributes to calls for land-use policies and should aid policymakers when making decisions about correcting the underlying causes of hypoxia. Policies may include increased assistance to agricultural producers to implement practices that reduce the flow of nutrients into water bodies, arising from agricultural activities.

---

[1] Fishing Zones 0 - 7 and 13 - 16 are not in the Estuary and Gulf of St. Lawrence and are not examined in this paper.

[2] CAN$1 = US$ 0.87 in 1991, not seasonally adjusted (Board of Governors of the Federal Reserve System (US), n.d.)

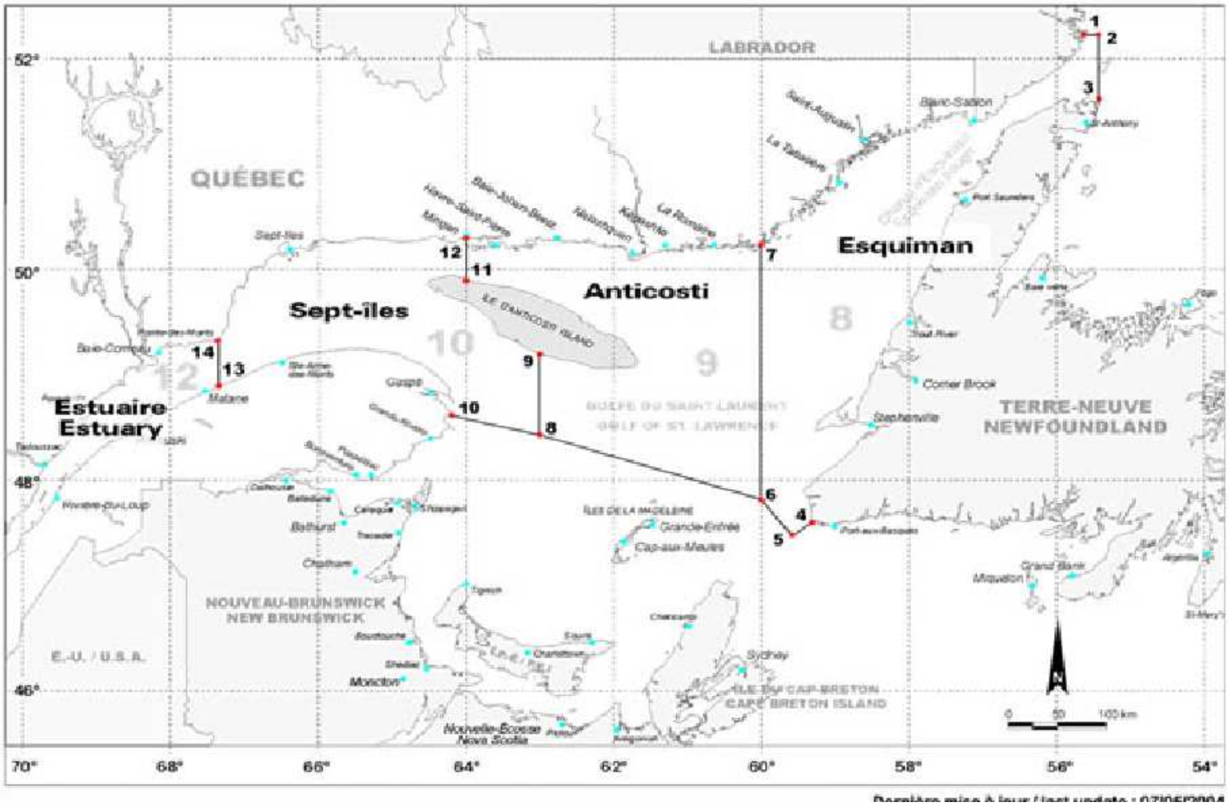


Figure 1.1: EGSL Northern Shrimp Fishing Zones 8, 9, 10 and 12; Source: DFO, Fisheries Management, Quebec Region Fishery Characteristics

This study makes several contributions. First, most previous studies analyzed the impact of hypoxia by examining changes in fish harvest (Zimmerman & Nance, 2001; Craig & Crowder, 2005; Massey et al., 2006; O'Connor & Whitall, 2007; Craig & Bosman, 2013; Smith et al., 2014; Purcell et al., 2017). Our study adds to economic literature on the effect of hypoxia on marine species by analyzing the relationship between oxygen saturation and shrimp abundance. Our approach also incorporates biological information about the species in addition to critical environmental conditions affecting Northern Shrimp. Interpretation of fisheries data to construct an appropriate biological growth relationship requires knowledge of the population, which may be unavailable. Our model uses logistic growth, reduces the number of assumptions that would have to be made in the absence of detailed knowledge about the population, thereby reducing assumption bias and data requirements (Maunder & Piner, 2014; Diaz & Solow, 1999). Using logistic growth, we were able to find a significant relationship between environmental factors and shrimp stock. Our use of oxygen saturation to account for varying degrees of hypoxia allowed us to account for other covarying environmental factors without having to explicitly include them in the model (Huang et al., 2010, 2012). Variation in hypoxic conditions is not the only source of variation in fish stock and it is possible that the impact of hypoxia is obscured by variability in other environmental factors (Diaz & Solow, 1999). The advantage of using oxygen saturation is that it also controls for the interaction of key environmental conditions, such as temperature, salinity and atmospheric pressure, with oxygen concentration. Finally, this is the first paper to analyze the economic impact of hypoxia on a commercially important shrimp species in the EGSL. Our model allows us isolate the impact hypoxia on Northern Shrimp stock and calculate the impact of changes in oxygen concentration on welfare (consumer and producer surplus).

The rest of this paper is organized as follows: Section 1.2 gives a brief background of the commercial fishery in the EGSL that will be analyzed, highlighting the state of shrimp stock, environmental conditions and the biological motivation for the hypothesis on the relationship between hypoxia and shrimp stock. Section 1.3 outlines the bioeconomic model and describes the evolution of the fishery in response to habitat changes. In Section 1.4, the bioeconomic model is empirically estimated using data from the Northern Shrimp fishery in the Estuary and Gulf of St. Lawrence and the empirical results are presented in Section 1.5. In section 1.6, a numerical simulation under different policy scenarios is discussed. In Section 1.7 the empirical and simulation results are briefly discussed and finally, Section 1.8 concludes and gives policy recommendations.

1.2 Background

The Northern Shrimp is a protandrous hermaphrodite species, with individuals first reaching sexual maturity as males, then changing sex and becoming females for the remainder of their life (Bourdages et al., 2020). This has implications for spawning, since the targeted individuals for harvesting are females. Variations in environmental conditions have been found to affect the population dynamics of the Northern Shrimp. The environment in which the Northern Shrimp develops have direct impacts on egg and larvae survival and also significantly affects the growth of juveniles and the growth trajectory of cohorts (DFO, 2020b; Pillet et al., 2016; Daoud et al., 2010). Despite the observed impact of the physical environment on the Northern Shrimp, it is by far the most abundant of all shrimp species in the Estuary and Gulf of St. Lawrence (EGSL) (DFO, 2014).

The Northern Shrimp is one of the most important crustaceans harvested in Canada and the fishery began in the EGSL in 1965 when the first commercial catches were made. In the EGSL, there have been changes in the commercial importance of different species over time, with variations in stock abundance. After the decline of the Atlantic cod stock in 1992, the commercial fishing industry on Canada's east coast underwent a period of change where groundfish has been replaced by shellfish as the main harvested species (DFO, 2016). Together with snow crab and lobster, the Northern Shrimp accounted for two-thirds of the total value of landings of marine species in Canada in 2011 (DFO, 2016). In 2019, commercial fish landings in Québec was valued at 3.9 million. Shrimp catch accounted for 10% of this value (DFO, 2020a). The Northern Shrimp fishery in the EGSL is regulated by the Department of Fisheries and Oceans, Canada and is divided in four zones - Esquiman (Zone 8), Anticosti (Zone 9), Sept-$\hat{\imath}$les (Zone 10) and Estuary (Zone 12) (see figure 1.1). In order to enter the fishery, a fishing licence needs to be purchased and a limit of 111 licences was implemented in 1980. This implies that the fishery is a restricted access fishery, characteristic of the world's most valuable fisheries (Reimer & Wilen, 2013). In addition to fishery licences, management measures for each zone includes a total allowable catch (TAC)[3], which limits fishing effort in each zone (DFO, 2020b) and aims to protect the ability of the population to reproduce. Despite management measures, since 2015, commercial fisheries in the Estuary and Sept-$\hat{\imath}$les have seen declines in abundance of more than 60%, while Anticosti has experienced a decline of 25%. Esquiman has modest declines of 29% for male and 8% for female shrimp. At the same time, total allowable catch has increased in all zones (Bourdages et al., 2018; DFO, 2020b). Low oxygen concentration has been identified as one of the main causes of Northern Shrimp varia-

---

[3] In conjunction with the TAC, fishery management allows an Individually Quota (IQ) scheme. However, there is not a uniform scheme across groups within zones, as some zones prefer that the quota is not transferable. Where trading occurs, this seems to be on a limited basis. Additionally, monitoring and enforcement of the IQ program have been relatively insufficient, so the tradeable quota seems to be ineffective in influencing harvests and profits.

tion in certain areas of the EGSL. Given the commercial importance of the species, understanding how the response to changing environmental conditions is of interest to both policymakers and stakeholders.

In the EGSL, deep water comes from the ocean and oxygen becomes more depleted as it flows upstream, with oxygen concentrations reaching 18-25% in the St. Lawrence Estuary and at the heads of Esquiman and Anticosti channels (Dupont-Prinet et al., 2013). In 2018, observed trends in the decline of oxygen saturation between 1923 and 2017 was two times that of the decline in saturation in other parts of the Atlantic (Claret et al., 2018). This deoxygenation is expected to continue into the future and will continue to affect marine species as they lose favourable oxygen conditions (Claret et al., 2018; Brennan et al., 2016). This is even more so, as climate change increases warming and causes fishes to migrate to deeper waters. The decline in oxygen has been observed to be most persistent in the Lower St. Lawrence Estuary (LSLE), which receives the largest freshwater discharge in North America (Gilbert et al., 2005, 2007; Katsev et al., 2007). The decline in oxygen concentration in the EGSL has been attributed to both climate change and human-induced eutrophication. While climate-change and global warming has been found to intensify naturally occurring hypoxia in the St. Lawrence River, hypoxia in coastal and estuarine environments are considered to be mostly caused by human activities. In this case, an increase in the influx of nitrates increases primary productivity and this contributes to the depletion of oxygen in the bottom waters of coastal and estuarine systems. In the EGSL, the increase in hypoxia has been shown to be positively correlated to increased land use for agriculture (Thibodeau et al., 2006).

Low dissolved oxygen and hypoxia have been found to affect shrimp growth and mortality through affecting the habitat quality (Huang et al., 2010; Chen et al., 2018). However, the impact of dissolved oxygen depends on other abiotic factors such as salinity, temperature and water pressure (Huang et al., 2010; Roman et al., 2019), as these factors impact the solubility of oxygen in water. The solubility of oxygen decreases with increased temperature and salinity but increases with atmospheric pressure. Oxygen saturation is a measure of how much oxygen is dissolved or how much oxygen water holds, as a percentage of the maximum oxygen concentration in seawater (Basher & Costello, 2020; Kutty, 1987). For example, seawater at 10°C with a salinity of 30 percentage points and at normal atmospheric pressure is considered fully saturated (100%) when the dissolved oxygen concentration reaches 12 milligrams per litre (mg/L) of seawater. Alternatively, with constant salinity but decreased temperature to 2°C, the same water body will hold ∼4 mg/L of dissolve oxygen at 100% saturation. Therefore, given that temperature and salinity remain constant changes in oxygen saturation captures changes in oxygen demand, as aquatic organisms and bacteria remove oxygen from water for respiration (Rounds et al., 2013). Hypoxia can be defined oxygen concentration of 40% and can range from mild hypoxia (saturation ∼40%) to severe hypoxia (saturation ∼22% and below). A direct negative impact of hypoxia on the Northern Shrimp was identified by Dupont-Prinet et al. (2013), where declining oxygen concentration negatively impacts the ability of Northern Shrimp to maintain the maximum metabolic rate. It was found that metabolic activities in both males and females were affected by hypoxia. The results of laboratory analysis further indicated that Northern Shrimp stock could be affected by hypoxia through the inflexibility in responding to metabolic demand such as foraging and migration, reduction is feed intake and overall limiting the growth performance of shrimp (Dupont-Prinet et al., 2013; Breitburg, 2002; Craig & Crowder, 2005; Dupont-Prinet et al., 2013; Chen et al., 2018; Magnoni et al., 2018). The impact on the growth rate may thus be through altering the physiological performance Northern Shrimp or the habitat capacity to maintain the population, which results in increased mortality (Breitburg, 2002; Dupont-Prinet et al., 2013; Craig & Crowder, 2005)

There are, however, large variations in stock abundance and oxygen concentration across zones (see Figure 1.2. For example, in 2019, Northern Shrimp adult stock in Esquiman (Zone 8), Anticosti (Zone 9), Sept-$\hat{\imath}$les (Zone 10) and Estuary (Zone 12) was approximately 23,184 tonnes, 19,314 tonnes, 19,914 tonnes and 3,980 tonnes, respectively. The large gap in adult stock between the first three zones and Zone 12 seen in 2019 was the general trend from 1990 - 2019 (DFO, 2020b). Similarly, average dissolved oxygen concentration from 1990 - 2019 was the lowest in Zone 12. Average dissolved oxygen concentration from 1990 - 2019 ranged from 48.87%

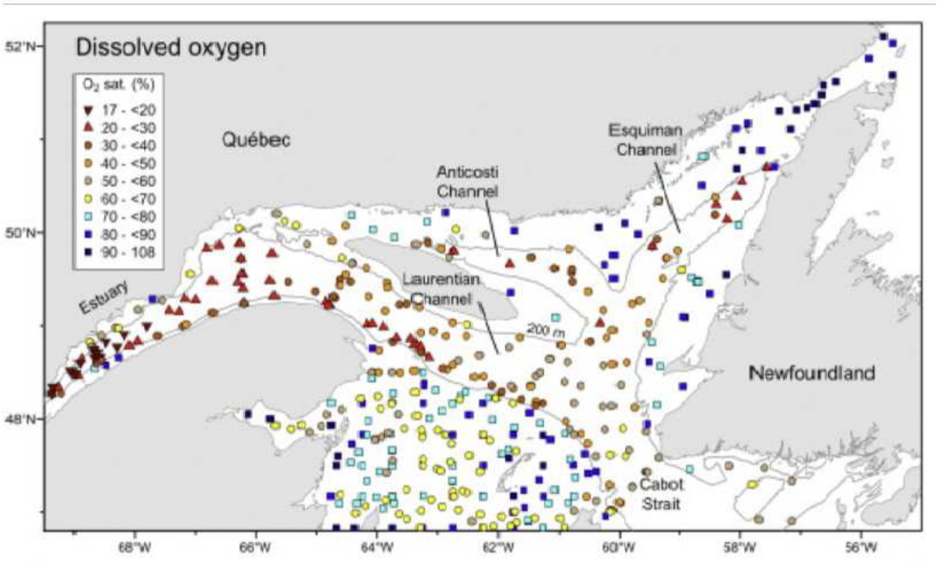


Figure 1.2: Bottom dissolved oxygen concentrations in the Estuary and Gulf of St. Lawrence in 2012 from Dupont-Prinet et al. (2013)

- 61.64% in Zones 8, 9 and 12, while average concentration was 35.45%. In addition, over the same period, dissolved oxygen concentration in Zone 12 fell below 22%, causing the zone to be classified as severely hypoxic in certain years and mildly to moderately hypoxic in others. While there is no evidence to suggest that hypoxia affects stock in Zone 12 different from the other zones, the significant difference in stock between Zone 12 and the other zones implies a significant stock effect that takes place in Zone 12 that does not occur in the other zones. Alternatively, the low and interannual variability stock abundance in Zone 12 compared to other zones may be due to changes spatial distribution in response to severely hypoxic conditions (Craig & Crowder, 2005; Zimmerman & Nance, 2001). There is, however, no evidence that hypoxia affects the spatial distribution of Northern Shrimp between the four fishing zones in the EGSL.

1.3 Bioeconomic Model

In order to investigate the welfare impacts of changes in oxygen saturation, we develop a bioeconomic model that links oxygen saturation to fish production. The underlying assumptions of this model are that oxygen saturation affects the biological growth of the fishery through the impact on habitat quality (Coiro et al., 2000; Cochran & Burnett, 1996; Pérez-Castañeda & Defeo, 2005) and that the fishery operates under regulated access, which, as previously stated, is the prevailng management regime of the EGSL Northern Shrimp fishery. Under these assumptions, changes in oxygen saturation is likely to affect the productivity of the fishery, consequently, impacting consumer and/or producer surplus. We further assume that changes in oxygen saturation does not cause shrimp migration across fishing zones, thereby eliminating spatial spillover effects which does not seem to be occurring in the Northern Shrimp fishery (see Section 1.2. We use a dynamic regulated access model where oxygen saturation affects shrimp growth through its impact on habitat quality to gain insight into the impact of variations in oxygen saturation on welfare.

As discussed in section 1.2, the EGSL is federally managed by Fisheries and Oceans (DFO) Canada. The DFO's management plan includes a quota on total allowable catch (TAC) in each shrimp fishing zone. Following Homans & Wilen (1997) let the regulator choose a quota for total allowable catch (TAC) in each fishing zone, i, based on the zone stock size at the beginning of the season.

$$Q_{it} = Q(X_{it}) \tag{1.1}$$

where $i \in \{1, 4\}$ represents each zone and t is year. The quota on TAC is specific to each zone and is assumed to be a function of its current fish stock, X_{it}. Harvest in each zone and period is a function of fish stock and effort, such that:

$$H_{it} = H(X_{it}, E_{it}) \tag{1.2}$$

We will assume that the total allowable catch is attained in each period, implying that

$$H_{it} = Q_{it}$$

Assuming $H_{it} \geq 0$ harvest, the regulator closes the fishing zone when the quota is attained.

The fishing industry produces effort through the use of capital, such as boats, fishing gear, equipment, etc., and labour. The Northern Shrimp industry is assumed to adjust fishing effort in response to industry profits, where vessels enter or exit the industry in response to to previous period profits (Knowler et al., 2001). This yields the following equation for fishing effort adjustment:

$$E_{it} - E_{it-1} = \phi(R(H_{it-1}) - c_{it-1}E_{it-1}) \tag{1.3}$$

where $R(H_{it-1})$ is fishing revenues and $\phi > 0$ is an adjustment coefficient. $R(H_{it-1})$ is concave in harvests such that $R_H > 0$ and $R_{HH} < 0$. $c_{it-1}E_{it-1}$ is total cost of effort, which is linear in effort. With ϕ being positive, it follows that (1)$E_{it} > E_{it-1}$ if profits are positive; (2) $E_{it} < E_{it-1}$ if profits are negative (loss) and; (3) $E_{it} = E_{it-1}$ if profits are 0. A higher value of ϕ suggests that there is a stronger adjustment of effort to changes in profits or losses than lower values. For example, fishers may be slower to enter the fishery when profits are being made than to exit when there are losses, given the substantial capital investment usually required.

Finally, stock biomass is affected by oxygen saturation, which, as noted in section 1.2, is a direct indicator of the presence of hypoxic conditions in the Northern Shrimp habitat. Let biomass be represented as:

$$X_{it} = F(Sit - 1, X_{it-1}) + X_{it-1} - H_{it-1} \tag{1.4}$$

where current shrimp stock in each zone X_{it} is determined by biological growth, $F(\cdot)$, previous period stock, X_{it-1}, and harvest H_{it-1}. $F(\cdot)$ denotes shrimp growth as a function of previous period water quality (oxygen saturation) and fish stock. Our assumption is that as oxygen saturation increases, the carrying capacity of the marine habitat improves and thus biological growth increases, such that $\frac{dF}{dS} > 0$. Conversely, declining saturation implies deteriorating marine habitat and lower biological growth, which has been observed for Northern Shrimp and other shrimp species (Breitburg, 2002; Chen et al., 2018; Craig & Crowder, 2005; Dupont-Prinet et al., 2013; Huang et al., 2010). $X_{it-1} - H_{it-1}$ is previous period escapement (shrimp that survived harvest). Equation 1.4

can be re-arranged as a dynamic equation describing the evolution of shrimp stock.

$$X_{it} - X_{it-1} = F(S_{it-1}, X_{it-1}) - H_{it-1} \tag{1.5}$$

Shrimp growth will be described using the generalized logistic growth function presented in Birch (1999). The biological growth function from equation 1.5 is therefore:

$$F(S_{it-t}, X_{it-1}) = \frac{rX_{it-1}(K(S_{it-1}) - X_{it-1})}{K(S_{it-1}) - X_{it-1} + vX_{it-1}} \tag{1.6}$$

where *r* is the intrinsic growth rate of shrimp and *K* is carrying capacity, which is a function of saturation. The value of v influences the functional form equation 1.6. $v > 1$ implies that the maximum growth rate occurs when the stock is less than $K/2$ and $v < 1$ implies exponential growth where growth rate exceeds $K/2$. Substituting for $F(\cdot)$ in equation 1.5 yields

$$X_{it} - X_{it-1} = \frac{rX_{it-1}(K(S_{it-1}) - X_{it-1})}{K(S_{it-1}) - X_{it-1} + vX_{it-1}} - H_{it-1} \tag{1.7}$$

Equation 1.7 illustrates the generalized logistic growth relationship between oxygen saturation and shrimp stock. Changes in oxygen saturation will directly impact carrying capacity, and thus the stock of shrimp available for harvest in the subsequent period. For $v > 1$, shrimp growth is described using a logistic growth function (Hvingel & Kingsley, 2006; Kar & Chakraborty, 2011). If we assume a value of $v = 1 + \frac{1-K(S_{it-1})}{X_{it-1}}$, then equation 1.7 becomes

$$X_{it} - X_{it-1} = rX_{it-1}(K(S_{it-1}) - X_{it-1}) - H_{it-1} \tag{1.8}$$

Equations 1.3 and 1.8 describes a dynamic model of a shrimp fishery, which incorporates the influence of oxygen saturation directly into the evolution of the shrimp population.

The economic impact of variations in oxygen saturation can be estimated by determining how they influence stock and harvest, and the resulting impact on welfare, through changes in consumer and producer surplus. However, this impact depends on the institutional arrangements, which determine the rents to be derived from the fishery. To help characterize and interpret institutional arrangements, adopt specific functional forms for shrimp market demand and revenues.

Given the large harvest in the EGSL Northern Shrimp zone relative to total market output, changes in harvest is is likely to affect shrimp price. This can be represented through assuming the industry is price-making, and prices adjust with changes is harvest. For tractability, we will specify an iso-elastic demand function for Northern Shrimp such that

$$p(H_{it}) = kH_{it}^{\frac{1}{\epsilon}} \tag{1.9}$$

where ϵ is the price elasticity of demand. It follows that the revenue function in each period is now $R(H_{it-1}) = kH_{it}^{\eta+1}$ for $\eta = \frac{1}{\epsilon} < 0$.

Given that the Northern Shrimp fishery is regulated by a quota system, the quota chosen in each time period puts a maximum cap on fish catch. As such, revenue in each period is restricted by the allowable catch limit, $R(H_{it}) = kQ_{it}^{\eta+1}$. Following Reimer & Wilen (2013), we assume constant returns to scale (CRS) technology, such that the marginal cost of effort is $c = c(w, r)$, where w is the wage rate and r is the rental rate of capital. Consequently, per period profits are

$$\pi_{it} = kQ_{it}^{\eta+1} - c_{it}(w, r)E_{it} \tag{1.10}$$

Changes is producer surplus due to variations in oxygen saturation can be calculated using equation 1.10. There could also be an impact on consumer surplus. For example, if the change in oxygen saturation causes harvest to change from H_0 to H_1 then the change in consumer surplus (CS) is

$$\triangle CS = \int_{H^0}^{H^1} p(H)dH - (p^1H^1 - p^0H^0) = -\frac{\eta[p^1H^1 - p^0H^0]}{\eta + 1} \tag{1.11}$$

Any changes in profits and/or consumer surplus are dependent on the assumptions about the institutions governing management of the fishery. We can gain insights by examining different institutional arrangements.

In open access fisheries, the effort level chosen by the industry is predicated by the assumption that entry continues until profits[4] (and resource rents) are driven to zero and the fish stock is over-exploited. A similar situation exists under regulated fisheries, except that the quota set by the regulator ensures that a healthy fish stock is maintained. However, a quota on TAC does not create well-defined property rights. Under regulated access, effort responds to profits and the market expands as demand increases, causing effort to increase above the efficient level. Regulators impose restrictions (such as shorter fishing season) to ensure that the total allowable catch is not exceeded. Given the restriction on supply, effort will depend on the ex-vessel price of fish and will adjust to meet the quota. However, with the restrictions imposed by the regulatory authority, costs will also rise, resulting in the dissipation of profits[5]. The industry effort chosen will be the profit dissipating level of effort and stock biomass will be constant. If the stock biomass is changing, both the regulatory instrument and the industry decision on effort will change (Homans & Wilen, 1997). Profits will be dissipated in the long run as the industry quota doesn't solve the problem of the 'race to fish' and overcapitalization of the fishery. In this scenario, it is also possible that profit dissipation could occur in every period (Homans & Wilen, 1997).

However, a licensing program and other restrictions on fishing effort may delay profit dissipation by limiting entry and exit. Licensing restricts the capital entering the fishery. Under a restricted regulated access fishery like the Northern Shrimp fishery in the EGSL, licensing puts a cap on the amount of capital in the fishery, by fixing the number of vessels, and thus restricts entry. Licensing, however, creates ill-defined property rights since the marginal increase in effort by one fisherman negatively impacts other fishermen. Licensing restricts the number of vessels entering a fishery but does not usually limit the size or technological capabilities of the vessels. Additionally, if labour is easily substituted for restricted capital, the effort exerted in a restricted access fishery

---

[4] Here, we differentiate between profit and resource rents. The former is a basic indicator of business performance of firms, while the latter is for the evaluation of the contribution of resources and industry to economic welfare. In the long run, observed profits are often used as an estimate of resource rent in a fishery (Asche et al., 2009)

[5] If management is able to provide a TAC that ensures that stock is higher than the open access level, there is the potential for resource rents, despite profits being zero (Asche et al., 2009). This, however, does not occur partly because of increased cost to catch the TAC and additionally because of the cost of the regulations.

will be the same as under an open access fishery and profits (and resource rents) will dissipate (Deacon et al., 2011; Reimer & Wilen, 2013). But under most scenarios, one would expect licensing and other restrictions to inhibit entry and exit and thus prevent profit dissipation, at least in the short run[6]. Such barriers to entry and exit may be why, despite the presence of a quota, the EGSL fisheries consistently made profits over several periods (DFO, 2006b). While there may be profits in the short run, this competition in the fishery still leads to rent dissipation in the long run and zero profits due to the resource being shared (Asche et al., 2009). We therefore have two possible scenarios for a regulated restricted access fishery:

Case 1. *Short Run Profit Dissipation.*

The regulator chooses a binding quota $Q_{it} = m + bX_{it}$ (Homans & Wilen, 1997) on TAC yearly, based on biological consideration. We assume that there is no restriction on capital and accompanying labour entering (leaving) the fishery quickly, in response to profits (losses), subject to the maximum[7] amount of capital allowed by the regulator. In the short run, the industry can increase (decrease) fishing effort by latent effort entering the fishery (Holland, 2011), increasing capital per boat (Stollery, 1986; Karpoff, 1985) or increasing labour (depending on substitutability) in response to perceived marginal cost. However, as in many fisheries, the regulator closes the fishery to prevent over-exploitation, resulting in increasing marginal cost of effort. We assume that marginal cost of effort to be a function of return on capital and the wage rate, $c_{it} = c_{it-1}(w, r)$. If there are no other restrictions that limit effort in the fishery, the 'race to fish' will result in profit and resource rent dissipation in each period. The rent dissipating level of effort is

$$E_{it-1} = \frac{k(m + bX_{it-1})^{\eta+1}}{c_{it-1}} \tag{1.12}$$

for $H_{it-1} = Q_{it-1} = m + bX_{it-1}$. When biomass changes, both the regulator instrument and the industry effort changes such that a temporary equilibrium is reached in every time period t. Effort

---

[6] Resource rents and profits cannot be used interchangeably in the short run. Out short run analysis focuses on profits since resource rents are dissipated in the short run even under restricted regulated access

[7] The maximum of capital is set by the number of licences allocated to the fishery.

in each period is dependent on stock and the regulatory parameter. Effort is further dependent on the elasticity of demand such that, if $-1 < \epsilon < 0(\eta < -1)$, $\frac{dE}{dX} < 0$ and if demand is elastic $\epsilon < -1(-1 < \eta < 0)$ then $\frac{dE}{dX} > 0$. If the regulator increases the quota in response to increased stock, under our zero profit assumption, changes in biomass would lead to changes in aggregate revenues but total industry profits are zero. Only consumer surplus is impacted as industry prices change in response to increased harvest. This is characteristic of a regulated open access fishery with short-run rent dissipation.

Case 2. *No Short Run Profit Dissipation.*

The regulator chooses a quota $Q_{it} = m + bX_{it}$. The fishery continues to attract effort as long as profits are positive and effort will decline if there are losses. However, licensing and other restrictions (e.g. season length) on fishing effort results in slow entry so that there can be nonzero profits in the current time period t, but not in the long run (Stollery, 1986). Short-run profits (losses) are made as the fishery transitions to the equilibrium level of industry effort. In addition to the binding quota, effort in each time period is restricted by the regulator through licensing and other measures, such that $E_{it-1} = \bar{E}_{it-1}$ is the limit determined by the regulator. The maximum effort can be defined as

$$\bar{E}_{it-1} = \frac{m + bX_{it-1}}{\bar{h}_{it-1}}; \bar{h}_{it-1} = \frac{H_{it-1}}{\bar{E}_{it-1}} \tag{1.13}$$

where h_{it} is catch per unit effort (CPUE). Equation 1.13 gives the effort required to catch the entire quota, and as such, puts an upper limit on the effort allowed allowed to enter the fishery. Profits are then residually determined by

$$\pi_{it-1} = k(m + bX_{it-1})^{\eta+1} - c_{it-1}\bar{E}_{it-1} \tag{1.14}$$

Substituting for the maximum allowable effort in equation 1.13:

$$\pi_{it-1} = k(m + bX_{it-1})^{\eta+1} - c_{it-1}\frac{m + bX_{it-1}}{\bar{h}_{it-1}} \tag{1.15}$$

Here, profits are a function of biomass, and changes in biomass will lead to changes the harvest quota and therefore profits. As the harvest quota changes, prices and consumer surplus will also change.

In the following sections, we will apply the model outlined above to the Northern Shrimp fishery in the EGSL by empirically estimating the impact of changes in oxygen saturation on Northern Shrimp growth rate. We will then use the results of the estimation, combined with the above bioeconomic model, to find welfare impacts of changes in oxygen saturation.

1.4 Empirical Strategy

We develop an empirical approach combined with the bioeconomic model to test the hypothesis that growth in the Northern Shrimp stock in the EGSL is positively impacted by increases in oxygen saturation. We employ a fixed effects panel analysis using data from the Northern Shrimp fishery in the EGSL for the four fishing zones over 1990 to 2018. Our empirical model assumes the impact of oxygen saturation is the same across all four zones, and as such, pools data on stock and oxygen concentration to identify the common impact across groups [8]. With no evidence of significant differential impacts of oxygen saturation across zones, our pooled data allows us to have more information and improve efficiency. We will, however, use fixed effects to control for other individual zone characteristics. Equation 1.8 can be re-written as a growth equation and will be adapted to include zone-specific dummy variables as illustrated in the equation below:

$$\frac{X_{it} - X_{it-1}}{X_{it-1}} = \alpha_1 + r(K(S_{it-1}) + rX_{it-1} + \alpha_2 \frac{H_{it-1}}{X_{it-1}} + D_z + \sum_{q=3}^{5} \sum_{z=8}^{10} \alpha_q D_z X_{it-1} + \epsilon_{it} \quad (1.16)$$

where D_z is a zone-specific dummy variable. For our model, we use Zone 12 as the reference zone, such that the dummy variables included are for Zones 8, 9, and 10 and $z \in \{8, 9, 10\}$.

For our application to the Northern Shrimp fishery in the EGSL, we will include zone specific stock effects. This is to specifically control for observed interannual variations in shrimp stock

---

[8] Biological study on the impact of hypoxia on Atlantic cod in the EGSL suggests uniform impact of hypoxia across zones (Lavaud et al., 2019).

as well as significantly lower shrimp stock in Zone 12, compared to other zones. As such, our specification of equation 1.16 will include an interaction variable that captures zone specific stock effects. This allows the impact of stock to have a different impact on the growth rate of shrimp stock in each zone. ϵ is the error term that is assumed to a normally distributed i.i.d. term.

For the period being analyzed, we will assume a linear relationship exists between Northern Shrimp carrying capacity and oxygen saturation such that[9]

$$K(S_{it-1}) = \omega S_{it-1} \tag{1.17}$$

With this functional form, we assume that some amount of oxygen saturation is essential for species survival, such that if $S_{it-1} = 0$, the species will be driven to extinction. This is supported by the biological literature (Dupont-Prinet et al., 2013; Purcell et al., 2017). Combining equations 1.16 and 1.17 and incorporating our assumption that harvest equals quota in every period (see equation 1.3) yields

$$\frac{X_{it} - X_{it-1}}{X_{it-1}} = \alpha_1 + \beta_2 S_{it-1} + \beta_3 X_{it-1} + \alpha_2 \frac{H_{it-1}}{X_{it-1}} + D_z + \sum_{q=3}^{5} \sum_{z=8}^{10} \alpha_q D_z X_{it-1} + \epsilon_{it} \tag{1.18}$$

where $\beta_2 = r\omega$ and $\beta_3 = -r$

Our parameter of interest in equation 1.18 is β_2. This gives the impact of variations in oxygen saturation on the growth rate of Northern Shrimp stock. If the β_2 is zero, we fail to reject the null hypothesis that oxygen saturation does not impact the growth rate of Northern Shrimp. If β_2 is non-zero, we reject the null hypothesis.

---

[9] Biological literature suggests that a quadratic relationship exists between oxygen saturation and carrying capacity, with carrying capacity declining at saturation levels above 100%. This specification was tested using data from the EGSL Northern Shrimp fishery from 1990 - 2018 but similar results were obtained, suggesting no improvement above the linear specification.

1.4.1 Data Description

To investigate the impact of oxygen saturation on Northern Shrimp stock in the EGSL, we utilize fishery and environmental data from 1990 - 2018. The key variables used to estimate equation 1.18 are summarized in Table 1.1; the remaining variables listed in the table are used in our policy simulations and welfare analysis (see Section 1.6). The outcome variable of interest is the growth rate of shrimp stock. Stock biomass[10], harvest and TAC data was obtained from the the 2019 Québec Region Northern Shrimp stock assessment. The stock assessment reports data collected from programs implemented to monitor the fishery and population status on an annual basis. Data is obtained for each zone and is annualized. A research survey is conducted every year in August from a DFO vessel and is used for calculation biomass indices using a geostatistical method. The dependent variable was obtained by calculating the percentage change in total shrimp stock.

Table 1.1: Variable Description

Variable	Name	Source
X_{it}	Total shrimp stock	Bourdages et al. (2020)
S_{it}	Average oxygen saturation	Galbraith et al. (2019)
Q_{it}	Total allowable catch (tonnes)	DFO (n.d.-b)
H_{it}	Total harvest (tonnes)	Bourdages et al. (2020)
p_t	Ex-vessel price per kg	Estimated
c_t	Cost per unit effort	Estimated
π	Profit	Calculated

The explanatory variable of interest is oxygen saturation. Data on oxygen saturation was obtained from the Department of Fisheries and Oceans' (DFO) Atlantic Zone Monitoring Program (AZMP) and other DFO surveys along with the 2018 report on the physical oceanographic conditions in the Gulf of St. Lawrence (Galbraith et al., 2019). The 2018 report also includes data

---

[10] Juvenile shrimp abundance was provided in numbers per zone. To find aggregate abundance, juvenile shrimp abundance in tonnes was estimated using a function for the length-weight relationship. Juvenile shrimp was assumed to have average carapace length 12.5mm (Bourdages et al., 2020) and average weight was estimated as $W_{it} = a_{ij} * 12.5^{b_{ij}}$. Parameters for length-weight relationship were obtained from Bourdages et al. (2020) of Northern Shrimp in tonnes was provided for each of the four zones. The weight-length relationship estimated in 1993 was used for the 1990-2004 period.

on surface and deeper layers temperature, salinity and dissolved oxygen. In each year, oxygen saturation is recorded at different longitudes and latitudes on a monthly basis. Oxygen saturation levels account for fluctuations in temperature, salinity and pressure changes. For the period, 37 million observations were received, which were each assigned a fishing zone and then aggregated to an annual level. Additional data required for modeling includes harvest, quota and other environmental variable. Landing data was used to represent harvest and this may differ slightly from catch data as some landing observations had to be removed from the catch analyses because they were missing or incomplete. Landings were obtained from fishermen logbooks. Quota for total allowable catch was obtained from the DFO Notice to Fish Harvesters. The DFO sets the quotas for the next year based on the projected harvest levels obtained by applying the decision rules of the current precautionary approach. Average annual oxygen saturation was obtained from Galbraith et al. (2019). Table 3.2 gives summary statistics for the variables of interest in our empirical estimation. From our summary statistics we note the difference in stock between Zone 12 and all other zones. We also see that Zone 12 has the lowest average oxygen saturation between 1990 - 2018, with a minimum of 21%, making it the only zone that experienced severe hypoxia during the period of analysis. Zones 9 and 10 experienced mild hypoxia during the same period.

Other fishery and environmental data was collected even though were not used in empirical testing. Catch data is required for numerical simulations under different institutional setting and was obtained from the 2019 Québec Region Northern Shrimp stock assessment. Catch data from fishermen logbooks corresponds to a catch and an effort realised by a vessel for a fishing day in a given location and is used to calculate catch per unit effort (CPUE). CPUE takes into account the changes in the fishing capacity and seasonal fishing patterns. Data on dissolved oxygen concentration, surface and bottom layer temperature, and salinity was retrieved from DFO surveys and may be useful for robustness checks.

Determining the consumer and producer surplus impacts of changes in oxygen saturation requires the price of harvested Northern Shrimp and production costs. The annual ex-vessel price of shrimp in each zone was estimated using isoelastic demand (see equation 1.9). We used price

Table 1.2: Summary Statistics

Variable	**Mean**	**Std. Dev.**	**Min.**	**Max.**
	Zone 8-Esquiman			
X_{it} [a]	31263.64	13719.37	7458.80	62937.47
S_{it} [b]	61.64	8.73	48.82	93.80
Q_{it} [c]	7088.29	2195.92	1117.50	10452.00
H_{it} [d]	7207.60	1993.22	3149.00	10243.59
	Zone 9- Anticosti			
X_{it}	48393.97	23594.36	16233.19	108719.08
S_{it}	57.86	10.59	39.70	95.11
Q_{it}	8034.55	2231.67	4200.00	12456.90
H_{it}	7471.59	1976.31	4162.00	10429.00
	Zone 10- Sept-îles			
X_{it}	61507.13	28418.24	15149.29	122848.14
S_{it}	48.82	12.68	29.97	95.11
Q_{it}	10965.32	3844.61	4267.00	15995.00
H_{it}	10678.12	3840.51	4174.56	15972.00
	Zone 12- Estuary			
X_{it}	2308.94	1128.83	343.59	5159.64
S_{it}	35.12	10.39	21.17	79.76
Q_{it}	1037.03	1416.33	239.00	8288.00
H_{it}	780.59	245.04	214.31	1117.17

29 observations for each variable in each zone

[a] Annual shrimp stock measured in tonnes.

[b] Percentage oxygen saturation, corrected for outliers.

[c] Actual quota on actual total allowable stock, measured in tonnes.

[d] Annual shrimp harvest, measured in tonnes.

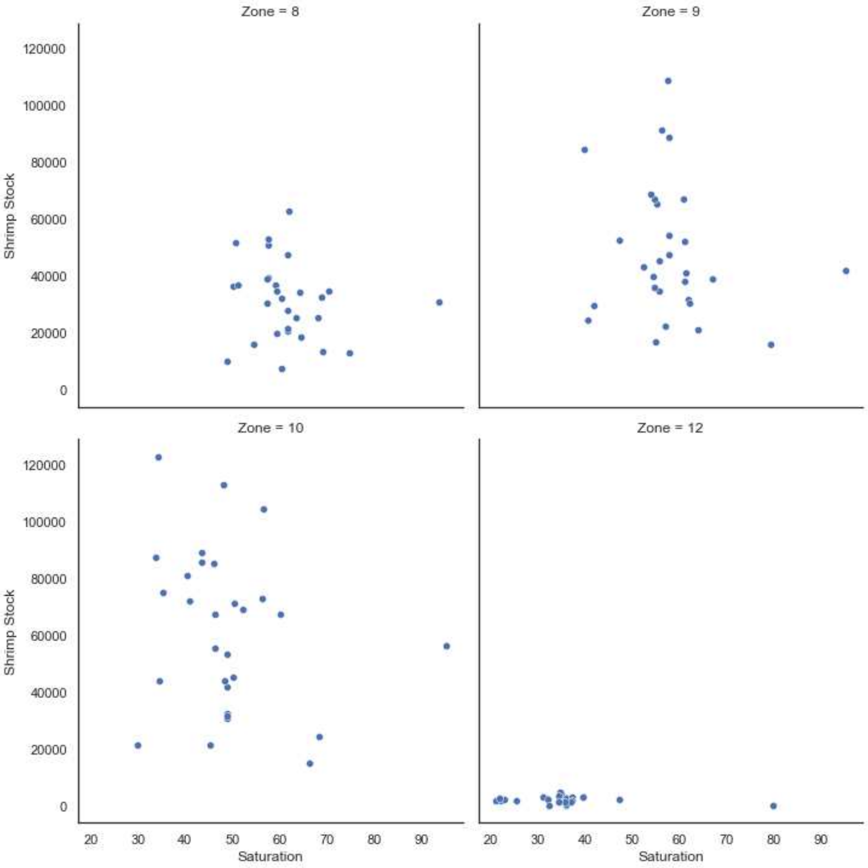


Figure 1.3: Correlations between shrimp growth rate and oxygen saturation

elasticity of demand of -0.5 to estimate price in each year, which is the average price elasticity across all zones during the period being examined. Per unit cost of effort was unavailable and was estimated for each zone based on the annual revenue. It was estimated that between 65% and 72% of annual revenues go toward shrimp fishery operating and maintenance expenses in 2004. These expenses include those expenses related to fishing operations/activities, including labour. The estimates of labour expenses include the amount paid to hired skipper and amount paid to crew net of crew contribution to operating expenses but did not include the amount paid to owner(s) or owner(s) employed as crew (DFO, 2006a). Given the lack of direct information on fishing cost, estimates were obtained using the 2004 Cost and Earnings Survey, Atlantic Region (DFO, 2006a). We estimated total cost estimated as approximately 30% of annual revenues in 2004. This percentage of total revenues was adjusted using Canada's annual producer price index (Statscan, n.d.) in each year. Finally, outliers were removed from each data series. The interquartile range (IQR) was used to determine outliers with the lower and upper bounds for observations being determined by $Q1 - 1.5 * IQR$ and $Q3 + 1.5 * IQR$, respectively. To preserve as many data points as possible, observations above the upper limit and below the lower limit were replace with these bounds.

1.5 Estimation and Results

The fixed effects estimation pools the data from all four zones and restricts the parameters to be equal. There is no evidence to suggest that the impact of the regressors would systematically differ across zones. A Chow test was performed on the parameters independently estimated. It failed to reject the null that there was no difference in the marginal effects of saturation for each zone. However, the test rejected the null that the marginal effect of stock was the same in Zone 12 and all other zones. This supports our use of the Zone 12 interaction variable, which allows the stock effect to be different in Zone 12. The Durbin-Watson test for first order autocorrelation and Breusch-Pagan tests for heteroscedasticity were performed on the individually estimated equations. The tests failed to reject the null hypothesis for zero autocorrelation but heteroscedasticity was indicated in our main equation. Robust standard errors were used but this did not have a large impact on the standard errors and the parameter significance remained unchanged.

1.5.1 Main Results

Table 1.3: Estimation Results

VARIABLES	(No FE) Growth Rate	(FE) Growth Rate	(FE) Growth Rate	(FE) Growth Rate
S_{it-1}	0.669**	0.778*	0.808*	0.544
	(0.321)	(0.463)	(0.443)	(0.359)
X_{it-1}	-0.000319*	-0.000589**	-0.0274***	-0.000416***
	(0.000170)	(0.000240)	(0.00827)	(0.000155)
$\frac{Q_{it-1}}{X_{it-1}}$	8.271	8.935	-9.912	-5.761
	(17.43)	(17.57)	(17.59)	(17.74)
D_8		-1.848	-44.43	
		(20.16)	(33.72)	
D_9		14.52	-45.91	
		(21.21)	(32.72)	
D_{10}		26.88	-61.92*	
		(20.99)	(32.36)	
D_{12}				56.25*
				(32.53)
$D_8 * X_{it-1}$			0.0259***	
			(0.00826)	
$D_9 * X_{it-1}$			0.0266***	
			(0.00826)	
$D_{10} * X_{it-1}$			0.0271***	
			(0.00826)	
$D_{12} * X_{it-1}$				-0.0263***
				(0.00909)
Constant	-19.51	-25.17	47.18	-4.012
	(19.07)	(20.17)	(29.05)	(22.57)
Observations	112	112	112	112
R-squared	0.066	0.099	0.202	0.148

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

Table 1.3 gives the estimation results for equation 1.18. We begin our estimation using simple OLS without fixed effects, as shown in column 1. In our pooled regression, the impact of saturation is positive and significant. However, the variable representing the impact of the quota has a positive sign, implying that as the quota, in proportion to overall stock, increases, growth rate also increases. This is unexpected and inconsistent with our bioeconomic model. In our second regression, zone fixed effects were used. Our results in column 2 indicate that saturation is insignificant when controlling for zone specific effects. However, the sign on the quota variable is again positive. The impact of previous stock is significant in both specifications. Given the apparent significance of previous fish stock in determining the current growth rate of fish stock, we included interaction terms, capturing the differential impact of stock withing each zone in determining growth rate. The results are reported in columns 3 and 4.

In our third specification, all variables had the expected signs. Our variable of interest is oxygen saturation, which was found to have a positive and significant effect on the growth rate of fish stock. The coefficient on saturation, β_2, is 0.808. This implies that $r\omega = 0.808$, where r represents the intrinsic rate of growth of shrimp stock. Given a positive rate of growth, the impact of oxygen saturation on the Northern Shrimp carrying capacity is also positive. The null hypothesis that oxygen saturation impacts on Northern Shrimp growth rate in the EGSL cannot be rejected. The coefficient on previous stock $\beta_3 = -0.0274$, is the negative of the intrinsic growth rate of shrimp stock. That is, $\beta_3 = -r = -0.0274$. As indicated by the literature, the negative sign implies density dependence. That is, as stock size increases, competition leads to a lower growth rate of shrimp stock. Assuming $r = 0.0274$, we can derive a value for ω in our saturation equation (see equation 1.17). The saturation coefficient implies that a 1-percentage point increase in oxygen saturation would lead to an increase in carrying capacity of 29.49 tonnes, on average. The other variables in our chosen equation had the correct signs. As suggested by the literature, an increase in the quota relative to stock has a negative impact on the growth rate of shrimp stock, $\beta_4 = -9.912$, suggesting that the fishery stock is is lower than the maximum sustainable yield. Changes in the relative quota, however, are insignificant in explaining shrimp growth rate. In equation 3 we included dummy variables and dummy-stock interaction terms for Zones 8, 9 and 10. All interaction terms were positive and significant, with almost identical values. The coefficients on the interaction terms suggest that previous stock has a less negative impact on the growth rate in Zones 8, 9 and 10 relative to Zone 12.

To highlight this impact in Zone 12, a final estimation was done, which only accounted for the effects in Zone 12. In equation 4, only a Zone 12 dummy an dummy-stock interaction term were included. Here we see that an increase previous stock size more significantly lowers the growth rate in the Estuary (Zone 12) relative to the other zones. In Zone 12, the coefficient on previous stock is 0.026 lower than other zones. This implies that an increase in previous stock leads to more competition and lower overall growth rate than in other zones. This impact in Zone 12 is significant and suggests that the environmental carrying capacity in Zone 12 may be lower than in the other zones. Additionally, when only the specific effects in Zone 12 are included, saturation becomes insignificant, further highlighting the the zone-specific stock effects.

1.5.2 Robustness

A robustness check was performed to see if the model results were affected by a different measure of hypoxia. Oxygen Saturation was replaced by dissolved oxygen (DO) in the third specification, while controlling for changes in cold intermediate layer temperature (T) and salinity (S). Dissolved oxygen was found to have a significant positive impact on shrimp growth rate. Alternative specifications for the impact of oxygen saturation were also used. The original model assumes a linear impact of oxygen saturation on carrying capacity. Log and quadratic specifications were employed and the results were similar to that of the linear specification. We tested for interactions between oxygen saturation and shrimp abundance. All models testing the various ways oxygen saturation could impact the growth rate of shrimp stock resulted in oxygen saturation have a consistent, positive impact on the growth rate of shrimp stock. Finally, to verify the assumption that the impact of oxygen saturation was not significantly different across zones, we estimated the model for each individual zone. The results are reported in Appendix A.2. When the zones were analyzed individually, oxygen saturation was found to be insignificant in explaining shrimp growth rate.

1.6 Welfare Effects

From the data provided on oxygen saturation in the four commercial Northern Shrimp fishing zones in the EGSL (see table 3.2), we see that only one of the four zones experiences moderate to severe hypoxia (oxygen concentration 40% and below). Zone 12 has oxygen concentration averaging 35% between 1990 and 2018, with some years having oxygen concentration below 22%. For the other zones, average oxygen concentration over the same period was not below 48%. Our policy simulation therefore focuses on the minimum increase in oxygen saturation required for the critical Zone 12, that has experience moderate to severe hypoxia. For Zone 12, a 5-percentage point increase in average oxygen saturation would cause oxygen saturation to increase above the critical level. However, increasing the oxygen concentration in Zone 12 by 5% cannot occur without also increasing oxygen saturation in all other zones. As discussed in Section 2, oxygen levels and hypoxic conditions are attributed to the combination of anthropogenic nutrient loads and underlying environmental factors across the entire EGSL, and it is impossible to target one fishing zone for an improvement in oxygen saturation. Without explicitly modelling the underlying causes for hypoxia in the EGSL, our counterfactual policy simulations will assume a 5-percentage point increase above average oxygen saturation across all zones from 1991 - 2018.

Using the parameters from Table 1.3, we simulate changes in the growth rate of Northern Shrimp stock in the EGSL in response to exogenous policy shocks to oxygen saturation under the two cases outlined above. First, equation 1.18 along with the estimated parameters, assuming existing oxygen saturation levels, was used to create a base case for our simulations. Second, the base case model was calibrated using actual data - stock was initialized as the actual stock in 1990 and the growth rate for each year from 1991 - 2018 was estimated using actual observed oxygen saturation and the implemented quota for each year. Our final step was to perform the policy simulation of a 5% increase in oxygen saturation to determine the consequent impacts on welfare, using the data on price, cost and profit (see Table 1.1 and Section 1.4.1). These steps were undertaken for two counterfactual cases (rent dissipation and no rent dissipation) outlined in section 1.3.

The average oxygen saturation for each zone from 1990 - 2018 was determined and was then increased by 5-percentage points. This was used as the level of oxygen saturation in each year for each respective zone. Given our assumption that the entire quota is fished each year, so $H_{it} = Q_{it}$, an additional parameter estimation for our quota rule was necessary. Using actual quota and stock observations, we estimated parameters *m* and *b* from the quota rule, $Q_{it} = m + bX_{it}$ (see Appendix C.1. These parameters were then used to derive a quota estimate from estimated stock and, consequently, estimated harvest in each year.

If rents dissipate as in *Case 1*, then changes in oxygen saturation will impact only consumer surplus. This was implemented in the base calibration by equating total revenues to total cost. The estimated growth rate, stock and harvest were calculated under the scenario of a 5-percentage point increase in oxygen saturation, using the quota decision rule and the estimated parameters from equation 1.18. Changes in total revenues are exactly balanced by an opposite change in total cost, ensuring that profits are always zero. Changes in consumer surplus were then calculated using equation 1.11, where p_0 and H_0 are price and harvest in the base case (before the change in oxygen saturation).

If rents do not dissipate (*Case 2*), both consumer surplus and profits will be affected. To implement this case, profits were calculated in the base calibration using equation 1.10. Using the estimated parameters from equation 1.18 along with our quota decision rule, we estimated the growth rate of shrimp stock, annual stock and harvest, resulting from the 5-percentage point increase above the average oxygen saturation for each zone. The change in consumer surplus and profits were calculated. The change in profits were calculated as the difference between profits in the counterfactual and the base case. Changes in consumer surplus was calculated using equation 1.11.

Additionally, a price floor, which was the minimum observed price in each zone, was implemented in each of the counterfactual cases to ensure that prices did not fall too low, as harvest changed. The change in consumer surplus and profits were discounted to 1990 using a 5% discount rate in the base case and counterfactual.

1.6.1 Simulation Results

The impact on stock evolution over time of a 5-percentage point increase in oxygen saturation above the mean for each zone, compared to the base case, is illustrated in Figure 1.4. In all zones, Northern Shrimp stock would have been higher than the base case in all years with an increase in oxygen saturation. Simulations show that the stock would have increase until year 2000 then remained relatively stable, if in each year oxygen saturation is maintained at 5-percentage points above the current zone averages.

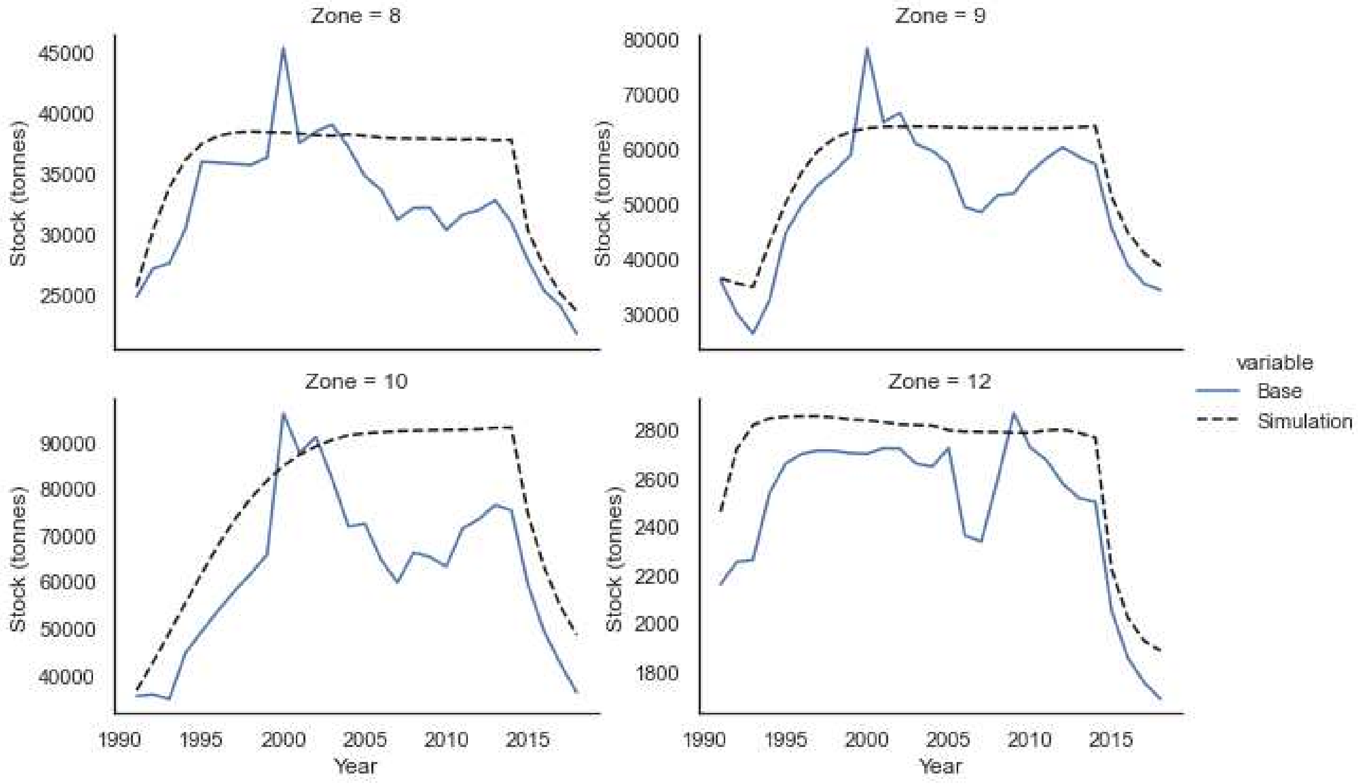


Figure 1.4: Stock evolution in base model and counterfactual with 5-percentage points increase in oxygen.

The results of simulation under *Case 1* are presented in table **??**. Under the assumption that rents are dissipated even in the short run, the increase in oxygen saturation would only impact consumer surplus. When rents are dissipated, the increase in revenues from the increase in stock is exactly offset by the increase in total cost of fishing. Exit and entry results in a level of effort that causes profits to dissipate. Nonetheless, there is an overall increase in consumer surplus from 1991 -2018 in all zones. The change in consumer surplus arises as a result of the fall in prices due to increased stock and harvest. The change in consumer surplus is dependent on the price elasticity of demand and are positive or negative depending on the change in price. The largest increase in welfare occurs in Zone 10, while Zone 12 has the smallest increase in welfare. This result is driven by the fact that actual stock in Zone 10 was cumulatively higher than all other zones over the period of analysis. Further increases in stock cause a reduction in price similar to other zones. However, given the volume of fish harvested and sold in Zone 10, the overall welfare gain will be the highest. The overall discounted change and annual average discounted change in welfare from 1991-2018 across all zones are approximately CAN$146 million and CAN$5 million, respectively, under *Case 1* (see Table **??**).

Table 1.4: *Case 1* change in consumer surplus (CS) with a 5-percentage point increase in oxygen saturation above mean from 1991 - 2018.

Zone	Total change in CS[a] (CAN$)	Average annual change in CS[b] (CAN$[c])
Esquiman (Zone 8)	7,986,275	285,224
Anticosti (Zone 9)	48,570,421	1,734,657
Sept-îles (Zone 10)	89,492,964	3,196,177
Estuary (Zone 12)	117,061	4,180
All Zones	146,166,723	5,220,240

[a] Total change in consumer surplus (CS) in Canadian dollars (CAN$) is estimated as a present value from 1991 - 2018 using a 5% discount rate.

[b] Annual average change in consumer surplus (CS) is the present value total, averaged over 28 years (1991 -2018).

[c] In 1991, CAN$1 = US$ 0.87

The increase in oxygen saturation does affect profits in *Case 2*, where the licensing and quota scheme prevents the ‘race to fish’ and the associated production externalities. Table 1.5 reports the change in net present value profits in each zone for *Case 2* from 1991 - 2018 if oxygen saturation was 5-percentage points higher than the zone average. Profits are positive in each year for all zones. The change in net present value profits is either positive or negative each year, with an overall change in net present value profits in all fishing zones, ranging from -14% to -1%. If baseline oxygen concentration is above average, policy that maintains average saturation causes stock and harvest to fall. With this fall in harvest, price and profits may increase, depending on the magnitude of these changes. However, in most years, a 5-percentage point increase above the average oxygen saturation for each zone resulted in higher oxygen saturation than actual observed in each year. This resulted in an increase in stock and harvest above observed stock and harvest. Given our assumptions about how prices change in response to harvest, as harvest increases, prices decrease. Assuming inelastic demand, this fall in price leads to an reduction in total revenues (as the demand for the good does not increase in proportion with the fall in price). As such, the more inelastic the demand for Northern Shrimp, the higher the losses to producers from improvements in oxygen saturation. Zones with higher baseline stocks experience larger declines in overall profit, as prices prices decline at a constant rate in response to the increase in harvest. The smallest reduction in profits is seen in Zone 12, reflecting very small changes in price and harvest with the small increase in oxygen saturation. The effect in Zone 12 is precipitated by the pre-existing low stock.

In *Case 2*, the price changes also lead to changes is consumer surplus. While changes in overall profits are negative, overall changes in welfare are positive in all zones, indicating a net benefit from increasing oxygen saturation. Welfare increases in three of the four zones. The largest increase in welfare is for Zone 10, approximately CAN$80.4 million, driven by the large change in consumer surplus. Given the decline in profits in Zone 12, there is a very small welfare gain of approximately CAN$83,360 over the 28-year period. The discounted change in overall welfare from 1991 - 2018 across all zones is approximately CAN$129 million under *Case 2*, which amounts to an annual average change of CAN$4.6 million (see Table 1.5, with a 5-percentage point increase in oxygen saturation.

Table 1.5: *Case 2* welfare change with a 5-percentage point increase in oxygen saturation above mean for each zone from 1991 - 2018.

Zone	Total change in CS[a] (CAN$)	Total change in Profit[b] (CAN$)	Total change in Welfare[c] (CAN$)	Average annual change in Welfare[d] (CAN$)
Esquiman (Zone 8)	7,986,275	- 1,266,693	6,719,582	239,985
Anticosti (Zone 9)	48,570,420	-6,706,011	41,864,409	1,495,157
Sept-$\hat{\imath}$les (Zone 10)	89,492,964	-9,070,236	80,422,728	2,872,240
Estuary (Zone 12)	117,061	-33,703	83,360	2,977
All Zones	146,166,723	-17,076,645	129,090,078	4,610,360

[a] Total change in consumer surplus (CS) in Canadian dollars (CAN$) is estimated as a present value from 1991 - 2018 using a 5% discount rate.

[b] Total change in profits or producer surplus (PS) in Canadian dollars (CAN$) is estimated as a present value from 1991 - 2018 using a 5% discount rate. Total revenues are calculated using demand elasticity of -0.5.

[c] Total change in welfare in Canadian dollars (CAN$) is estimated as the sum of present value profits and consumer surplus from 1991 - 2018.

[d] Annual average change in consumer surplus (CS) is the present value total, averaged over 28 years (1991 -2018).

1.7 Discussion

The results of our analysis support the hypothesis that increases in oxygen saturation will have positive impacts on Northern Shrimp stock. Empirical analysis shows that a 1-percentage point increase in oxygen saturation will increase the growth rate of Northern Shrimp by 0.8-percentage points, which is economically and statistically significant. This is observed through the impact on Northern Shrimp carrying capacity.

Our simulations further indicate that there are positive welfare effects of increasing oxygen saturation. If oxygen saturation is 5-percentage points higher than the average in each zone, there is a net increase in fish stock above the baseline scenario. As there are years when actual oxygen saturation is more than 5-percentage points above the zone average, we find that the change in fish stock is negative in particular years with the overall change being positive. Alternate simulations were carried out, where oxygen saturation was increased above the actual percentage in each year (see Appendix C.1). In this case, fish stock was higher than baseline stock every year. The change in fish stock resulting from changes in environmental conditions has welfare implications, and who loses and who gains depend on institutions governing fishery management. One of the important implications of our results is that the welfare changes from an improvement in oxygen saturation vary significantly across zones. For example, we find the largest change in welfare occurred in Zone 10, driven by the large increase in consumer surplus. Zone 10 is the most productive fishing zone in the EGSL, despite having relatively low oxygen saturation over the period of analysis. An increase in oxygen saturation benefits consumers by reducing the market price, with only a very small impact on revenues.

Zone 12 presents an interesting case. Recall that Zone 12 had the lowest average oxygen saturation and also experiences intermittent hypoxia. Zone 12 also had significantly lower annual stock than the other zones. Empirical testing indicated that the stock effect is insignificant. Simulation indicated that a 5-percentage point increase in oxygen saturation above the average had the smallest impact on welfare in Zone 12. While the increase in oxygen saturation would almost double the annual stock, the stock in Zone 12 is sufficiently low, such that there would be very little changes in prices. Additionally, the increase in stock resulting from the 5-percentage point increase in oxygen saturation led to only modest increases in the annual quota and harvest. Our *Case 2* simulations showed that the change in the sum of net present value profits between 1991 - 2018 is approximately zero. The overall welfare change would still be positive, implying that increasing oxygen saturation is beneficial. Additionally, Zone 12 would require a much greater increase in oxygen saturation if significant welfare increases are to be had.

Our simulation results indicate that there are positive benefits from reduction in hypoxia (increasing oxygen saturation) in the EGSL. However the implementation of policy to achieve this depends on the cost of of the policy. Similar studies have been carried out in the Gulf of Mexico where the summer hypoxic zone is one of largest in the world (second to the Baltic Sea dead zone). The hypoxic zone in the Gulf of Mexico ranges from 5000 square kilometers to 22,000 square kilometers and concerns over the increasing size led to the formation of the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force in 1997 (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2020b). The task force is responsible for designing policies for the reduction of the hypoxic zone from approximately 14,500 square kilometers to 5,000 square kilometers. Research into policies that would reduce the hypoxic zone to the target suggests that an approach where the 550 most cost-effective agricultural subwatersheds that deliver nutrients into the Gulf are targeted for cropland conservation investments would cost approximately US$ 2.7 billion annually (Rabotyagov et al., 2014). Given the size size and reduction target, another suggested method of reducing the hypoxic zone in the Gulf of Mexico is to convert all row crop land to grassland, which would cost over US$ 388 billion and the goal would be met in two years (Whittaker et al., 2015). This, however, would likely have implications for commodity prices and employment given that the area contains much of the US Corn Belt. The hypoxic zone in the EGSL in 2021 reached 9700 square kilometers and the cost for implementing policies to reduce hypoxia will depend on whether the policies are targeted or non-targeted and how much of the hypoxic zone will be targeted (Rabotyagov et al., 2014).

1.8 Conclusion

Valuation of the water quality in the EGSL is important for two main reasons. Firstly, despite the observed decline in water quality, human-induced nutrient pollution (eutrophication) continues to be a major threat to the ecosystem. Research has shown that eutrophication has impacted marine life at all levels and has thus impacted the structure and functioning of ecosystems (Todd et al., 2010). This study highlights the impact of one of the consequences of pollution on a marine fishery that has experienced significantly declines in dissolved oxygen in the past and continues to experience this decline. To our knowledge, this is the first study to examine the impact of oxygen saturation on the welfare benefits associated with the Northern Shrimp. Secondly, the decline commercially important species has resulted in calls for better policies policies to manage land use change in order to protect valuable species (Mahmood et al., 2016; Meyfroidt et al., 2022; E. B. Barbier, 2022). The policy options include removing environmentally harmful subsidies in related sectors and more targeted polices that control the sources of pollution (E. B. Barbier, 2022; Xia et al., 2020).

The results of the impact of oxygen saturation on shrimp stock growth are consistent with evidence from the literature that indicates a positive relationship between oxygen saturation and shrimp stock (Kungvankij et al., 1986; Wei et al., 2009; Rahman et al., 2020). The welfare analysis also indicates the importance of institutional arrangements (open access, regulated open access, regulated restricted access) and market conditions when analyzing who gains and who loses from policy implementation. In our simulations, we examine two different fishery management regimes that lead to opposing outcomes, the case when short-run profits are dissipated and the case when they are not. Based on actual cost and earnings data for the Northern Shrimp fishery over 1990 - 2018 (DFO, 2006a), the case in which profits are always dissipated in the short-run is the less likely scenario. Changes in oxygen saturation will most likely affect both profits and consumer surplus. However, regardless of the institutional arrangements and fishery management regime, we found that a reduction in hypoxia (increased oxygen saturation) would prove beneficial, by increasing economic welfare. Given large enough changes in price in response to increased quota and harvest, the fall in price is enough to reduce profits significantly in Zones 8 and 9. In our simulations, fisherman would experience reduced profits while consumers would benefit, with a positive net effect. Consequently, our case study on the welfare impacts of water quality on Northern Shrimp has a number of policy implications.

Additional policy actions are needed for reducing the proximate causes of hypoxia in the EGSL. Eutrophication (i.e., anthropogenic nutrient and organic enrichment of waters) is a major contributing factor to the increasing hypoxic conditions in coastal waters worldwide, which is exacerbated by global warming. Controlling the sources of anthropogenic nutrients (e.g. fertilizers, sewage) will need to be implemented in the EGSL to reduce hypoxia and its impacts on the Northern Shrimp fishery. A first policy step would be to examine social and economic changes that have impacted water quality policies. This would include the role of agricultural markets in increasing human-induced eutrophication in the St. Lawrence River. To control the human-induced influx of nitrogen and phosphorus from non-point sources, a nutrient permit and trading program can be established, similar to the plan implemented by the Miami Conservancy District (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2020b). This is similar to a tradeable carbon permit system and may require scientist to set a Total Maximum Daily Load (TMDL). Policy-based approaches would include increased assistance from Agriculture and Agri-Food Canada to agricultural producers for implementation of practices that reduce the influx of nutrients to water bodies. This policy has proven beneficial in the United States, where the restoration of 1.4 million acres of wetlands has been attributed to policies providing support to agricultural producers (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2020b).

Additional policies may be needed to limit any undesirable consequences for the fishing industry from policies to control hypoxia. These negative impacts may be mitigated by implementing price floors that prevent excessive reductions in profits. In our simulations, price floors equalling the minimum observed prices were implemented. Higher price floors would be beneficial to the fishing industry. Alternatively, an adjustment of the quota rule that puts stricter rules on how harvest changes with increasing fish stock may prevent prices from falling too quickly or too low.

Our empirical estimation and welfare analysis adds to the growing literature on ecosystem services valuation, but it is not without limitations. Data on stock biomass and abundance did not allow estimation of possible separate effects on juvenile as opposed to adult abundance. This would have allowed us to to examine the the impact of saturation at different lifecycle stages. Additionally, while we were able to find a significant impact of oxygen saturation on Northern Shrimp stock in the EGSL, we were unable to test this relationship assuming alternative biological growth functions. (Tian et al., 1993; Hernandez-Llamas & Ratkowsky, 2004; Parrack, 1978). Another limitation related to econometric estimation and broadly using the *environment as an input* technique for valuing environmental quality, is the assumption that there are no ecological thresholds. Our model does not account for the fact that there may be a lower limit of oxygen saturation below which the fishery would collapse, which would have drastic welfare impacts. While identifying threshold responses to changes in oxygen saturation would make management easier, (1) our period of analysis does not include a point of fishery collapse and (2) the biological literature gives no clear indication of what this threshold is. As such, this is outside of the scope of this study.

Another important research issue when analyzing the impact of hypoxia on fisheries is accounting for spatial effects, especially for species that inhabit the bottom layer of the water column. Hypoxia has been found to affect shrimp fleet behaviour through the effect on the spatial distribution of the species, causing migration from deoxygenated areas (Zimmerman & Nance, 2001; Craig & Crowder, 2005; Craig & Bosman, 2013; Smith et al., 2014). In the event of migration, the relationship between hypoxia and fish stock in a particular site is determined by the species behavioural response to variation in water quality and density across sites (Craig & Bosman, 2013; Massey et al., 2006; Domenici et al., 2007). As an additional extension to our analysis, it could be important to include a variable representing migration across sites since migration affects stock abundance at a particular site. This creates additional difficulty when analyzing stock effects at a single site, as stock changes may be due to relative hypoxic conditions. Our analysis entailed examining fish stocks from adjacent fishing sites and captures effects of relative hypoxia. In the long run, hypoxia only affects stock abundance, as the positive habitat change through migration is balanced out by the negative change caused by overcrowding (Massey et al., 2006). As such, it may not be as important to account for migration in long run analysis. This, however, may be explored in further research.

Other limitations are related to policy simulations. In our policy simulation, we assumed a 5-percentage point increase in oxygen saturation because it would take the critical Zone 12 above the hypoxic threshold, and this was the simulated increase across all zones. However, water loses oxygen as it flows between zones, so higher a larger increase would be needed in zones in the upper portions of the river in order to achieve a 5-percentage point increase in Zone 12, which is in the Lower St. Lawrence Estuary (Gilbert et al., 2005). It may be more intuitive to model the pathway though which a policy intervention would lead to an increase in dissolved oxygen across the different zones. This would require data on the source of pollution in the EGSL and the biological and hydrological mechanisms through which pollution leads to hypoxia. This is an area for future research.

The results from policy simulations are sensitive to assumptions made about how prices change in response to changes in Northern Shrimp harvest in the EGSL. Our assumption that the EGSL Northern Shrimp industry has market power and the demand for Northern Shrimp products is inelastic resulted in changes in consumer surplus regardless of whether or not restrictions on fishing effort prevented profit dissipation and reductions in profit when dissipation was avoided. However, the ex-vessel price of Northern Shrimp from the EGSL is not only dependent on the supply of Northern Shrimp in the EGSL and the rest of the world, but also on the supply of all other shrimp species across the world. As such, fishers in the EGSL have very little control over Northern Shrimp prices and the industry may be closer to a price-taking. Under this scenario, changes in harvest in the EGSL would only impact industry profits (see Appendix C.1). With the same increase in oxygen saturation, the EGSL Northern Shrimp fishing zones would experience changes in profits ranging from -6% to 16%. The results presented in this analysis can, there, be viewed as a range of welfare impacts depending on the correct market characteristics, ranging from perfectly competitive to monopolistic competition.

Finally, another area of further research would be to analyze different quota schemes and the impact of changing the quota rule on the changes in prices and profits. This is important for policy implementation. As more scientific research emerges on the sources of nitrogen and phosphorus and relationship between nutrient pollution and oxygen concentration in the EGSL, policy recommendations can be more targeted to achieve the desired increases in oxygen concentration that will be welfare improving. While the results presented do not address the implications for resource rents, they add to the literature showing that under restricted regulated access, improvements in water quality can improve both stock status and economic welfare, at least in the short run.

Chapter 2

Quantifying the Impact of Nutrient Pollution in a Freshwater Fishery: The case of Yellow Perch in the St. Lawrence River, Québec

2.1 Introduction

Human activities have increased the flow of nitrogen and phosphorous over the earth and this has many bioeconomic consequences. With increased agricultural production and human and animal waste, we find that the water quality in many of the world's freshwater ecosystems, like lakes and rivers, are being degraded due to nutrient pollution, often resulting in eutrophication. Phosphorus and nitrogen are two of the most important nutrients needed for plant life but when loads in the water become too high, they throw off the balance of the food web, from plankton to fish, and the geochemical cycles in which these two elements participate. When these natural fertilizers are in excess, they can cause an overgrowth of organic matter, heavy algal blooms that can be toxic, dissolved oxygen depletion (hypoxia and anoxia) that can be lethal to fish and invertebrates, loss of marine habitat, changes in species distribution, and ultimate, loss of biodiversity (Abdel-Tawwab et al., 2019; Breitburg, 2002; Gannon et al., 2009; Heisler et al., 2008; Howarth et al., 2000; Council, n.d.; Nguyen et al., 2016; Small et al., 2014; Svendsen et al., 2018). The bioeconomic effects of eutrophication are poorly understood despite extensive research on many ecosystems. Eutrophication disrupts ecology through direct (e.g. on fish) and indirect (e.g. on habitat) pathways and some coastal ecosystems are more susceptible to eutrophication than other (Howarth et al., 2000). Although the biological impact of eutrophication has been widely documented (Desprez et al., 1992; Jacobson et al., 2017; Sandström & Karås, 2002; Giacomazzo et al., 2020), the economic impact of nutrient pollution has been primarily focused on the more visible consequences of eutrophication, such as harmful algal blooms (Jin et al., 2008; Kahn, 1988; Dodds et al., 2009; Sampat et al., 2021) or hypoxia and dead zones (Craig & Crowder, 2005; Smith, 2007; Smith et al., 2014; Massey et al., 2006; Purcell et al., 2017; Smith & Crowder, 2011; Huang et al., 2010; Huang & Smith, 2011; Huang et al., 2012) rather than the underlying cause (nutrient loadings) (Smith, 2007; Smith & Crowder, 2011; Simonit & Perrings, 2005; Knowler et al., 2001; Knowler & Barbier, 2005; Nguyen et al., 2016). In the limited analysis of the impact of nutrient loading on fisheries, we see that nutrient pollution while nutrient has led to changes in the food availability and even the overgrowth of predators in fisheries (Knowler et al., 2001; Knowler &

Barbier, 2005; Simonit & Perrings, 2005; Nguyen et al., 2016). Additionally, very little research has been done on the impact of nutrient pollution on freshwater fisheries (Simonit & Perrings, 2005) This represents a gap in the economic literature as a fishery can be impacted by nutrient pollution without a dramatic manifestation of its consequences. That is, nutrient pollution can still have a significant impact on a fishery even though there is no algal overgrowth or hypoxia (Mailhot et al., 2015; Giacomazzo et al., 2020). Even less analyzed is the impact of nutrient pollution on fish habitat. Recent biological literature has highlighted the impact of nutrient pollution on submerged aquatic vegetation, a key indicator of fish habitat quality (Giacomazzo et al., 2020; Hudon et al., 2012). However, the exact nature of this relationship between submerged aquatic vegetation and the key contributors to nutrient pollution differs among freshwater systems. Given that nutrient pollution can have a direct impact on fish and an indirect impact through the impact on their habitat, the consequence of nutrient abatement is uncertain, thereby opening a direct and indirect pathway to analyse the impact of eutrophication on fisheries. (Giacomazzo et al., 2020; Knowler & Barbier, 2005). This relationship between water quality and fish production, which is already complex at its core because of the influence of several factors over the lifecycle of a fish, can be further complicated by the institutions surrounding the fishery.

Fisheries are managed with a broad range of institutional structures and these impact both the biological sustainability and economic profitability of a particular fishery even with good habitat quality (Hilborn et al., 2005). Unregulated or open access fishery are characterized by the ‘tragedy of the commons’, where individuals maximize their own welfare leads to the overexploitation, given no restriction on effort (Hardin, 2019; Hilborn et al., 2005). This fishing pressure continues until there are no more profits to be obtained from the fishery as well as biologically unsustainable stock. In recognition of this, management measure such as gear restrictions and seasonal closure (Crutchfield, 1961; Hilborn et al., 2005) In such a regulated open-access fishery, the effects of the economic parameters (profits and effort) can be decoupled from the effects on biomass and as a result, long run biomass is usually higher than under open-access (Homans & Wilen, 1997). Other management measures include restrictions on entry (e.g. licences, ITQs) (Crutchfield, 1961;

Hilborn et al., 2005). Such limited access fisheries provide an even better outcome, where the fishery can be economically successful and at the same time, biomass can be maintained at the healthy level (Reimer & Wilen, 2013; Deacon et al., 2011). As such, the property rights surrounding a fishery will play an important role in determining how changes in nutrient loads impact the stock status in fishery and how the costs of nutrient pollution are distributed (Knowler & Barbier, 2005). With changes in habitat quality, the level of stock and the amount of effort required to maintain a biologically sustainable and economically profitable fishery will vary. As such, we can use the variations in stock and profitability to quantify the impact of eutrophication under different institutional structure.

This paper addresses the impacts of nutrient pollution on fish stock and commercial catch and the resulting welfare impacts in a restricted access freshwater fishery. Our interest is therefore, to identify the pathways through which variations in nitrogen and phosphorous concentrations, and their ratio, impact fish stock and to evaluate the impact on fishery rents in a restricted access freshwater fishery. This is important because nutrient pollution degrades water quality and key ecosystem services. Policymakers need to understand how changes in policy to address nutrient pollution can impact economic outcomes, and how these outcomes can further impact fishery resources, when designing policy tools. We hypothesize that the impact of nutrients on fish stock depends on the balance of nitrogen and phosphorous loads rather than the absolute loads.

The Lac Saint-Pierre Yellow Perch fishery receives nutrients enriched waters from several tributaries of the St. Lawrence River. In the Quebec region of the St. Lawrence River, the water quality guideline for protection of aquatic life is 0.03 mg/L of phosphorus and 0.63 mg/L of nitrogen. However, between 2015 and 2017, “phosphorus and nitrogen levels at the majority of water quality monitoring stations along the St. Lawrence River were above water quality guidelines more than 50% of the time" (Environment and Climate Change Canada, 2018). Water quality interventions implemented in 1972 have led to a reduction in phosphorus concentrations but nitrogen concentrations have continued to increase. This degradation in water quality has been linked to the poor stock status of freshwater fish species, namely the Yellow Perch (Giacomazzo et al., 2020; Hudon et al., 2012), through two pathways. The first pathway is a direct pathway by which excess nitrogen and phosphorus negatively impacts Yellow Perch at different life cycle stages and thereby abundance. The second pathway is an indirect pathway by which the nutrient imbalance caused by the reduction in phosphorus, and exacerbated by increase in nitrogen, has negatively impacted Yellow Perch habitat (by reducing submerged aquatic vegetation), such that a restoration of the nitrogen-phosphorus ratio (henceforth, N:P ratio) would prove beneficial to the fishery. A unique contribution of our study is the explicit inclusion of this indirect pathway, to better understand the effects of nutrient abatement decisions on ecosystems. Furthermore, the Lac Saint-Pierre Yellow Perch fishery is regulated by the Government of Québec, which implies different long run characteristics than under open access, so the model can also simulate specific welfare impacts of abatement policies and thus, guide fisheries decisions.

This paper contributes to the literature on the economic impacts of nutrient pollution on fisheries and in particular, freshwater fisheries. The aim is to contribute to filling the gap in research done in freshwater fisheries given that nutrient pollution impacts freshwater fisheries differently from coastal or marine fisheries. Additionally, by explicitly modelling nutrient loads as determinants of fish stock, we add to the literature that focuses on the underlying cause of variation in fish stock rather than their consequences. We develop a bioeconomic model that links fish production to the major nutrients used as indicators of water quality. First, we model fish abundance and survival as direct and indirect functions of water quality. Next, we model fish harvest as a being impacted by fish abundance and water quality, Finally, we quantify the effect of changes commercial harvest on welfare, based on the existing regulations surrounding the fishery, by examining commercial fishery profits.

The model is then applied to Lac Saint-Pierre through numerical simulations that recreate the paths of Yellow Perch stock and harvest under current nutrient loads. The results of of numerical analysis indicated that the N:P ratio may be more important for determining Yellow Perch stock than either nitrogen or phosphorus concentrations independently. Stock and harvest are further simulated under different policy scenarios where the N:P ratio is varied. Despite success in reducing nutrient pollution in Lac Saint-Pierre, policies implemented to reduce phosphorus concentrations have led to an imbalance in the N:P ratio that has negatively impacted fish stock and industry profits. We find that any further reductions in phosphorous concentration relative to nitrogen concentration negatively impacts fish stock and subsequent harvest and profits, while reducing nitrogen concentration is beneficial for increasing industry profits. On the other hand, if the simulations were taken to the extreme where phosphorus is increased, the results would indicate a phosphorus deficiency, which would not reflect the reality of Lac Saint-Pierre and instead corresponds to a strong imbalance in the N:P ratio. Simulations suggest that a 5% reduction in the N:P ratio would have still resulted in fishery closure after the 2011 season, however, there would be an increase in profits of CAN$16,566 from 2005 - 2011. With higher reductions in the N:P ratio, the fishery would have likely be kept open after 2011. Simulations suggest that a 20% fall in the N:P ratio would have led to a CAN$4 million increase in profits from 2005 - 2014. These reductions in the N:P ratio would involve implementing policies measures, comparable to those implemented for phosphorus reduction, to significantly reduce nitrogen loads. While this relationship is not proven to be causal (given data constraints), it reflects the complex ecological interactions between nutrients and the habitat composition, which is the second pathway that we emphasize.

The rest of this paper is organized as follows. Section II will briefly review literature on the economic impact of eutrophication on fisheries. Section III gives background to Lac Saint-Pierre, the case study area. Subsequently, we develop a regulated open-access bioeconomic model that exploits the direct and indirect pathways through which nutrient pollution has been found to impact the Lac Saint-Pierre Yellow Perch fishery. In section V, we parameterize the model using data from Lac Saint-Pierre and changes profits from nutrient reduction is calculated under different scenarios. In section VI, we discuss the findings from our policy simulations and the policy implications and conclude in section VII.

2.2 Literature Review

Eutrophication represents a change in the trophic state of water caused by excessive nutrient inputs, driven by land use trends, where large-scale commercial crop cultivation increases the use of external inputs such as fertilizers (Islam & Tanaka, 2004; Council, n.d.). Several studies have highlighted the relationship between nutrient loading, its consequences and fish stocks. One of the most research consequences of nutrient pollution is hypoxia. Hypoxia has become a major concern in fisheries given the persistent decline in oxygen in the Gulf of Mexico. Huang et al. (2010) quantified this effect on Brown shrimp in North Carolina fishery Brown Shrimp using data at the fisherman level (microdata). This unique approach allowed the authors to control for correlated factors and for spatial and temporal aspects of the ecosystem. The three step approach included linking the stock of brown shrimp and commercial harvest to environmental factors using the Von Bertalanffy Growth function. This allowed the authors to evaluate the impact of environmental factors on the growth of individual shrimp within the fishery. The authors provided conclusive evidence of the impact of of hypoxia on a shrimp fishery where previous literature had found that hypoxia had no severe economic impacts of hypoxia. The authors, however, focused on one of the consequences of nutrient pollution instead of actual nutrient loadings or concentration, which is the focus of this paper.

Knowler et al. (2001) studied the effects of phosphorus loading on an anchovy fishery in the Black Sea and Knowler & Barbier (2005) developed a dynamic, stochastic bioeconomic model to study the impact of nutrient loading on the same fishery under the probability of predator invasion. This model explicitly included nutrient pollution as a factor impacting recruitment. This model was additional novel due to the two assumed states of the world - one with no predator invasion and one with predator invasion - which would occur with different probabilities. The analysis included a numerical simulation of anchovy stock, catch, effort and profits under different nutrient pollution scenarios. This model explicitly linked changes in nutrient pollution and changes in anchovy stock and the impact on the profits from the fishery. Smith (2007) and Smith & Crowder (2011) carried out similar studies when valuing the ecosystem services provided in the Neuse River Estuary, a North Carolina blue crab fishery. The authors developed a bioeconomic model to investigate the impact of hypoxia on fishery rents. The relationship between nutrient pollution and hypoxia was explicitly modeled through the effects of nitrogen loading on algae growth, which resulted in hypoxia, and it was found that the present discounted value of fisheries rent would increase by US$ 2.56 million with a 30% reduction in nitrogen loading. Nguyen et al. (2016) implemented a more general approach that involved constructing a damage function for eutrophication that includes both the polluting sector and the fishery sector that are linked to each other by nutrient loading from the agriculture sector. The damage function was empirically applied to an East Baltic cod fishery in the Baltic Sea. Thanh (2013) carried out a similar bioeconomic study examining the impact of nitrogen loading on the East Baltic cod by maximizing the net present value (NPV) of the net benefits from the fishery. Compared to the above studies, our study aims to analyze the impact of nutrient pollution in a freshwater fishery.

Simonit & Perrings (2005) examined the impact of nutrient loading from fertilizer runoff from agricultural land on stock and harvest in a freshwater fishery in Lake Victoria. The fishery model uses Chlorophyll *a*[11] concentration as a proxy for phytoplankton density, which directly responds to nutrient loading. The paper also explicitly models the interaction between fertilizer runoff and Chl *a* concentration. To our knowledge, this is the only published paper that has examined the interaction between nutrient loading and fish stock in a freshwater fishery. The study found that the model with the environmental variable, linking fish stock and harvest to increased nutrient loading, better approximated the observed path of fish harvest than the standard catch-recruitment model. The model proposed in this paper also examines nutrient pollution in a freshwater fishery but takes a more direct approach, like Nguyen et al. (2016)

Emerging biological studies have highlighted the importance of examining the impact of nutrient pollution on submerged aquatic vegetation, which plays an important role in biogeochemical recycling in freshwater ecosystems. Submerged aquatic vegetation generally includes rooted vas-

---

[11] Chlorophyll used in oxygenic photosynthesis light harvesting as well as in converting energy of absorbed photons to chemical energy. One way we estimate the number of phytoplankton in the ocean is by measuring the amount of chlorophyll a in the water (Björn et al., 2009).

cular plants that grow up to the water surface (EPA, 2006) and support critical ecological services including shelter, habitat, and a food source for many organisms. As such, submerged aquatic vegetation abundance, which is an important component of aquatic systems, is a key indicator of water quality. Kahn & Kemp (1985) used a catch-recruitment model to estimate the relationship between submerged aquatic vegetation and fish stock and harvest. Submerged aquatic vegetation was modelled as impacting the carrying capacity of the fishing environment. A reduction in submerged aquatic vegetation was found to reduce not only the environmental carrying capacity but also the sustainable yield associated with each level of fish stock. Reduction in submerged aquatic vegetation abundance has been attributed to increased concentrations of suspended solids, herbicides, and plant nutrients (Kahn & Kemp, 1985). In particular, the abundance of submerged aquatic vegetation is associated with low levels of nutrients - good water quality (Y. Zhang et al., 2016). Eutrophication can shift the submerged aquatic vegetation abundance to the over production of phytoplankton and other harmful algae. The dome-shaped relationship between submerged aquatic vegetation abundance and nutrient concentration has been documented in many studies. Submerged aquatic vegetation abundance increases as water bodies move from nutrient-depleted to moderately enriched but decreases as nutrient concentrations further increase (eutrophication), while algal growth rates increase (O'Hare et al., 2018; Kahn & Kemp, 1985). Although there is clearly a relationship between nutrients and submerged aquatic vegetation, and the resulting impact on fish stock has not been empirically tested. The model presented in section IV can be used to empirically test both the direct and indirect relationship between nutrient pollution and fish stock and harvest.

2.3 Background to the Fishery

2.3.1 Study Area: St. Lawrence River

The Great Lakes-St. Lawrence River Basin is in the eastern region of North America and is the second-largest river network in North America in annual discharge, holding approximately 18% of the world's freshwater. It is a natural migration route of approximately 1,500 km that connects the

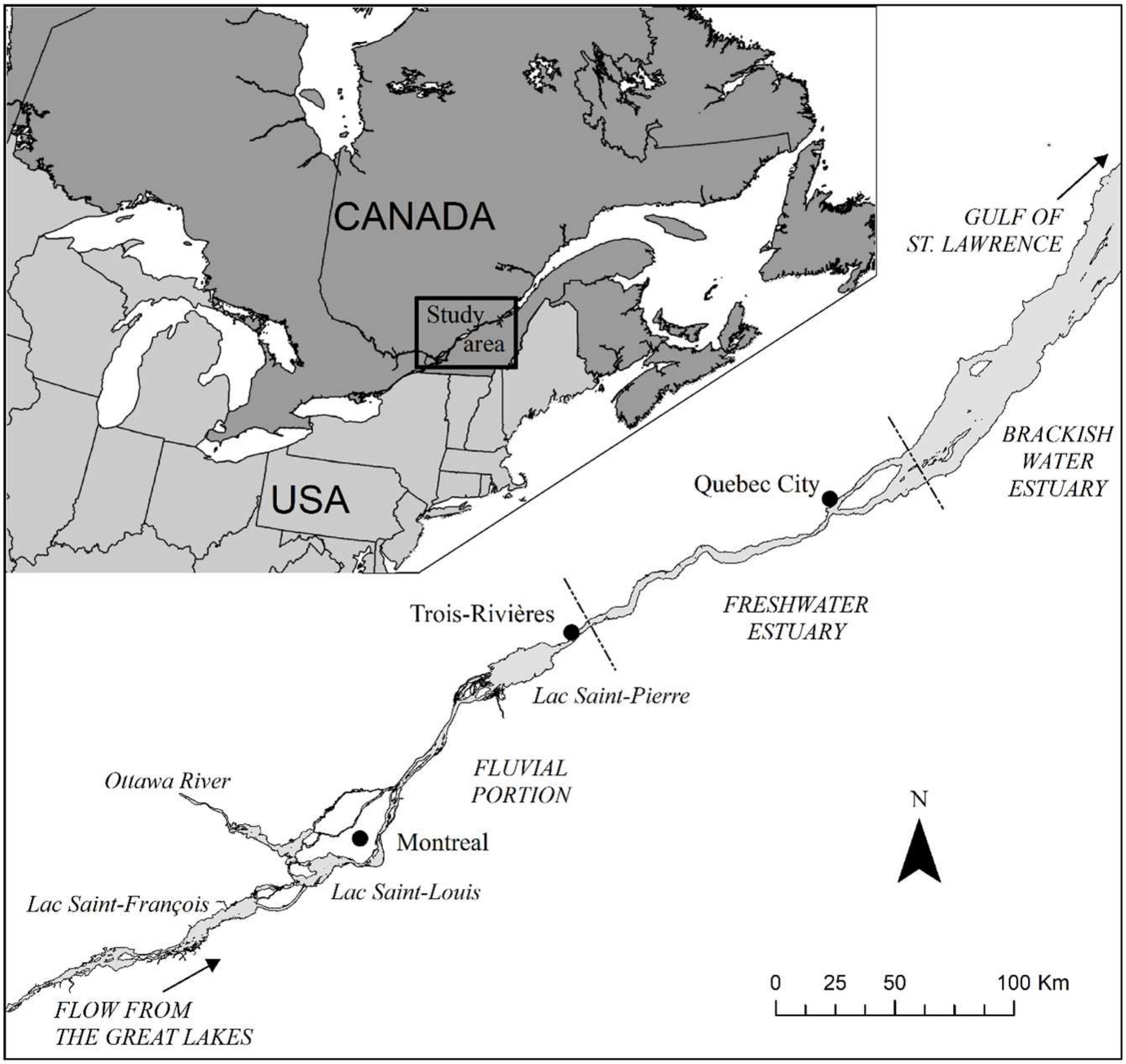


Figure 2.1: Great Lakes–St. Lawrence River basin and detail on the course of the St. Lawrence River.

Great Lakes to the Atlantic Ocean, which is also among the world's most important commercial waterways. The St. Lawrence River is generally divided into three main sections: a typically fluvial section, part of which will be the focus of this paper, extending from the outlet of the Great Lakes to Trois-Rivières followed by an estuarine section composed of a long freshwater estuary progressively influenced by the tide, which reaches a tidal range of almost 6 m nearby Quebec City, and brackish water (salinity 0. 2-25 ppm), and further out a marine section that joins the Gulf of St. Lawrence (salinity >25 ppm; see Figure 2.1). With its complex topography, the river portion presents a mosaic of varied aquatic habitats in which three large fluvial lakes with slow current (Lakes Saint-François, Saint-Louis and Saint-Pierre; Figure 2.2), follow one another, separated by narrow sectors with fast current and dotted with numerous islands and a few archipelagos. The water system includes numerous tributaries, which supply the St. Lawrence River with water of highly variable quality. In the river portion, these water bodies mix very little due to the absence

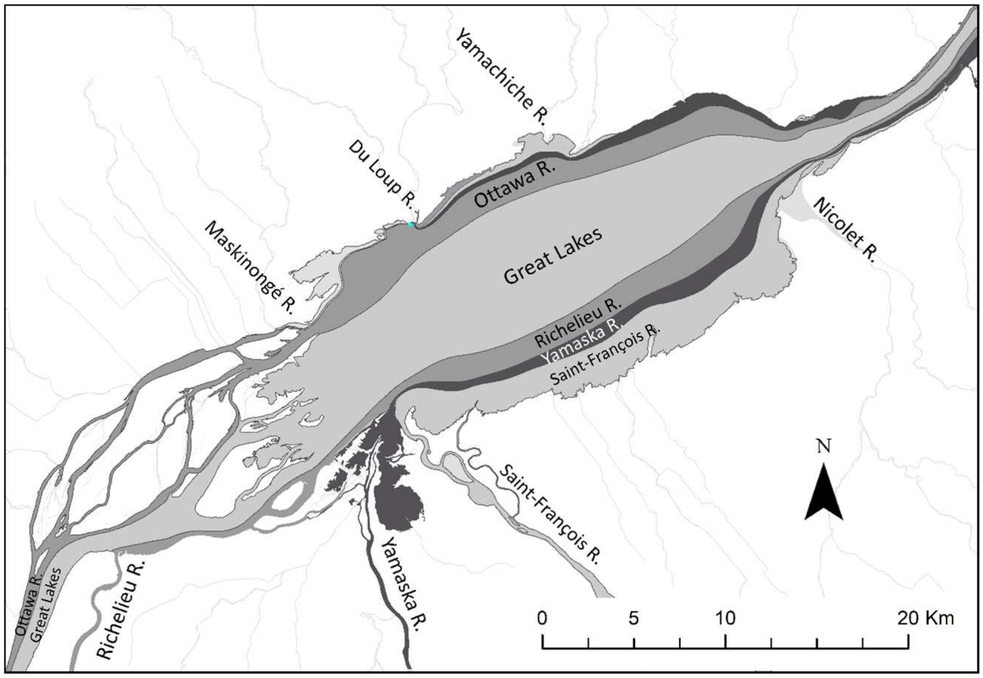


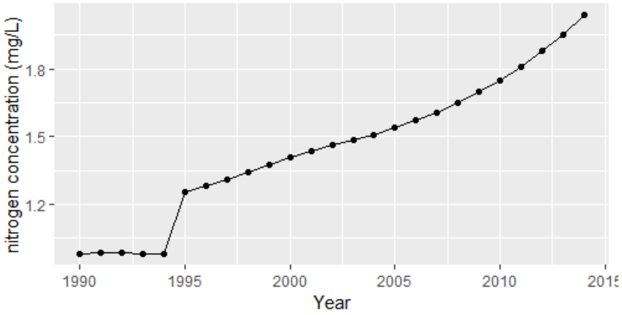
Figure 2.2: The study area is in Lake Saint-Pierre, the largest fluvial lake in the St. Lawrence River, where water quality is strongly influenced by the flow of several tributaries. These tributaries generate contrasting water masses, flowing side by side for tens of kilometers.

of tides and flow side by side for hundreds of kilometers. Water from the Great Lakes is clear, low in nutrients and flows into the deep central portion of the St. Lawrence River, where a navigation channel has been excavated. In contrast, water from other tributaries, which is brown in color, nutrient-laden, and sediment-laden, flows in the shallower portions between the center and the banks of the St. Lawrence River (e.g. (Frenette et al., 2006; Hudon & Carignan, 2008)). These impact the most productive aquatic habitats in the system. Lac Saint-Pierre (see Figure 2.2), which is the most downstream of the river portion, is among the most nutrient-rich receiving environments and is increasingly eutrophied. This creates an interesting study area for analyzing the impact of nutrient pollution on species and will be the focus of our analysis.

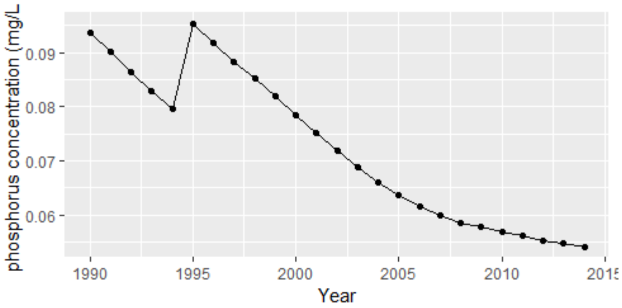
2.3.2 Water quality degradation

In recent decades, temperature, turbidity and nutrient concentrations have shown a marked increase in Quebec's waterways including the St. Lawrence River (Hudon et al., 2018). To reduce water pollution, the Quebec government adopted a law on environmental quality (1972), which was followed by interventions to reduce nutrient concentrations in aquatic ecosystems (Giacomazzo et al., 2020; Mailhot et al., 2015). These policies have been successful for phosphorus since between 1970 and 2016, a 2-fold decrease in their concentration in the waters of the St. Lawrence River was observed. However, during the same period, the concentration of nitrogen increased, which further unbalanced the N:P ratio (see Figure 2.3). At the same time in Lac Saint-Pierre, a significant increase in turbidity was observed, accompanied by a marked decrease in submerged aquatic vegetation (see Figure 2.4). The original submerged aquatic vegetation was replaced by a proliferation of cyanobacteria (harmful algae to the fishery) and some hypoxia episodes were evidenced in these St. Lawrence River receiving waters (Pick, 2016; Vis et al., 2008).

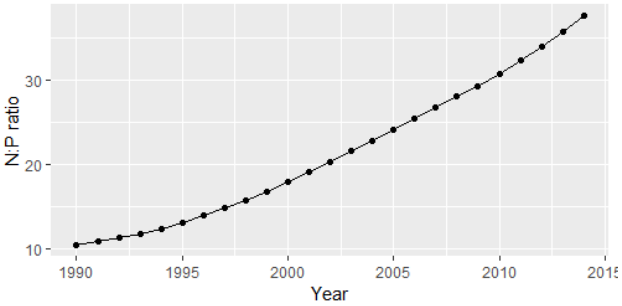
A few studies conducted in Lac Saint-Pierre have attempted to understand the complex mechanisms explaining the decline of submerged aquatic vegetation and the links with fish production (Giacomazzo et al., 2020). In their study of an excessively nutrient-laden Lac Saint-Pierre tributary, Huang et al. (2012) describes that high nutrient loads at the mouth of the tributary coupled



(**a**) Nitrogen concentration



(**b**) Phosphorus concentration



(**c**) N:P ratio

Figure 2.3: Nutrient concentrations in Lac Saint-Pierre 1987 - 2014

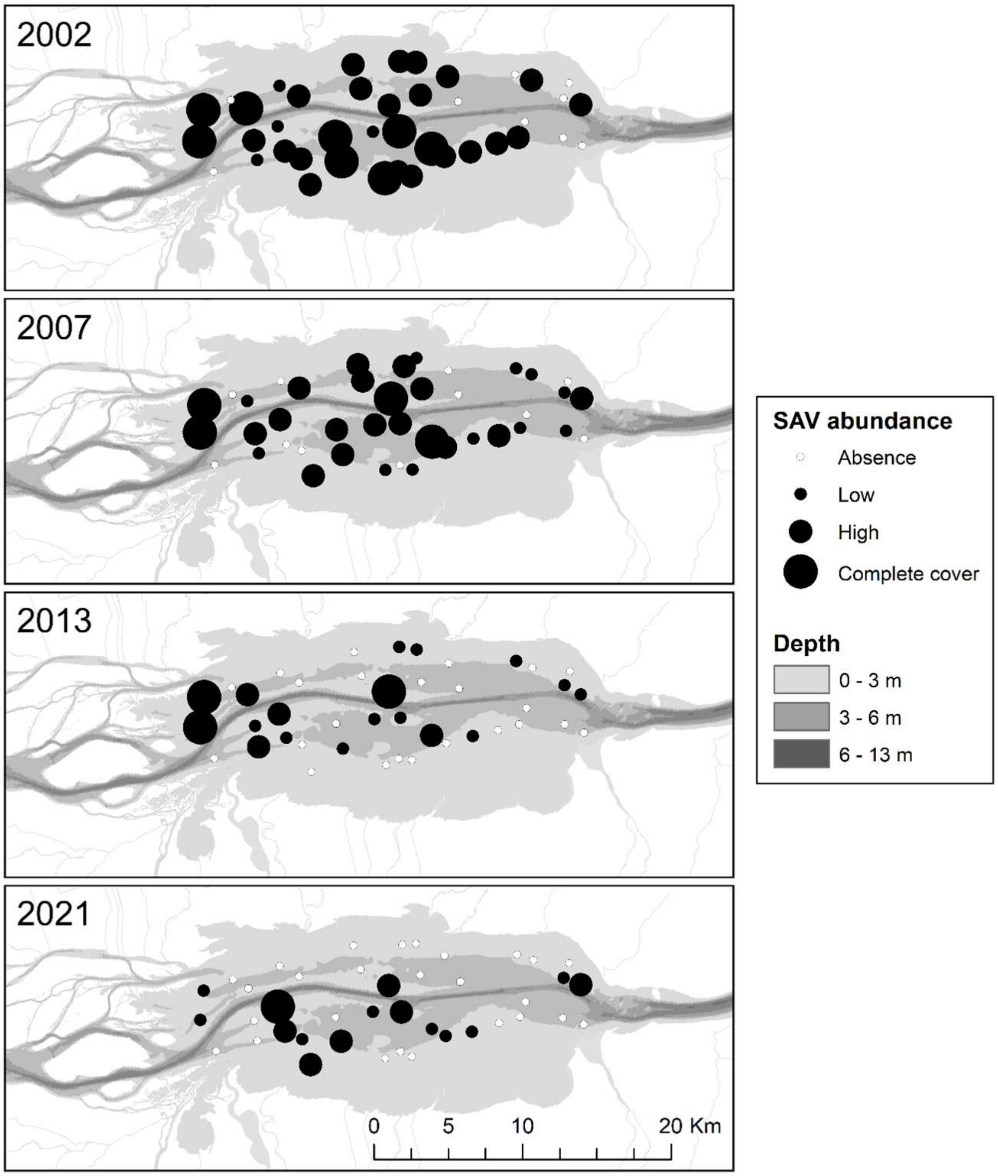


Figure 2.4: Changes in the abundance of submerged aquatic vegetation (SAV) in Lake St. Pierre over the period 2002 to 2021 (adapted from Magnan et al. (2022)).

with low water levels led to reduced submerged aquatic vegetation abundance in Lac Saint-Pierre. The passage of nutrient-rich water through the dense submerged aquatic vegetation beds, which acted as a purification system, led to a drastic reduction in nitrogen and phosphorus concentrations downstream of the lake, eliminating submerged aquatic vegetation and promoting the proliferation of cyanobacteria capable of fixing nitrogen and thus better adapted to these new oligo-mesotrophic [12] conditions (Huang et al., 2012). According to Hudon et al. (2012), localize submerged aquatic vegetation in the plume of this tributary (Saint-François River, Figure 2.1) has declined due to increasing nitrogen concentrations, which, in contrast to phosphorus, has not decreased. In addition, Giacomazzo et al. (2020) identified a strong positive relationship between phosphorus and submerged aquatic vegetation in Lac Saint-Pierre, as well as a negative impact of total suspended solids, indicating that low phosphorus concentrations may be the cause of the low submerged aquatic vegetation abundance. This is supported by the observed trends in nutrients illustrated in figures 2.5a and 2.5b. Dennison et al. (1993) found that the decline in submerged aquatic vegetation abundance was related to increasing amounts of nutrients and sediments in Chesapeake Bay, which is the more typical case. This implies that the impact on nutrients on submerged aquatic vegetation abundance is not quite clear cut, and may depend on the particular nutrients being considered and their relative concentrations. Therefore, when focusing on nutrient removal to return aquatic systems to 'clean' states, it is also important to examine the balance in the N:P ratio. Stockner et al. (2000) suggests that with the decline in phosphorus content of fertilizers, nutrients may need to be reintroduced to some aquatic ecosystems so as to return to a balanced N:P ratio and restore sufficient fisheries production levels. This highlights the role of maintaining a nutrient balance rather than absolute nutrient concentrations for the maintenance of productive fisheries, given that 'clean' water does not imply maximum productivity. Hudon et al. (2012) and Vis et al. (2008) attributed the increase in cyanobacteria in Lac Saint-Pierre to the N:P imbalance rather than absolute concentrations of the nutrients (trends between N:P ratio and submerged aquatic vegetation

---

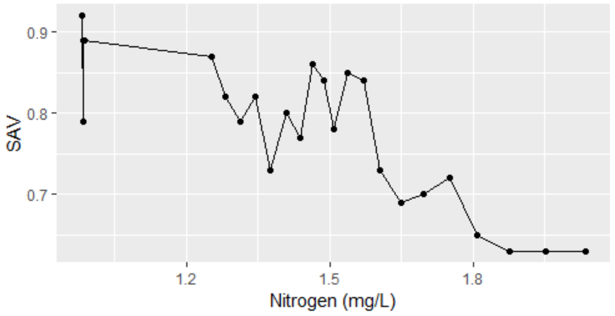
[12] We will classify oligotrophic as phosphorus concentration < 0.01mg/L and nitrogen concentration < 0.3mg/L, mesotrophic as phosphorus concentration 0.01 - 0.02 mg/L and nitrogen 0.3 - 0.5 mg/L and eutrophic as phosphorus concentration > 0.02mg/L and nitrogen concentration 0.5 - 1.5mg/L.

shown in figure 2.5c). This imbalance in the N:P ratio can indicate either a nitrogen or phosphorous limitation. However, the absolute concentration of phosphorus does not indicate a phosphorus limitation (Hudon & Carignan, 2008), and as such, the N:P imbalance is clearly due to excess nitrogen. In addition to the impact on submerged aquatic vegetation, nutrient pollution also has been found to have a direct negative impact on aquatic production in Lac Saint-Pierre and ultimately fish growth and stock abundance Giacomazzo et al. (2020); Hudon et al. (2012). Changes in nutrient concentration and imbalance in the N:P ratio have therefore had a significant impact directly on aquatic species and indirectly by altering their habitats. While other factors such as as turbidity and temperature may ultimately impact submerged aquatic vegetation abundance and fish growth (Giacomazzo et al., 2020), the focus of this paper is the impact of nutrient concentrations.

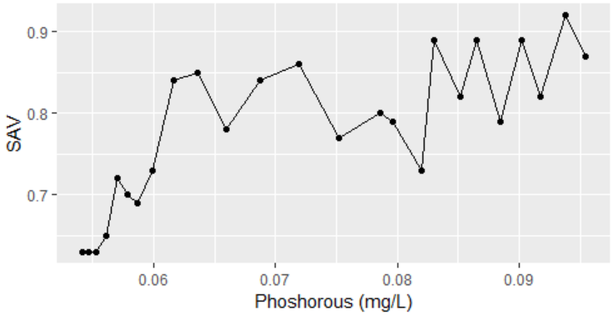
2.3.3 Yellow Perch Fishery

The Yellow Perch (Perca flavescens) is medium-sized species, common in the freshwater of the St. Lawrence River, widely known for its recreational and commercial values. It is a species considered adaptive, resilient and fertile , which was or still is dominant in the fish community of the St. Lawrence River depending on the sectors. Indeed, tagging, life history traits and genetic studies indicate that there are several independent populations in the St. Lawrence system (Mailhot et al., 2015). These local populations, which are exposed to different environmental conditions, have remained healthy in the upstream portion of the St. Lawrence system (Lac Saint-François and Lac Saint-Louis), while they have declined in the downstream portion (Lac Saint-Pierre and in the portion just downstream), making this an interesting situation to study from a scientific perspective.

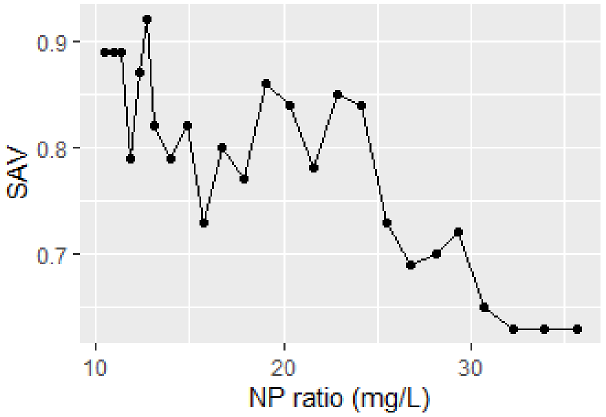
Once dominant in Lac Saint-Pierre, Yellow Perch showed signs of a marked decline in the 1990s. In less than 15 years, Yellow Perch has become a marginal species in Lac Saint-Pierre, resulting in the closure of commercial and recreational fisheries since 2012 (Mailhot et al., 2015; Magnan et al., 2022). The moratorium has been extended to 2027 and the stock status is assessed every 5 years using abundance of 1- and 2-years old Yellow Perch and healthy habitat (e.g. water quality and submerged aquatic vegetation). While Yellow Perch stocks have remained healthy



(a) Submerged aquatic vegetation and Nitrogen



(b) Submerged aquatic vegetation and phosphorus



(c) Submerged aquatic vegetation and N:P ratio

Figure 2.5: Relationship between submerged aquatic vegetation and nutrients

in the two upstream riverine lakes, Yellow Perch abundance in Lac Saint-Pierre declined by 70-86% between 1972 and 2011. Recreational fishing is allowed in the three largest riverine lakes in the St. Lawrence River, however, Lac Saint-Pierre (by the mid 1960's) was the only one that allowed a commercial Yellow Perch fishery. This fishery in Lake St. Pierre is a restricted access fishery, managed by the Quebec government, which monitors annual landings (commercial and recreational) as well as a 5-year monitoring of population dynamics (e.g. mortality, age structure, number of mature females). These indicators showed that the Yellow Perch stock was at risk and was subject to excessive fishing pressure. From 1997 - 2002, management policies regarding commercial fishing were implemented to reduce harvest and included minimum catch sizes and late season openings (Mailhot et al., 2015). In 2002, the first permit buybacks began and the number of permits was gradually reduced from 42 to 6 in 2008. Despite the reduction in fishing effort between 2002 and 2005, the Lake St. Pierre Yellow Perch stock continued to decline. In 2005, catch limits were introduced as an addition to bag limits and permit buybacks, but it became clear that factors other than fishing pressure were contributing to the stock decline. In 2012, a moratorium was put in place on recreational and commercial fisheries for a period of 5 years, which has been active since and was extended in 2022 for a third 5-year period.

It took several factors together to weaken the Lac Saint-Pierre ecosystem and especially Yellow Perch stock, recruitment, and key habitats. Studies following the moratorium cite overfishing, loss of spawning habitat through agricultural practices directly in the floodplain, water quality degradation in juvenile and adult rearing habitats, discharge regulation that cut off some of the spring spawning potential, predation by double-crest cormorant (de La Chenelière et al., 2014; Mailhot et al., 2015; Foubert et al., 2020). These studies also suggested that the introduction of exotic species and adverse climatic conditions should be considered as potential threats to the Yellow Perch stock. So, despite significant efforts that have eliminated fishing pressure, the Yellow Perch population is not recovering due to other factors such as nutrient pollution that further limit its recovery by directly affecting fish growth and survival and indirectly affecting their habitats. Figure 2.3 shows the trends in phosphorus and nitrogen loads from 1987 - 2011 in Lac Saint-Pierre. We see an increasing trend for nitrogen concentration in Lac Saint-Pierre while phosphorus concentration has been steadily declining and is close to the concentration that is cited as the most suitable to sustain aquatic life (Environment and Climate Change Canada, 2018). In a study by Giacomazzo et al. (2020) in Lac Saint-Pierre, Yellow Perch growth was found to be negatively associated increases in phosphorus and suspended solids. While no direct relationship was identified between nitrogen loads and Yellow Perch growth, Giacomazzo et al. (2020), however, found a positive relationship between the N:P ratio and Yellow Perch growth in Lac Saint-Pierre. This is despite phosphorus being a critical nutrient for growth in freshwater systems. Given the negative relationship between growth and phosphorus concentrations, this indicates that higher N:P ratio may be beneficial for Yellow Perch growth (Kolka et al., 2019; Giacomazzo et al., 2020), corroborating claims that it is the optimal balance between the macrominerals that is important for promoting optimal fish growth Storebakken et al. (2000). However, as mentioned before, submerged aquatic vegetation abundance in Lac Saint-Pierre is negatively associated with N:P ratio, and as such, the impact of increasing nitrogen relative to phosphorus loads would negatively impact habitat quality (through reducing submerged aquatic vegetation abundance) (Giacomazzo et al., 2020). This would then have a negative indirect impact on Yellow Perch stock, given the positive relationship between stock and habitat quality (Giacomazzo et al., 2020). As such, changes in the nitrogen and phosphorous loads may have an indeterminate effect on Yellow Perch stock, given the opposing impacts of a change in N:P ratio on Yellow Perch growth and habitat quality.

While Lac Saint-Pierre is the only of the three largest fluvial lakes in the St. Lawrence River that allows commercial fishing, the other fluvial lakes also experienced declining Yellow Perch stock over the same period. Factors including overfishing and pollution were cited for the loss of stock in Lac-Saint Louis. However, according to Mailhot et al. (2015), management measures that were put into place in Lac Saint-Louis to reduce chemical contaminant led to the recovery of the species in that lake. These management measures implemented under the 1970s Québec law on environmental quality resulted in the reduction of chemical contaminant by 50% to 90% since the 1980s. The hypothesis that the recovery of the species in the lake was due to reductions in fishing pressures was tested but this hypothesis was rejected (Mailhot et al., 2015) and it was concluded that the reduction in nutrient pollution positively impacted Yellow Perch stock in Lac Saint Louis. The trend in nutrients in Lac Saint-Pierre is similar to what was observed in Lac-Saint Louis, creating a likely scenario that nutrient pollution also caused the decline in Yellow Perch stock in Lac-Saint Pierre.

Given that nutrient pollution may have played a large role in the decline of the fish stock, this warrants quantifying the impact of nutrient pollution by estimating the loss in profits that was driven by nutrient pollution. In the following sections we will analyze the link between water quality, submerged aquatic vegetation and Yellow Perch stock to quantify the impact of nutrient pollution on Lac Saint-Pierre Yellow Perch fishery. The water quality variables identified as key stressors in LSP are phosphorus concentration, nitrogen concentration, turbidity (TSS) and water level (Giacomazzo et al., 2020). We hypothesize that the nitrogen phosphorus balance significantly impacts Yellow Perch by directly affecting growth and indirectly through the impact on submerged aquatic vegetation abundance. We expect increases in phosphorus concentration to negatively affect fish growth and positively affect submerged aquatic vegetation abundance in Lac St. Pierre, with the overall impact being determined by the relative concentration of phosphorous with respect to nitrogen (N:P ratio).

2.4 Bioeconomic Model

The aim of this paper is to quantify the effects of water quality on the Yellow Perch fishery, by examining changes in welfare arising from changes in nutrient concentrations. To do this, we assume that nutrient concentration impacts Yellow Perch directly altering fish growth and rate of survival and indirectly by altering fish habitat through changes in submerged aquatic vegetation. We also assume that the fishery operates under restricted access, which determines the economic impact of changes in nutrient pollution. Under these assumptions, we first develop a dynamic biological model that links the stock of Yellow Perch to water quality. The water quality variables of interest are nitrogen concentration (N), phosphorus concentration (P) and their ratio (N:P). We will add turbidity and water level and average annual temperature additional control variables.

We develop a model where the stock and recruitment in each period *t* are directly impacted by water quality, V. Adult stock, juvenile abundance and harvest in period *t -1* determine stock in period *t*. Water quality in period *t -1*, V_{t-1}, also impacts stock in period *t* directly and, indirectly through the impact on submerged aquatic vegetation S. In period *t*, the available stock and the allowable fishing effort determines the level of harvest and subsequent profits from the fishery. These key relationships are indicated in Figure 2.6 and are further described in the following bioeconomic model of water quality, stock, harvest and profits for Yellow Perch.

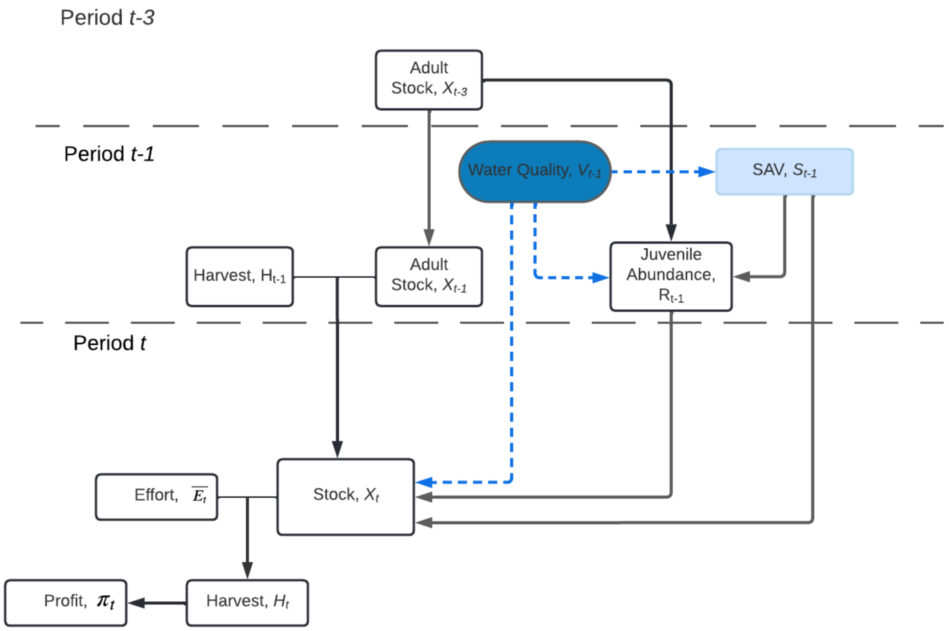


Figure 2.6: Conceptual model showing interaction between water quality, V, Yellow Perch stock and harvest.

We will begin by outlining production in the fishery using a stock-recruitment model, where regeneration of the fish population is dependent on previous fish stock and mortality. The stock-recruitment relationship associates stock size in each year with recruitment which results from the stock spawning two years prior. Let fish stock in the current year be recursively determined by the following equation:

$$X_t = (1 - \eta_{t-1})X_{t-1} + R_{t-1} - H_{t-1} + F(V_{t-1}, S(V_{t-1})) \tag{2.1}$$

where X_t is Yellow Perch stock measured in kg of biomass; η_{t-1} is natural mortality and $(1-\eta_{t-1})$ represents the probability of survival in the absence of fishing. X_{t-1} net of harvest, H_{t-1}, is a measure of escapement, representing adult stock that survived fishing mortality. R_{t-1} is juvenile recruitment to the stock in period t. Based on the evidence presented in Section 2.3, water quality, is assumed to affect current stock directly and indirectly through impacting submerged aquatic vegetation (S). These water quality effects on stock are reflected in the model through $F(V_{t-1}, S(V_{t-1}))$. The direct effect is captured through assuming $\frac{dF}{dV} > 0$, implying that improvements in water quality improves fish stock, and the indirect effect through $\frac{dF}{dS}\frac{dS}{dV} > 0$, which implies that water quality improvements increase submerged aquatic vegetation, which in turn positively impacts fish stock.

Follow Figure 2.6, juvenile recruitment is determined in period *t -3*. A separate equation will be used to describe the evolution of juvenile Yellow Perch as a function of adult abundance three(3) periods prior, and independent of current fish stock. Most widely used recruitment models include the Beverton-Holt Model (R. J. H. Beverton, 1954; R. Beverton & Holt, 2012) and Ricker Model (Ricker, 1954), which model recruitment as dependent on the density of the spawning stock (Subbey et al., 2014). These models assume either an asymptotic or dome-shaped relationship between recruitment and spawning stock. With no evidence from the biological literature on a density dependent relationship between Yellow Perch recruitment and spawning stock, we will use a simple density independent model to describe juvenile recruitment (Quinn & Deriso, 1999; Spigarelli et al., 1981). For the purposes of our model, we will assume that juvenile Yellow Perch join the adult fishable stock in period *t* . The number of recruits that joins fish stock year *t* is a function of the number of eggs produced by the adult Yellow Perch population and the probability of survival from eggs to recruits between periods *t-3* and *t-1*. Additionally, recruits are subject to natural mortality during settlement in period *t-1*. Following Spigarelli et al. (1981), this can be represented as:

$$r_{t-1} = EUB \times \frac{X_{t-3}}{2} \times e^{-M_1 \triangle d_1 - M_2 \triangle d_2 - \omega} \tag{2.2}$$

where r_{t-1} is juvenile abundance measured in numbers of Yellow Perch; X_{t-3} is adult Yellow Perch spawning biomass. *EUB* estimates the eggs per unit of biomass of adult Perch and $EUB \times \frac{X_{t-3}}{2}$ is the total number of eggs produced in the spawning period (Spigarelli et al., 1981). Eggs hatch about three weeks after spawning and are subject to mortality both during spawning and larval stages. M_1 and M_2 represent mortality at spawning and larval stages, respectively. ω is the natural rate of mortality of juveniles, which is assumed to be equal to rate of mortality of adult stock. Additionally, $\triangle d_1$ and $\triangle d_2$ represent the duration of time between spawning and larval and larval and young-of-year juveniles, respectively. Equation 2.2 gives the number of Yellow Perch in that will become juveniles in period *t-1*.

To convert the number juvenile recruits into biomass, we will model weight as a function of length using a allometric length to weight function. The length of Yellow Perch at different life stages is data that is easily collected. The relationship between body weight (W) and length (J) for nearly all species of fish can normally be represented by

$$W = \alpha J^{\delta} \tag{2.3}$$

where α is the intercept parameter or shape coefficient, δ is the allometric parameter and L is the length of age-1 Yellow Perch. The length of Yellow Perch can be modeled using the Von Bertalanffy growth function, which describes the growth of fish over their lifespan and has been found to give reliable estimates of Yellow Perch growth (Bisping et al., 2019; Power & van den Heuvel, 1999; Spigarelli et al., 1981). The length of Yellow Perch at any point in time can be calculated as a function of the maximum attainable length and the rate of approach to the maximum length [13].

---

[13] Length at age x can be calculated using Von Bertalanffy growth function where $J(x) = J_{\infty}(1 - e^{-V(x-x_0)})$ and J_{∞} = asymptotic length, V = growth constant x_0 = age at which length is 0 (assumed to be 0).

Given the age at recruitment and length of Yellow Perch at recruitment, we can convert numbers of recruits to recruitment biomass using the allometric length weight relationship. Juvenile biomass of Yellow Perch can be represented as follows:

$$R_{t-1} = r_{t-1}W \tag{2.4}$$

where R_{t-1} is biomass that will form a part of the next year's fishable stock. Equation 2.4 can be used to estimate recruitment biomass under the scenario that there are no water quality impacts. However, water quality affects survival of eggs and larvae in period *t-3*, young-of-year juveniles in period *t-2* as well as recruits in period *t-1*, directly and indirectly through the impact on submerged aquatic vegetation. It is assumed that juveniles are affected by water quality some time between spawning and age-1 (*t-2*) and during juvenile settlement in period period *t -1*. We will, therefore, model water quality as having a contemporaneous effect (impacting juvenile mortality during settlement) (Mailhot et al., 2015) and also a dynamic effect, (affecting juvenile mortality one period prior through spawning stock). Under this scenario, juvenile recruitment can be estimated using the following equation:

$$R_{t-1} = r'_{t-1}W \tag{2.5a}$$

$$r'_{t-1} = EUB \cdot \frac{X_{t-3}}{2} e^{-M_1 \triangle t_1 - (M_2 + G(V_{t-2}, S(V_{t-2}))) \triangle t_2 - (\omega + U(V_{t-1}, S(V_{t-1})))} \tag{2.5b}$$

The contemporaneous effects of water quality occur in period *t-1*, $U(V_{t-1}, S(V_{t-1}))$ and increase juvenile mortality above the natural rate of mortality, ω, and dynamic effects through the water quality function, $G(V_{t-2}, S(V_{t-2}))$ in the period between spawning and juvenile settlement.

To close the model, assumptions will be made about the evolution of effort. Following Deacon et al. (2011), the fisherman's production (*f*) is a concave function of capital and labor and the decline in stock is proportional to the rate of effort. In any current period *t* the Yellow Perch fishery employs capital *K* and labor *L* during a season of fixed length *T* to harvest the given (i.e. predetermined) stock of fish X_t. We assume $T \leq 1$, defines the season length as the fraction of the year that the season is open, such that, $f(L,K)T_t$ is total fishing effort over the year. It follows that the fishing mortality or the rate of harvest is determined by the fisherman's production. Given the effort and allowable catch restrictions, current harvest H_t can be depicted as

$$H_t = X_t(1 - e^{-f(L,K)T}) \tag{2.6}$$

where X_t is the fish stock at the beginning of the each year's fishing season. Total effort in the fishery in the current period can further be denoted as $E_t = f(L,K)T$ and equation 3.5 can be rewritten as $H_t = X_t(1 - e^{-E_t})$. In each period, firms incur labor cost only while the season is open but incur capital costs over the entire year. Total cost of fishing can be represented as a function of total effort

$$C_t = c(E_t) \tag{2.7}$$

A Québec regulatory body imposes restrictions on fishing to prevent over-exploitation of Yellow Perch stock in Lac Saint-Pierre. These restrictions include a licensing scheme and a quota, Q_t, on total allowable catch (TAC) implemented in 2005. By restricting the number of fishing licences, the regulatory body restricts the capital that enters the fishery. In this restricted access fishery, each fisherman would like to increase his capital and labour combination to increase his profits, but given the capital restriction, he is only able to increase effort by increasing his labour. However, the industry ability to increase effort depends on the substitutability between capital and labour. For example, in *Case 1*, if capital and labour are perfect substitutes, the restriction on capital will not impact the industry's ability to expand effort (Deacon et al., 2011; Reimer & Wilen, 2013). In this case, the industry effort will be the same as effort in an open access fishery. This arises through incomplete assignment of property rights, which is not solved by the implementation of fishery licences. The fisher's inability to claim property rights creates a disincentive to foregoing current harvest in return for greater future benefits of harvesting from a stock of greater biomass. As stocks are fished down, industry marginal cost of effort will increase causing profits to be dissipated. The industry effort will be the rent dissipating level of effort. Using p_t to denote price of harvested fish, industry effort can be obtained using the profit equation

$$\pi_t = pX_t(1 - e^{-E_t}) - c(E) = 0 \tag{2.8}$$

However, with binding regulations, biological stock under regulated open access will be greater than under open access, which allows harvest and total revenues to increase with effort.

In *Case 2*, if capital and labour are not easily substitutable (eg. if capital and labour must be used in fixed proportions), effort in the restricted entry fishery will be lower relative to open access. Fixing capital has the effect of increasing the marginal cost of effort, causing industry to reduce effort exertion, which allows the level of rent dissipation to be constrained. In this case, industry profits are

$$\pi_t = pX_t(1 - e^{-E_t}) - c(E_t) \lesseqgtr 0 \tag{2.9}$$

Additionally, with the implementation of the quota scheme in 2005, harvest is each period depends on the total allowable catch (TAC) quota restriction. While it is possible for actual harvest to fall below the limit on TAC, in practice, we find that actual harvest is usually very close to the maximum. As such, we will assume that $H_t = Q_t$, implying that quota is fully fished. It follows that equation 3.5 can be re-written as $Q_t = X_t(1 - e^{-E_t})$. This leads to an expression for effort as a function of the the current quota and stock

$$E_t = -ln(1 - \frac{Q_t}{X_t}) \geq 0 \tag{2.10}$$

Given the added quota restriction, the fishery resembles a regulated restricted access fishery. Profits are positive or negative in the short run and can be expressed using equation 2.9

Equations 2.1 - 2.10 details the bioeconomic model that can be used to determine the impact of changes in nutrient concentration on welfare in a restricted entry fishery. Following Reimer & Wilen (2013), in the next section we will investigate the two cases described above.

2.5 Numerical Simulation

The observed stock status and habitat quality of the Lac-Saint Pierre Yellow Perch fishery provides a reasonable illustration of the importance of understanding habitat-fishery linkages. To understand the potential effect of nutrient pollution on Yellow Perch, we conduct a series of numerical simulation using the bioeconomic model above. The numerical exercise allows us to explore the two pathways through which nutrient pollution may affect fish stock, and in turn, economic welfare: (1) directly through fish kills (hereafter direct effect), and (2) indirectly through reducing submerged aquatic vegetation, thus reducing food and shelter for species (hereafter indirect effect).

To carry out our numerical simulations we will use equations 2.1, 2.8 and 2.9 to examine the impact of changing nutrient concentrations on adult and juvenile Yellow Perch stock. Equations 2.5 and 3.5 defines variables R_t and H_t, and equation 2.10 specifies effort, E_t under the quota system. Before moving forward with parameterizing the model, we fist have to decide on the functional forms for our water quality (V) impact in equations 2.1 and 2.5. We assume that the water quality impact, V, is the cumulative impact of selected water quality indicators - nutrients (phosphorus and nitrogen), temperature and suspended solids. To partially define the between these water quality indicators and fish stock, we will follow Simonit & Perrings (2005); Nguyen et al. (2016), in assuming a quadratic relationship between nitrogen concentration and fish stock. We will further assume that phosphorus concentrations and submerged aquatic vegetation linearly impact stock. Thus, the equations below detail the direct and indirect impact of nutrients on fish stock. Recall equation stock equation, equation 2.1, where the current stock is directly and indirectly impacted by previous period water quality V_{t-1}, and represented as $F(V_{t-1}, S(V_{t-1})$. We define $F(\cdot)$ as a function of nitrogen, phosphorus, temperature and total suspended solids as follows

$$F(V_{t-1}, S(V_{t-1}) = N_{t-1} + P_{t-1} + Z_{t-1} + TSS_{t-1} + N_{t-1}^2 + S(V_{t-1}) \tag{2.11}$$

As described above, we further assume that water quality affects recruits at different points from spawning to settlement. From equation 2.5, recruitment is both a function of water quality in the period of recruitment, $t-1$, and spawning, $t-2$. The separate water quality impacts are represented in the recruitment function using $U(\cdot)$ and $G(\cdot)$, respectively, which are further defined as

$$U(V_{t-1}, S(V_{t-1})) = N_{t-1} + P_{t-1} + Z_{t-1} + TSS_{t-1} + N_{t-1}^2 + S(V_{t-1}) \tag{2.12a}$$

$$G(V_{t-2}, S(V_{t-2}) = N_{t-2} + P_{t-2} + Z_{t-2} + TSS_{t-2} + N_{t-2}^2 + S(V_{t-2}) \tag{2.12b}$$

Nitrogen (*N*) phosphorus (*P*) and total suspended solids (*TSS*) are measured in mg/L, and temperature (*Z*) is measured in degrees Celsius. From our water quality functions, in each period $S(V_t)$ is submerged aquatic vegetation abundance, measured as a fraction of the biological potential, and is a function of water quality. As such, water quality indirectly impact fish stock through its impact on submerged aquatic vegetation. In each period, submerged aquatic vegetation cover either increases or decreases with the concentration of nutrients and other water quality indicators, such that

$$S(V_t) = S(N_t + P_t + TSS_t + Z_t) = S_0 e^{N_t + P_t + TSS_t + Z_t} \tag{2.13}$$

The use of these indicators is based on the findings of Hudon et al. (2012) and Giacomazzo et al. (2020) as well as the general literature on the impact of nutrients on freshwater ecosystems (McDowell et al., 2020; Daldorph & Thomas, 1991). The literature suggests that changes in phosphorous and nitrogen concentrations and the resulting N:P ratio impact submerged aquatic vegetation abundance, and more specific literature on Lac Saint-Pierre suggests the inclusion of temperature as an additional indicator. Furthermore, water quality degradation is defined as an increase or decrease in these water quality indicators, resulting in a shift in trophic level (eg. from oligotrophic or 'clean' to eutrophic or polluted).

Finally, equations 2.8 and 2.9 define profits and consequently, effort E_t, under the case of rent dissipation and no rent dissipation, respectively. In order to obtain effort under these two cases, we need to make assumptions about the cost function $c(E)$ and the quota restriction. For simplicity we will assume that total costs are linear in effort such that $c(E_t) = cE_t$. Following (E. Barbier, 2019), we assume that the quota is a fixed proportion of fish stock in each period, $Q_t = qX_t$

We next proceed to parameterize our model. The parameters for our numerical simulation are obtained from the literature and estimation, where possible, and are described below.

2.5.1 Parameterization

Simulations are based on the Yellow Perch fishery in Lac Saint-Pierre, Québec. This fishery is the only commercial freshwater fishery in the Québec region of the St. Lawrence River, and has been indefinitely closed due to declining fish population. In this section we will also address the question of whether a reduction in the nutrients would have postponed or even prevented the closure of the Lac Saint-Pierre Yellow Perch fishery and estimate the potential impact of nutrient pollution on welfare. This fishery provides an excellent context for illustrating the value of water in aquatic systems. However, despite the observed decline in fish stocks since the 1980s, only limited biological data has been collected from the fishery. Given this limitation, estimation of the effect of nutrients on fish stock was not feasible and a numerical technique was instead employed.

Recall that we will use equations 2.1, 2.8 and 2.9 from our bioeconomic model to investigate the impact of nutrients on fish stock. This requires parameterizing these equations as well as subequations. Parameters for the direct impact of nutrients on Yellow Perch were based on estimations done by Nguyen et al. (2016), as their functional forms are similar to those used in the water quality functions presented in equation 2.12. Parameters for the impact of nutrients in the indirect effect were obtained from OLS estimation using data on submerged aquatic vegetation cover and nutrient concentration in Lac Saint-Pierre. submerged aquatic vegetation in each year was estimated by

$$S_t = \alpha S_0 e^{\beta_1 N_t + \beta_2 P_t + \beta_3 TSS_t + \beta_4 Z_t + \epsilon_t} \tag{2.14}$$

Parameter estimates for the relationship between submerged aquatic vegetation and nutrients (equation 2.14) are presented in Appendix B, which were determined from empirical analysis. Data for the empirical estimation of equation 2.14 were obtained form recorded observations of submerged aquatic vegetation and nutrients in Lac Saint-Pierre. Given the limited knowledge of the relationship between submerged aquatic vegetation and fish stock, we were unable to find estimates of the impact of submerged aquatic vegetation on Yellow Perch adult and juvenile stock, as described by the last term in equation 2.11. In our analysis, we randomly selected values from a standard normal distribution representing high and low impacts of submerged aquatic vegetation. Additionally, given the modest size of the fishery relative to other Yellow Perch fisheries in Canada, we assume that the industry is price-taking. Price in each year is assumed to be the estimated industry average price. This average price was estimated using producer price index and the actual price in 2011. Given the lack of information on production costs, total cost in each year was estimated as 70% of total revenues in each year (DFO, 2006a). Other biological parameters were taken from the literature on Yellow Perch, though this may not be specific to the Lac Saint-Pierre commercial fishery. Table 2.1 shows the key biological and economic parameters used in the simulation. Dynamic paths for Yellow Perch stock and harvest from 1990 - 2011 and 2012 - 2014 can be simulated using actual water quality data and effort from 1987 -2014 and initial stock and submerged aquatic vegetation cover from 1987, in addition to other parameters presented in Table 2.1.

Management measures implemented under the 1970s Québec law on environmental quality led to the reduction in phosphorus concentrations in Lac Saint-Pierre, making alternative policies to increase phosphorus untenable. We will therefore focus on policies aimed at changing phosphorus concentrations. To test whether a similar reduction in nitrogen concentration would have limited the decline in Yellow Perch stock and delayed the closure of the fishery, we begin by carrying out counterfactual simulations adjusting the nitrogen concentration while keeping phosphorus fixed at actual levels, to see the impact on fishery stock and harvest. We assume that the point of fishery closure is at a stock level equalling the actual 2011 stock. Additional simulations replacing nitrogen and phosphorus concentrations with the N:P ratio are then carried out to assess the importance of the N:P ratio in determining fish stock.

Table 2.1: Key parameters

Parameter	Symbol	Estimate	Source
		Biological	
Allometric weight	δ	0.03	
Natural mortality	η	0.13	Mailhot et al. (2015)
Average juvenile length	L	140mm	Mailhot et al. (2015)
Eggs per unit biomass	EUB	65316	Spigarelli et al. (1981)
Spawning mortality	M_1	11.51	Spigarelli et al. (1981)
Larvae mortality	M_2	5.5	Spigarelli et al. (1981)
1987 Stock	X_0	13,083,128kg	
Season length	T	0.67[a]	Mailhot et al. (2015)
		Environmental[b]	
Oligotrophic			
Phosphorus	< 0.01 mg/L	Istvánovics (2010); Beiras & Beiras (2018)	
Nitrogen	< 0.3 mg/L	Istvánovics (2010); Beiras & Beiras (2018)	
N:P	< 15		
Mesotrophic			
Phosphorus	0.01 - 0.02 mg/L	Istvánovics (2010); Beiras & Beiras (2018)	
Nitrogen	0.3 - 0.5mg/L	Istvánovics (2010); Beiras & Beiras (2018)	
N:P	15 - 17		
Eutrophic			
Phosphorus	> 0.035 mg/L	Istvánovics (2010); Beiras & Beiras (2018)	
Nitrogen	0.5 - 1.5mg/L	Istvánovics (2010); Beiras & Beiras (2018)	
N:P	> 17		
		Economic	
Cost of effort	c	67% of total revenues	
Average price	p	CAN$6/kg in 2011	
Quota restriction	q	0.03	

---

[a] Fraction of 12 months commercial fishing allowed. This represents the average length of season over the period of analysis.

[b] Nitrogen concentration above 0.63mg/L and phosphorus concentration above 0.03mg/L are considered excessive.

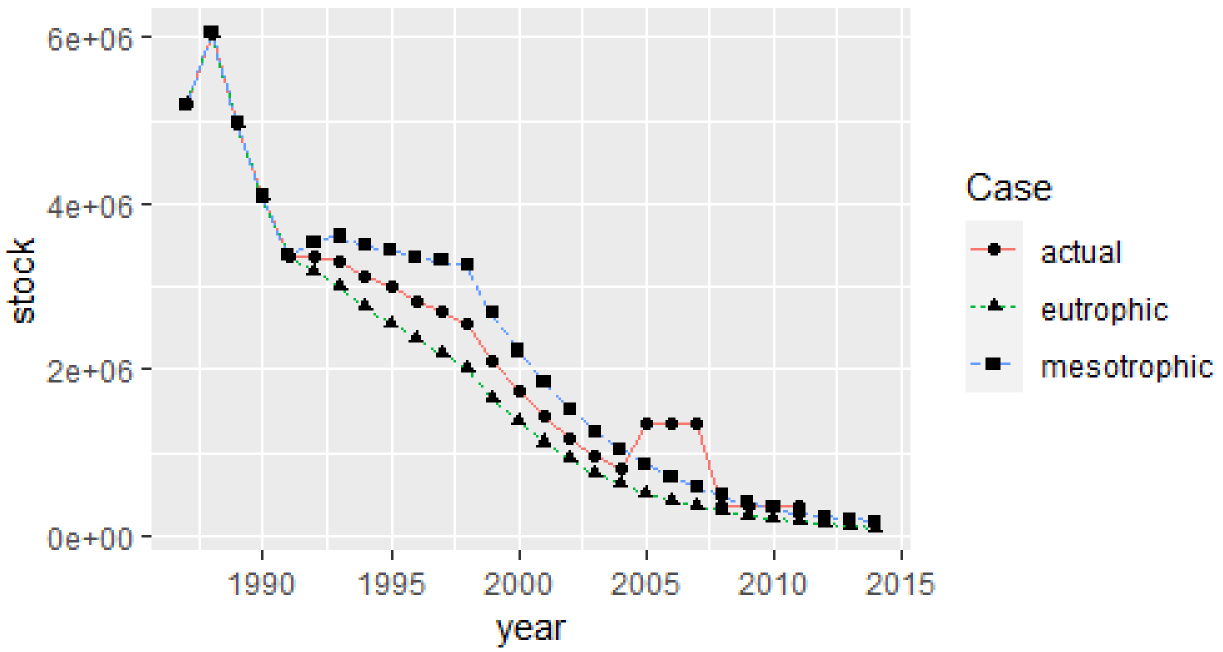
2.5.2 Numerical Results

To understand the role of nutrients in actual stock outcomes, we run sets of 28-year simulations of stock and harvest, from 1990 to 2014, using available water quality data from 1987-2014. Across each year in a 28-year simulation, we assume that variations in stock are due to variations in recruitment and nutrients. We run five sets of 28-year simulations under nutrient pollution scenarios ranging from severely eutrophic to mesotrophic. In our simulations, a lake is eutrophic if it is phosphorus limited. That is, given that phosphorus is considered the limiting nutrient in lakes, a lake is eutrophic if it has too high nitrogen concentration. From table 2.1, this implies a nitrogen concentration above 1.5mg/L. Alternatively, a mesotrophic lake is equally nitrogen and phosphorus limited. In other words, it has the optimal amount of dissolved nitrogen considered needed for growth of aquatic organisms. A mesotrophic lake has nitrogen concentration between 0.3mg/L and 0.5mg/L (see Table 2.1).

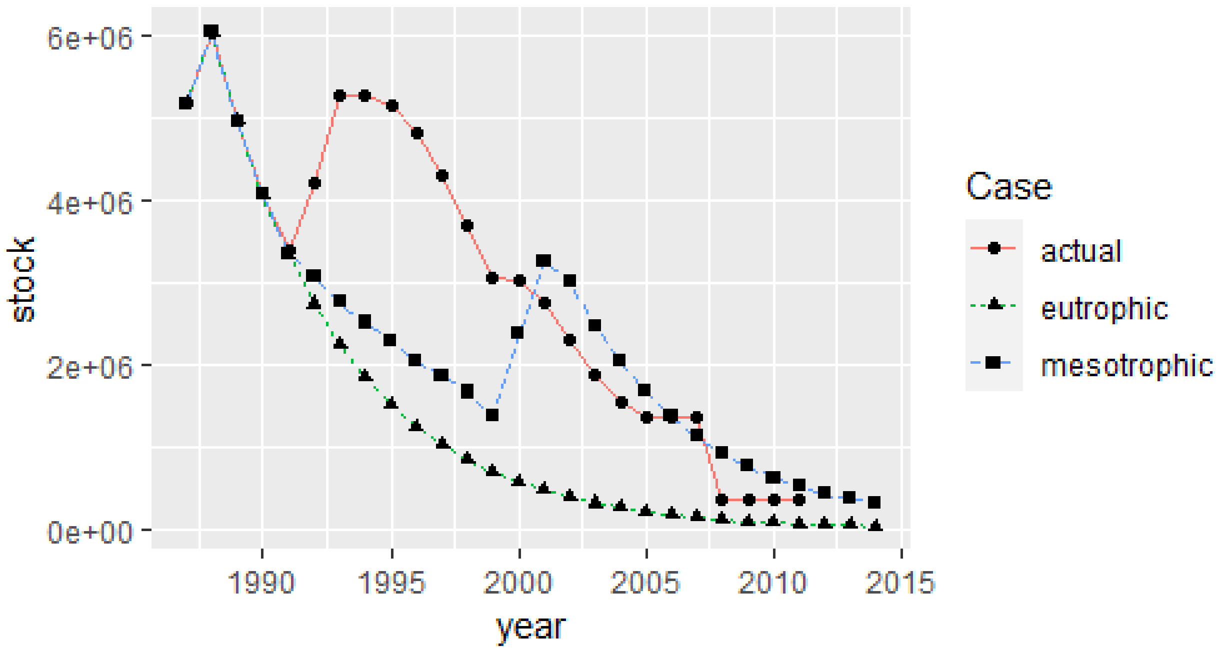
Figure 2.7 illustrates the simulated trends over 1990 to 2014 of the direct and indirect effects of nutrients on stocks under different nutrient pollution scenarios. In the top panel, we present an example of the evolution of stock from 1990 to 2014 under different nitrogen pollution scenarios (eutrophic, actual and mesotrophic). The 'actual' series shows the stock evolution under the existing nutrient conditions in Lac Saint-Pierre.

Figure 2.7 has a few implications. First, we see that actual stock is almost identical to stock under the eutrophic (nutrient pollution) counterfactual when nitrogen concentration is used as the nutrient pollution indicator. This implies that, using nitrogen concentration as the trophic level indicator, the current trophic state in Lac Saint-Pierre can be classified as being closer to eutrophic. We also see that a linear difference in stock between the actual and mesotrophic (no nutrient pollution) counterfactual. That is, in all time periods, stock is higher under no nutrient pollution that it is when there is nutrient pollution.

As a means of comparison we carry out a similar simulation where the N:P ratio is used instead of nitrogen and phosphorus. In this specification, the N:P ratio is assumed to have a quadratic direct effect and submerged aquatic vegetation exponentially responds to changes in N:P concentration.



(a) Stock comparison using nitrogen indicator



(b) Stock comparison using N:P ratio indicator

Figure 2.7: Snapshot of the combined effect (direct and indirect) in a Bioeconomic Simulation. Annual stock with nutrient pollution compared to no nutrient pollution.
Note: The top panel depicts total stock under the actual nitrogen concentrations, eutrophic nitrogen concentration (1.5 - 2.5mg/L) and mesotrophic nitrogen concentration (0.2 - 0.3mg/L) . The bottom panel depicts total stock under the actual N:P ratios, eutropic N:P ratio (25 -28) and mesotrophic N:P ratio conditions (13 -15).

Equations 2.11 and 2.12 can be re-written using N:P as

$$F(V_{t-1}, S(V_{t-1}) = NP_{t-1} + Z_{t-1} + TSS_{t-1} + NP_{t-1}^2 + S_{t-1} \tag{2.15}$$

and

$$U(V_{t-1}, S(V_{t-1})) = NP_{t-1} + Z_{t-1} + TSS_{t-1} + NP_{t-1}^2 + S_{t-1} \tag{2.16a}$$

$$G(V_{t-2}, S(V_{t-2}) = NP_{t-2} + Z_{t-2} + TSS_{t-2} + NP_{t-2}^2 + S_{t-2} \tag{2.16b}$$

where $S_t = S_0 e^{NP_t + TSS_t + Z_t}$. As before, S_t is empirically estimated and the parameters used in simulating fish stock when the N:P ratio is considered important. For our other water quality effects, estimates of the elasticities were taken from the literature. In our simulations, the N:P ratio is then adjusted to represent different trophic levels (eutrophic and mesotrophic). Here we find a more accurate representation of the evolution of stock in the fishery. Yellow Perch stock under the actual N:P ratio was higher than under the simulated mesotrophic conditions until 2000. This implies that that while the concentration of nitrogen was above the optimal concentration for a eutrophic lake, the optimal balance between nitrogen and phosphorus concentrations was maintained. Figure 2.7 further indicates that after 2000 there was a shift in the trophic state such that the stock fell below what would be achieved in a mesotrophic system. However, actual stock from 2001 - 2011 was higher than stock would have been under extreme eutrophic conditions.

To summarize, our simulations above examine changes in stock within severity of nutrient pollution. We find some evidence that there is a significant positive correlation between mesotrophic (optimal nutrient concentrations) conditions and stock when the N:P ratio is used as the indicator of nutrient concentration. We further find evidence that the optimal N:P ratio for submerged aquatic vegetation growth is higher than that needed for Yellow Perch growth. The overall impact of changes in the N:P ratio depends on which effect dominates.

2.5.3 Policy Simulations

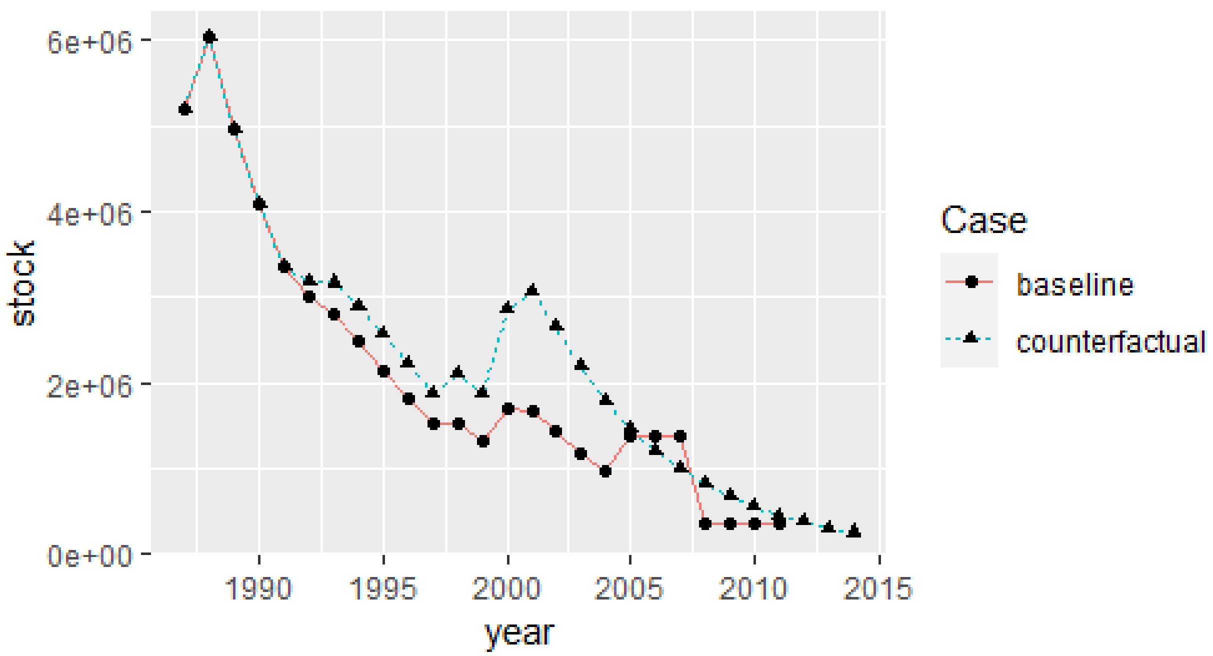
We can use our numerical simulations to address two policy questions: (1) could the implementation of policies to reduce the N:P ratio in 2005 delay or prevent the closure of the fishery in 2011? and (2) what is the welfare gain from reducing the N:P ratio in 2005? To answer both questions, we can examine how reducing the N:P ratio would impact stock and harvest under the scenario that profits are not dissipated[14]. We simulate stock from 1990 - 2014 under different N:P reduction scenarios and use the actual stock in 2011 to define the stock at which the fishery is biologically unsustainable and would require closure. To answer the second question, we then calculate the changes in profits from 2005 - 2014 (assuming that the fishery remained open after 2011). While lowering the N:P ratio would increase stock before 2005 (see fig 2.3c), the implementation of the quota on catch prevented the industry from maintaining previous catch levels despite a declining stock status. Additionally, in 2005, the N:P ratio was above 24 and continued to increase, causing Lac Saint-Pierre to transition to being hypereutrophic (severely phosphorus limited) (Barica, 1990).

Figure 2.8 shows the evolution of stock under different N:P ratio reduction scenarios. Our simulations suggest that if the N:P ratio was 5% lower than the actual in each year, Yellow Perch stock would have been higher in each year, except 2006 - 2007. If we assume that the fishery closes once the stock reaches the actual 2011 level, our simulations show that a 5% reduction in the N:P ratio would not have delayed closure. However, this reduction in the N:P ratio would have resulted in higher stock over majority of the period of analysis. Greater reductions would have been likely to delay fishery closure (see Appendix B). Figure 2.8 shows that a 10% reduction in the N:P ratio would have delayed fishery closure for at least an additional three years (see Appendix B).

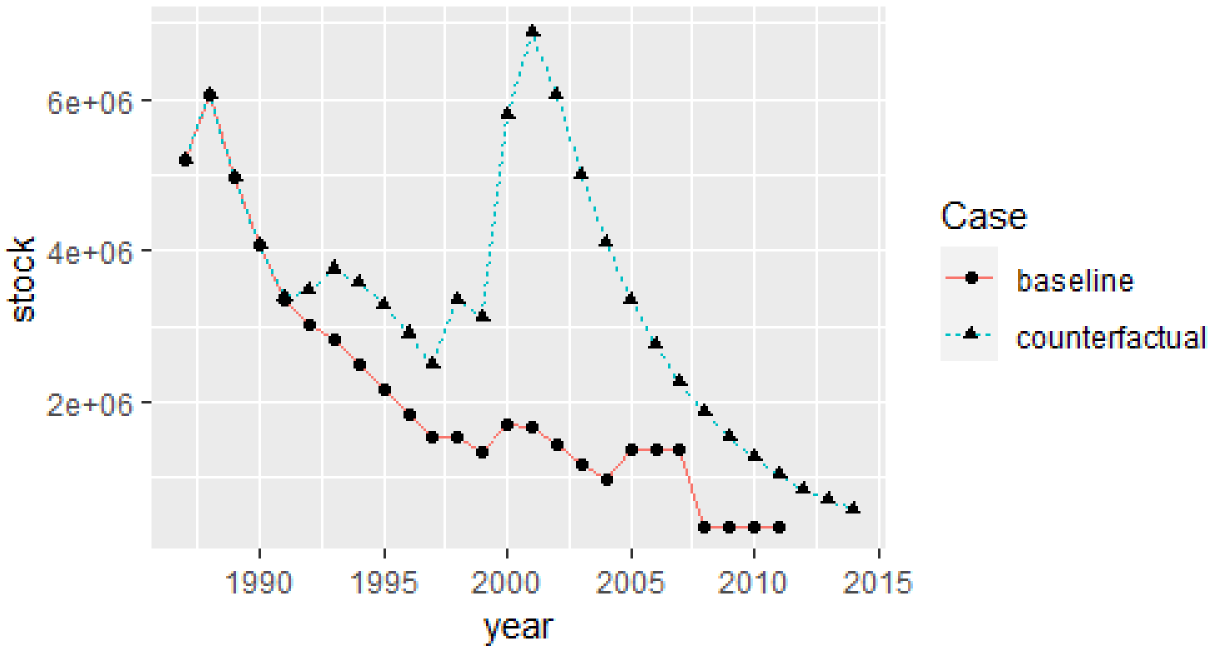
The welfare impact of reducing the N:P ratio can be determined by examining changes in net present value profits resulting from changes in the N:P ratio and subsequent harvest. This depends on the Yellow Perch market structure. Given the relatively small size size of the Lac Saint-Pierre

---

[14] We assume that the Lac Saint-Pierre fishers are price takers such that changes in the N:P ratio are likely to only impact profits. Under the scenario that profits are dissipated in all periods, there is no impact on profits.



(a) 5% reduction in N:P ratio



(b) 10% reduction in N:P ratio

Figure 2.8: Comparison of actual Yellow Perch stock against stock under different policy simulations. Note: The top panel (figure *a*) depicts total stock under the baseline scenario using actual N:P ratio versus stock under counterfactual simulations using 5% lower N:P ratios (counterfactual). The bottom panel (figure *b*) depicts total stock under baseline scenario using actual N:P ratio versus stock under counterfactual with 10% lower N:P ratios (counterfactual).

Yellow Perch harvest relative to the total output in the Canadian sector (Kinnunen, 2003), we assume that the fishery is price-taking. Changes in harvest will only impact profits, if any is made. Additionally, with the quota restriction implemented in 2005, the fishery begins to resemble a regulated restricted access fishery and assume that profits are represented by equation 2.9 with effort determined by equation 2.10. The results from 4 simulations are presented in table 2.2.

The sum of changes in net present value profits from reductions in the N:P ratio from 2005 to 2014 are shown in table 2.2. As expected, we see an increase in net present value profits with reductions in the N:P ratio. This, however, depends on the percentage change in the N:P ratio. A 5% reduction in the N:P ratio would have resulted in an increase in discounted [15] profits of CAN\$16, 566 [16] over the actual profits made in 2005 - 2011. The fishery, however, would have been closed in 2011 preventing additional profits from being made. A 10% reduction in the N:P ratio would have resulted in an increase in profits of CAN\$260,193 from 2005 - 2014. This represents 18.2% of actual 1990 - 2011 profits.

Table 2.2: Sum of changes in profits[a] 2005-2011 and 2012 - 2014 due to reductions in N:P ratio

Percentage reduction in N:P ratio	Sum of profits 2005 - 2011 (CAN\$)[b]	Sum of profits 2012 - 2014 (CAN\$)	Total change inprofits 2005 - 2014 (CAN\$)	% of actual 1990 - 2011 profits
5%	162,220	Closed	16,566	1.15
10%	371,368	34,480	260,193	18.2
15%	1,059,935	98,407	1,012,687	71
20%	3,813,614	354,060	4,022,020	281

[a] Profits are presented net present value profits in Canadian dollars (CAN\$) and are discounted back to 1990 using a 5% discount rate.

[b] In 1990, CAN\$1 = US\$ 0.86.

The results of the simulation support the findings by Hudon et al. (2012) on the impact of nutrient pollution in Lac Saint-Pierre. Here we find that a reduction in the N:P ratio would increase Yellow Perch stock and harvest and resulting fishery profits. This outcome supports the findings

---

[15] using a 5% discount rate

[16] CAN\$1 = US\$ 0.86 in 1990, not seasonally adjusted (Board of Governors of the Federal Reserve System (US), n.d.)

of Mailhot et al. (2015) that reductions in nutrient pollution in the St. Lawrence River can lead to the recovery of species. A major caveat to these results is the assumed importance of submerged aquatic vegetation to fish stock. We assumed a relatively low impact of submerged aquatic vegetation on Yellow Perch stock. For our simulations, we assumed that a one unit increase in submerged aquatic vegetation cover increase stock by 0.8kg. With higher elasticities, the overall impact of reducing the N:P ratio is lower, as the negative impact of higher N:P ratios on submerged aquatic vegetation becomes more dominant. Additionally, our chosen functional form, while guided by the literature, assumes that lower N:P ratios are always beneficial, and lead to higher stock. As such, our simulations do not consider the direct and indirect impacts of too little nutrients on stock.

2.6 Conclusion

The aim of this paper is to quantify the impact of nutrient pollution on the Yellow Perch in Lac Saint-Pierre. This is done by carrying out simulations to test two hypotheses: 1)the N:P ratio is an important indicator water quality in Lac Saint-Pierre and (2) improving water quality in Lac Saint-Pierre is welfare-improving. Current N:P ratios, which have very high values, could theoretically reflect either phosphorus limitation or excess nitrogen. In Lake Saint-Pierre, which is not phosphorus deficient, it is certainly a nitrogen excess. This highlights the importance of the nutrient balance when assessing water quality in freshwater systems. For Yellow Perch in Lac Saint-Pierre, further reductions in phosphorus concentrations below current levels may negatively impact fishery productive and welfare. This is because any further reductions in phosphorus will cause the N:P ratio to increase, which may be detrimental to fish stock. Instead, the N:P ratio should be reduced through lowering nitrogen concentrations. Simulations show that the overall impact of a reduction in the N:P ratio on Yellow Perch abundance and net present value profits depends on the magnitude of the direct impact of nutrients relative to the indirect impact of nutrients on fish stock. Even when low elasticities are chosen for the indirect impact, reductions in the N:P ratio lead to increases in stock. This implies that the negative indirect impact of high N:P ratios outweigh the direct benefits from higher ratios on the Lac Saint-Pierre Yellow Perch.

Given that reductions in the N:P ratio would positively impact fish stock, there are implications for fishery closure. Simulation results indicate that reductions in the N:P ratio of 5% would not have impacted the moratorium being implemented. From figure 2.8 we see that with a 5% reduction in the N:P ratio, Yellow Perch stock in 2011 would have been very similar to actual stock in that year. Assuming that the 2011 stock is the critical biomass at which fishery closure occurs, a 5% reduction would not have prevented fishery closure. However with a 10% reduction in the N:P ratio, stock in 2011 - 2014 would have been higher than actual 2011 stock and the life of the fishery would be extended, at least until 2014. Higher reductions in the N:P ratio would potentially extend the life of the fishery even longer.

While more work needs to be done to fully explain the welfare impacts of nutrient reduction in Lac Saint-Pierre, these results give some direction. From the results we see that the current trend in phosphorus concentrations has led to a scenario where further reductions will not lead to improvement of profits unless nitrogen concentrations are also addressed. The observed trend in phosphorus concentration has been as a result of policies implemented in Québec to reduce nutrient pollution caused by agriculture. These policies have cause a shift from using fertilizers with high phosphorus content to fertilizers with high nitrogen content (Giacomazzo et al., 2020). As a result, phosphorus concentrations have fallen to acceptable standards while nitrogen concentrations have continued to increase. Efforts should now be continued to ensure that nitrogen loads are effectively reduced. Given the potential loss in welfare from reducing concentrations of either nitrogen or phosphorus, nutrient reduction policies should target maintaining a nutrient balance rather than absolute nutrient loads. The implementation of agricultural policies to reduce the influx of nitrogen into the fluvial lakes, similar to those implemented for phosphorus, may achieve this goal.

While more work needs to be done to fully explain the welfare impacts of nutrient reduction in Lac Saint-Pierre, these preliminary results give some direction. From the results we see that the current trend in phosphorus concentrations has led to a scenario where further reductions will not lead to improvement of profits unless nitrogen concentrations are also addressed. The observed trend in phosphorus concentration has been as a result of policies implemented in Québec to reduce nutrient pollution caused by agriculture. These policies have cause a shift from using fertilizers with high phosphorus content to fertilizers with high nitrogen content (Giacomazzo et al., 2020). As a result, phosphorus concentrations have fallen to acceptable standards while nitrogen concentrations have continued to increase. Efforts should now be continued to ensure that nitrogen loads are effectively reduced. Given the potential loss in welfare from reducing concentrations of either nitrogen or phosphorus, nutrient reduction policies should target maintaining a nutrient balance rather than absolute nutrient loads. The implementation of agricultural policies to reduce the influx of nitrogen into the fluvial lakes, similar to those implemented for phosphorus, may achieve this goal. In Lac Saint-Pierre, 85% of total nitrogen loads are of agriculture origin (Hébert and Blais, 2017). At a finer scale, the improvements made in each of the watersheds could potentially be tracked individually in each of the Lac Saint-Pierre water bodies. Restoration of water quality and nutrient balance are essential conditions to rebuild the biological production of Lac Saint-Pierre and ultimately the Yellow Perch fishery. In parallel, the significant increase in turbidity has played an important role in the decline of SAV and the fishery (Giacomazzo et al., 2023), and this variable represents an important avenue for future research and regulation.

There are, however, some limitations to our analysis. The first is the lack of data on fish stock in Lac Saint-Pierre. Consequently, stock of Yellow Perch was estimated from catch per unit effort data and harvest was estimated using the assumed functional form. Additionally, we were unable to obtain accurate Yellow Perch price and cost data for each year. We were also unable to obtain biological data that gives estimates of the relationship between stock and SAV and nutrients. The results of our simulation and estimated welfare impacts are therefore sensitive to the selected parameters and assumed relationships, and may change significantly if the actually relationships vary significantly with our estimates. Nonetheless, our analysis provides the first account of the welfare implications of water quality degradation given its direct and indirect impact on fish stock.

Chapter 3

Estimating water quality impacts on snow crab in the Estuary and Gulf of St. Lawrence River (EGSL), Canada

3.1 Introduction

After the decline of the Atlantic cod stock in 1992, the commercial fishing industry on Canada's east coast underwent a period of change where groundfish has been replaced by shellfish as the main harvested species (DFO, n.d.-c). The snow crab (Chionoecetes opilio) is one of the most important crustaceans harvested species in Canada and in 2013 exports of snow crab was valued at $434 million, second only to lobster exports. The snow crab commercial fishery in the Estuary and Gulf of St. Lawrence (EGSL) began in the late 1960s, but in the early 1970s and again between 1987 and 1989, there was a significant decrease in landings. The 1987-1989 decrease was attributed to low crab recruitment in the late 1970s (DFO, 2019), which did not rebound in the 1980s. This was despite the management measures implemented by the Department of Fisheries and Oceans (DFO) when they assumed responsibility for the fishery in 1983. These management measures included issuing licences and in 1991, total allowable catches (TACs) were established in each area to limit catches. The Estuary and Northern Gulf of St. Lawrence areas are currently characterized by a stable or slightly decreasing fishable biomass. Overfishing was identified as the main contributor to low fish stock even in current periods, despite the implementation of licenses and quotas. However, the DFO recognized that the chosen quota level in each year may correspond to harvest that could lead to low biomass in the following year (DFO, 2015). This is given that snow crab stock is subject natural fluctuations over 8 to 12 year periods. During each period, recruitment fluctuates between consecutive low-abundance year-classes consecutive moderate-to-high abundance year-classes. As such, the quota chosen by the DFO may have been too high to account for the dips in recruitment, leading to overall declines in stock and landings. In addition to these factors that may lead to low biomass, water quality has been fluctuating due to both land-use change and climate change. These fluctuations include increased temperature and salinity and low dissolved oxygen concentration.

With fluctuating water quality in the EGSL, concerns have been raised about the link between declining water quality and the decline in snow crab abundance. The water quality indicators that have been identified as possibly impacting snow crab abundance include temperature, dissolved oxygen and salinity. Snow crabs prefer cold water temperatures and are most often found on sandy or muddy bottoms. They prefer saltwater with temperatures below 4°C and oxygen saturation above 70% (DFO, 2019). In the Gulf of St. Lawrence, a negative relationship has been identified between the temperature and snow crab depending on the stage in the life cycle (Marcello et al., 2012; Zisserson & Cook, 2017; Mao et al., 2019; Hardy et al., 1994). On the other hand, preliminary research into the impact of oxygen saturation on snow crabs has found that the species may be very tolerant to low dissolved oxygen (McLeese & Watson, 1968; Qyli et al., 2017). Despite the preferred level of dissolved oxygen, results from experiment showed that mean oxygen consumption declined below levels needed for normal activity below oxygen saturation of about 34% dissolved oxygen, implying a much lower biological threshold for snow crabs. However, again this may be the case only during particular life stages. Snow crab larvae and juveniles have been found to be less tolerant to higher temperatures, requiring temperatures lower than 4°C and having higher oxygen thresholds than adult snow crabs. In the EGSL, the impact of environmental factors on larvae and juvenile snow crabs has been posited as a possible explanation of low observed abundance even during the natural cycle (DFO, n.d.-c).

While the literature on water quality impacts on the snow crab is sparse, we have somewhat identified the channels through which dissolved oxygen impact snow crab abundance. In the early stages of the snow crab life cycle, we find that dissolved oxygen may negatively affect spawning activity by impacting whether spawners are able to produce or not, how many eggs they produce and larval survival. At the recruitment stage, this water quality variable impact the survival of juveniles by disrupting feeding activity. Finally, dissolved oxygen may impact the overall metabolic activity of adult crabs in general. The aim of this paper is to develop a model that can be used to quantify the impact of low dissolved oxygen (and hypoxia) on snow crab given the varying tolerance throughout the life cycle. This is important for several reasons. 1) snow crab is an economically valuable species in Canada and fluctuations in abundance with have welfare impacts for stakeholders and the economy at large. 2) The impact of low dissolved oxygen will vary depending on the stage in the life cycle. Adult snow crabs may be tolerant to low dissolved oxygen while juveniles may not. 3) Identifying the impact of water quality requires the use of a model that separates the impact on adults from the impact on juveniles. We will develop a stock-recruitment model that describes the relationship between adult and juvenile abundance and water quality. The model will be used to test the hypothesis that low dissolved oxygen (and hypoxia) is negatively associated with juvenile and adult snow crab abundance in the EGSL.

The rest of this paper will be organized as follows: Section 3.2 will give a brief background of the commercial snow crab fishery in the EGSL that will be analyzed, and the biological motivation for the hypothesis on the relationship between hypoxia and snow crab stock. Section 3.3.1 outlines the regulated open access bioeconomic model and describes the evolution of the juvenile and adult fishery in response to environmental changes. In Section 3.4, the bioeconomic model is empirically estimated using data from the snow crab fishery in the Estuary and Gulf of St. Lawrence (EGSL) and in section 3.5, numerical simulations are carried out to estimate the welfare impacts of declining oxygen in the EGSL snow crab fishery.

3.2 Background

3.2.1 Snow crabs and the environment

Snow crab (*Chionoecetes opilio*) is a crustacean belonging to the *Oregoniidae* family. Like other crustaceans, this crab grows by passing through successive moults during which it sheds its exoskeleton (shell) and absorbs water in order to increase the volume of its new shell. Moulting generally takes place between April and June. Newly moulted snow crabs are called “white crabs” because of the immaculate white colour of their abdomen. In both sexes, growth ceases for good after a so-called “terminal” moult, which occurs at variable sizes. Males with a carapace width of over 40 mm that have not yet undergone their terminal moult can be identified by their small claws and are called “adolescents” (or “sub-adults”). Males that have undergone their terminal moult can be identified by their larger claws relative to those of adolescents; they are called “adults”. Delays in moulting or skip moulting can occur in immature and adolescent crabs and these events appear to be related to density-dependent factors. The size of adult males varies between about

40 and 165 mm and adult females between 40 and 100 mm. As such, adult males are usually targeted for fishing, which creates a conflict between the fishery and the reproductive success of the stock. Snow crabs live for slightly more than seven years after their terminal moult and their shell condition changes during this period, first hardening for several months after moulting and then deteriorating in the last years of life. High-value commercial males are available to the fishery from about eight months to four years after the terminal moult, depending on the region.

Snow crab populations in the Estuary and northern Gulf of St. Lawrence show natural fluctuations in abundance over 8 to 12-year periods. There are consecutive years of low recruitment followed by years of moderate to high recruitment. The emergence of recruitment waves and troughs is not perfectly synchronous in the northern Gulf due to factors related to the productivity of the snow crab stocks and the natural environment. Environmental factors may impact the survival of larvae and juveniles, which could influence abundance. Stock assessments for snow crab that ignore these factors and focus on variations in biomass density rather than total biomass incorrectly absorb such environmental variations into catchability. With variable environmental factors, explicit modeling of habitat is necessary for abundance estimation. Two of the most important environmental factors that are thought to impact the Snow Crab in the EGSL are temperature and dissolved oxygen. Snow Crabs prefer the cold intermediate layer (CIL) in the Gulf of St. Lawrence and saltwater (salinity above 26% and temperatures below 4◦ C. However, a review of recent oceanographic data has shown that in recent years the CIL has been warming and this trend in warming may lead to changes in abundance in the long run. Additionally, the Gulf of St. Lawrence has been experiencing changes that have led to low dissolved oxygen. Snow Crabs prefer oxygen saturation above 70% and the current levels of dissolved oxygen may be detrimental to both adult and juvenile Snow Crabs.

Shortage of dissolved oxygen is a growing problem that has dramatic impacts on marine life and ecosystems. This decline in oxygen can be caused by climate change, since warmer water holds less oxygen, but is exacerbated by human influences. The amount of oxygen molecules that water can hold at a given temperature, pressure and salinity is defined as oxygen saturation, to which there is an upper limit. For example, seawater at 10°C with a salinity of 35 and at normal atmospheric pressure is considered fully saturated (100%) when the dissolved oxygen concentration reaches 6.4 millilitres per litre (mL/L) of seawater, and 50% saturated when it reaches 3.2 mL/L. Hypoxia is considered to be oxygen saturation below 40%, and seawater with oxygen saturation below 20% being considered severely hypoxic. Historical data indicate a decline in dissolved oxygen in the deep waters of the Gulf of St. Lawrence since 1932, with the lowest levels of dissolved oxygen being recorded in the Laurentian Channel, where oxygen saturation has been routinely been in the range of 20% saturation since the mid-1980s. About one third of this decline since 1932 has been attributed to increased oxygen demand in the deep water due to higher bacterial respiration and sediments. Since oxygen is essential for aerobic metabolism, marine life can suffer impacts can suffer including reduced growth rates, lower reproductive success and higher mortality in even mildly hypoxic waters (30 to 50% oxygen saturation). Preliminary research in the Estuary and Gulf of St. Lawrence suggests that snow crab is tolerant to hypoxia, however, very little is known about the effect of hypoxia on crustaceans in the Estuary and Gulf of St. Lawrence (DFO, n.d.-a).

Biological literature on the relationship between environmental factors and snow crab is sparse. However, studies of other crab species show that they are impacted by variations in dissolved oxygen at different stages in the life cycle.

Tomasetti et al. (2018) investigated the individual and combined consequences of low dissolved oxygen (DO) and low pH on blue crab larvae survival through a series of short-term experiments. Experiments found that low dissolved oxygen and low pH significantly reduced larval survival. In addition, if low dissolved oxygen falls below a minimum threshold, it can cause death of the blue crab population. Low pH had no effect on blue crab and there was no interaction between dissolved oxygen and pH. While severe hypoxic events may lead to death of species, some species are able to withstand periods of hypoxia at different stages in the life cycle. Blue crab larvae are more resistant to low dissolved oxygen concentration than new juveniles, and are less likely to succumb to the absence of oxygen (Tankersley & Wieber, 2000). Adult blue crabs are generally considered tolerant of hypoxia. Despite their tolerance, adult blue crabs may experience stress in the form of reduced growth, feeding, locomotion, or fecundity (Tankersley & Wieber, 2000). Furthermore, lowered dissolved oxygen alters behavior, which can make some organisms more vulnerable to predation.

Similarly, stone crabs show increased sensitivity to decreases in dissolved oxygen (DO) concentrations though this is dependent on their size during juvenile stages. Smaller juveniles exhibit high rates of mortality when exposed to severely low dissolved oxygen concentrations, while larger juveniles show signs of physiological stress demonstrated by an increase in anaerobic metabolism (Gravinese et al., 2020). According to Bao et al. (2020), the negative effects of hypoxia on species should not be ignored as chronic hypoxia impacts the physiological responses of species and further increases the risk of pathogen infection.

3.2.2 EGSL Snow crab fishery

The Snow Crab fishery in the Estuary and Gulf of St. Lawrence (EGSL) comprises of areas 12A, 12B, 12C, 13, 14, 15, 16, 16A and 17 (see Figure 3.1) that are managed by the Department of Fisheries and Oceans (DFO), Canada. The fishery began in the late 1960s and has experienced fluctuations in landings throughout its lifetime. Fluctuations in the late 1980s were attributed to recruitment failure in the late 1970s but landings rebounded in in the early 1990s to reach a record high in 1995. The Quebec Department of Agriculture, Fisheries and Food was responsible for management of the Snow Crab fishery until 1983, when the DFO assumed responsibility for managing this fishery. This was followed by a number of management measures that were intended to limit fishing effort. Licences and quotas on total allowable catches (TACs) were established in each area in 1991 to limit catches. Stock assessments are used in the process of determining the quota restriction and are adjusted annually in each area based on resource fluctuations. Despite quotas on TAC, in 2003 a reduction in quotas was imposed due to overfishing and this led to a significant decrease in harvest (DFO, 2019). Additionally, moratorium was imposed on snow crab fishing in Area 13 in 2003 due to a decline in catch per unit effort (CPUE) and in average crab size. Stock status was monitored and the moratorium was eventually lifted in 2008. Data from the fishery indicates that, on average, harvest was equal to the quota in each year from 2000 - 2016, with the exception 2000 - 2003.

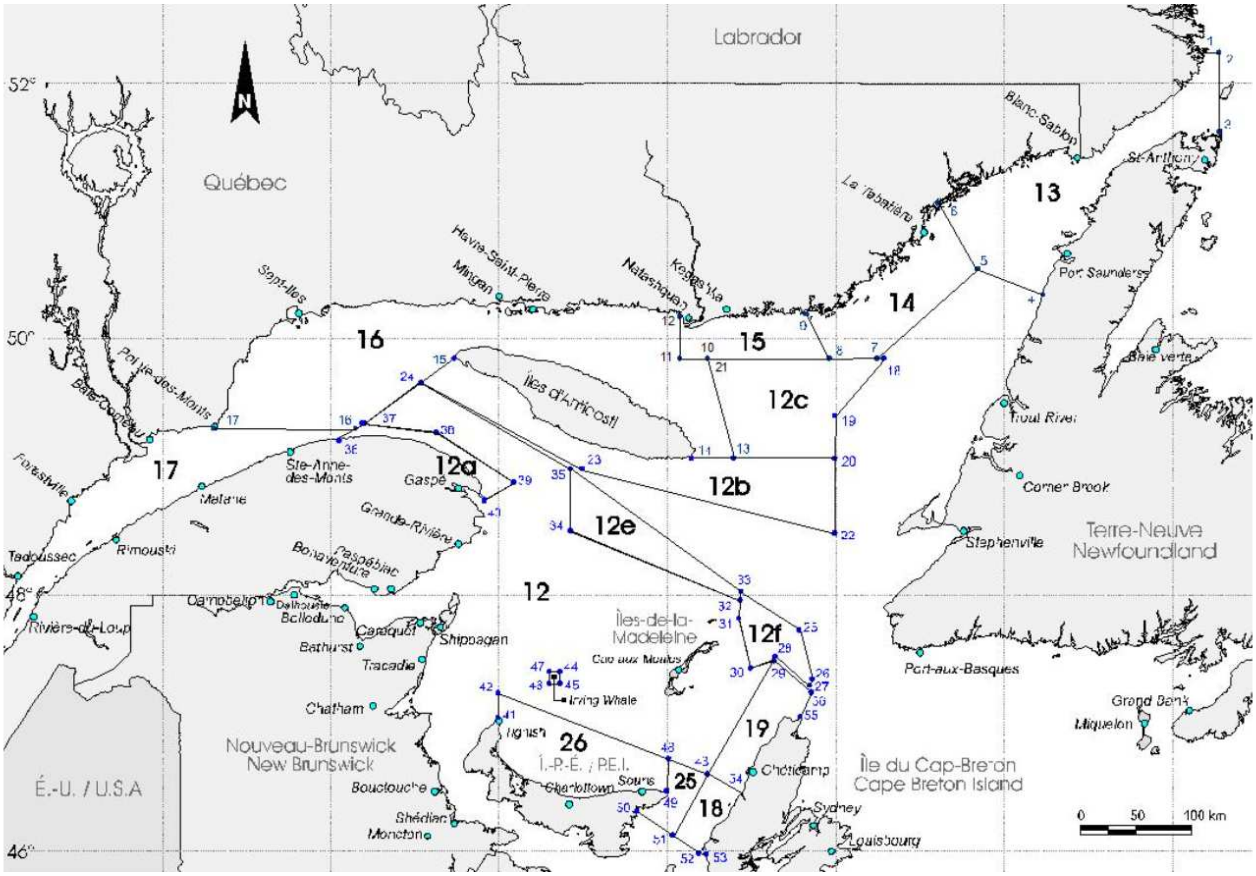


Figure 3.1: EGSL snow crab fishing zones 12A, 12B, 12C, 13, 14, 15, 16, 16A and 17 ; Source: Snow crab - Estuary and Northern Gulf of St. Lawrence Inshore Areas (12A, 12B, 12C, 13, 14, 15, 16, 16A and 17), 2021

The Canadian snow crab fishery is one of the three most valuable marine species in Canada, averaging CAN\$586[17] million in 2010, and has been the world's largest single supplier of snow crab. However, the EGSL only supplies approximately 15% of the total snow crab volume (DFO, 2019). The value of snow crab from the EGSL has been declining over time but recent data shows a significant increase in value during 2016 and 2017. This increase in value has been attributed

---

[17] CAN\$1 = US\$ 0.97 in 2010

to an increase in landing as well as increases in the average landed price. Canadian snow crab is both consumed domestically and traded internationally, and as such, the average landed prices for snow crab in Quebec and Canada generally follow the price trend in a reference market. For example, in recent years, the U.S. supply of king crab and snow crab has been shrinking resulting in a decrease in the global supply of snow crab. This resulted in an increase in the demand for Canadian snow crab an increase average landed prices (DFO, 2019). A large proportion of snow crab landed in the EGSL is exported by processing plants, and 93% of this export goes to the the U.S. This implies that exports are highly dependent on the U.S market and prices in the EGSL are sensitive to fluctuations in U.S crab supply. In coming years, the Canadian snow crab industry is expected to face increased competition from Norway and Russia, which is expected to depress international prices and further impact prices faced by fishers in the EGSL. The combination of these factors implies that the industry is price-taking, having very little control over price trends.

The snow crab fishery in the EGSL is a significant contributor to employment in the region but fluctuations in water quality can have significant impacts on fish production, making the job market precarious. Fluctuations in water quality will have significant impacts on welfare and understanding this impact is of interest to fishery managers and other stakeholders. The model developed below describes the relationship between Snow Crab abundance and dissolved oxygen, based on the biological literature on this relationship from other crab species. We will then empirically test this model using data from the Snow Crab fishery in the EGSL and carry out policy simulations to determine the welfare impact of variations in oxygen saturation.

3.3 Bioeconomic Model

The model developed will highlight the main relationships between stock and water quality as identified in the literature and will specifically reflect snow crab fishery divisions in the Estuary and Gulf of St. Lawrence (EGSL). Snow crab grows by passing through successive moults during which it sheds its exoskeleton (shell) and absorbs water in order to increase the volume of its new shell. Females spawn and eggs are carried by mature females for up to 2 years depending on food availability and environmental factors. Eggs are hatched from April to June when the larvae become pelagic and then the larvae settle to the bottom in autumn to winter. Once settled to the bottom, snow crab grow rapidly, moulting (shedding shells) approximately twice a year. The first inter-moult stage can be referred to as the pre-recruit stage. At about age 4, snow crabs reach a carapace width of 12mm and this will be used as the age of recruitment. This implies that there is an approximate 4-year lag between spawning and recruitment (Zheng & Kruse, 2006). In the period t-5, spawning is assumed to be affected by water quality. Spawning in period t-5 further determines the number of juvenile recruits in period t-1. In each period t-1, water quality determines the survival of recruits as well as adult abundance. In period t, adult abundance is determined by adult stock from previous period that survived fishing mortality as well as juveniles that have reached maturity. In each period, adult stock is impacted by water quality.

3.3.1 Modelling Strategy

Based on the biological literature, we assume that water quality impacts snow crab at different stages in the life cycle (adult, juvenile and larvae). Adult snow crab abundance in each period can be described as a function of current water quality variables, *V*, stock from the previous period that did not naturally die, X_{it-1}, recruitment, R_{it-1}, and previous period harvest, H_{it-1}, such that,

$$X_{it} = f(V_{it-1}) + (1 - M)X_{it-1} + R_{it-1} - H_{it-1} \tag{3.1}$$

Where i is index for the zone and t is index for the time period, measured in years. For simplicity, natural mortality (*M*) is assumed to be constant. X_{it} from the stock equation will be a measure of mature male and female snow crabs, measured in kilograms. Current snow crab stock is determined the amount of snow crabs from the previous period that survived natural mortality and were not harvested plus recruitment. Recruitment, R_{t-1}, can be defined as juveniles from previous period that survived to join adult stock in the current year.

Given our assumption that water quality affects snow crab at different stages, we employ a single period lag on the water quality, *V*, to reflect the impact on adults that survived harvest as well as juveniles that are about to become a part of the next period's adult stock. Additionally, water quality may impact snow crab during juvenile settlement and this can be modelled using a recruitment model.

The Ricker model depicts the relationship between spawning stock biomass and recruitment, depending on fitted parameters (Subbey et al., 2014). These parameters allow recruitment to be density dependent. Snow crab recruited into the next year's stock are assumed to have have been hatched 4 years prior such that in each period,

$$R_{it} = S_{it-4}e^{a-b*S_{it-4}} \tag{3.2}$$

where S_{it-4} is adult spawning stock in period 4. Equation 3.2 assumes that water quality does not impact recruitment. To adjust this assumption, we can include a four-year lagged impact of water quality between spawning and juvenile recruitment in the current period.

$$R_{it} = S_{it-4}e^{a-b*S_{it-4}+c*f(V_{it-4})} \tag{3.3}$$

The Ricker spawner-recruit model can be linearized through log transformations. Dividing through by S_{it-4} and taking natural logs yields

$$ln\frac{R_{it}}{S_{it-4}} = a - b * S_{it-4} + c * g(V_{it-4}) \tag{3.4}$$

Finally, harvest is defined as a functional relationship between effort level and crab stock in time period *t*. As noted in section 3.2, the snow crab fishery of the EGSL is managed by a quota. Given the quota (Q_{it}) on total allowable stock set annually for each zone in the fishery, the snow crab industry is restricted to an overall catch that does not exceed the quota. In most fisheries, annual harvest is often equal to the quota set, such that,

$$H_{it} = H(E_{it}, X_{it}) = Q_{it} \tag{3.5}$$

And industry profits are a function of the ex-vessel price of snow crab (p) and per unit cost of effort (w)

$$\pi_{it} = p_{it}H(X_{it}, E_{it}) - w_{it}E_{it} \tag{3.6}$$

The economic impact of variations in oxygen saturation can be estimated by determining how they influence stock and harvest, and the resulting impact on welfare, through changes in producer surplus. However, this impact depends on the institutional arrangements, which determine the rents to be derived from the fishery. In open access fisheries, the effort level chosen by the industry is predicated by the assumption that entry continues until profits (and resource rents) are driven to zero and the fish stock is over-exploited. A similar situation exists under regulated fisheries, except that the quota set by the regulator ensures that a healthy fish stock is maintained. The Department of Fisheries and Oceans (DFO) assumed responsibility for managing the snow crab fishery in 1983. Management measures include limiting fishing effort by limiting catches through a global quota system and through limits on the number, size and type of traps used by fishers, as well as the establishment of a minimum legal carapace length. Total allowable catches (TACs) and individual quota (IQ) programs were established in each fishing area in 1991. TACs are global quotas and are used to determine individual quotas (DFO, 2019). However, a quota on TAC does not create well-defined property rights. Regulators impose restrictions (such as shorter fishing season) to ensure that the total allowable catch is not exceeded, which eventually result in the dissipation of profits (Homans & Wilen, 1997). Alternatively, a licensing program like the one that exists for the snow crab fishery in the EGSL may delay profit dissipation by limiting entry and exit. Licensing restricts the capital entering the fishery. Assuming that EGSL snow crab fishers are price takers, we have two possible scenarios:

1) *Profit dissipation*: Following Homans & Wilen (1997), to ensure that the current stock is not over-exploited the regulator chooses a quota as some proportion (b) of stock in each period $Q_{it} = c + bX_{it}$ on TAC yearly, based on biological considerations (E. B. Barbier, 2019). We assume that there is no restriction on capital and accompanying labour entering (leaving) the fishery quickly, in response to profits (losses), subject to the maximum amount of capital allowed by the

regulator. In the short run, the industry can increase (decrease) fishing effort by additional fishers entering the fishery (Holland, 2011), increasing capital per boat (Stollery, 1986; Karpoff, 1985) or increasing labour in response to perceived marginal cost. However, the regulator closes the fishery to prevent over-exploitation, resulting in increasing hours worked by existing fishers. If there are no other restrictions that limit effort in the fishery, the 'race to fish' will result in rent dissipation in each period and profits are driven to zero. The rent dissipating level of effort is

$$E_{it} = \frac{p_{it} H(X_{it}, E_{it})}{w_{it}} \tag{3.7}$$

When biomass changes, both the regulator instrument and the industry effort change. If the regulator increases the quota in response to increased stock, under our zero profit assumption, changes in biomass would lead to changes in aggregate revenues but total industry profits are zero. No welfare changes occur under this scenario, making this scenario uninteresting.

2) *No profit dissipation*: the regulator chooses a quota as some proportion of stock in each period $Q_{it} = c + bX_{it}$. The fishery continues to attract effort as long as profits are positive and effort will decline if there are losses. However, licensing and other restrictions on fishing effort results in slow entry and delay profit dissipation in the current time period t, but in the long run, resource rents dissipate and profits are driven to zero. Effort, E_{it}, is assumed to evolve according to the fisherman's effort decision and the quota set, which is in effect an effort restriction. The quota chosen in each time period puts a maximum cap on the effort level that can be chosen by the fisherman. Effort is then bounded by the regulator such that $\bar{E}_{it}$ is a limit determined by the regulator (Hoff & Frost, 2008). Assuming that fishermen reach the maximum total allowable catch (TAC) each time period, this implies that fisherman's choice of effort is such that $E_{it} = \bar{E}_{it}$. That is, the effort level chosen in each fishery equals the maximum effort determined by the quota. Representing the quota for each zone in each year as Q_{it}, effort is described as,

$$\bar{E}_{it} = \frac{Q_{it}}{\bar{h}_{it}}; \bar{h}_{it} = \frac{H_{it}}{\bar{E}_{it}} \tag{3.8}$$

Where $\bar{h}_{it}$ is the maximum catch per unit effort (CPUE). Equation 3.8 further sets harvest equal to the quota, given that fishermen put in the maximum effort needed to meet the quota. Profit from the crab fishery is determined residually from the from the harvest level and efforts, which are bounded by the exogenously imposed quota. Profits are determined by

$$\pi_{it} = p_{it}H(X_{it}, \bar{E}_{it}) - w_{it}\bar{E}_{it} \tag{3.9}$$

Changes in biomass will lead to changes the harvest quota and therefore profits.

In the following sections, we will apply the model outlined above to the snow crab fishery in the EGSL by empirically estimating the impact of changes in water quality on adult and juvenile snow crab and estimating the welfare impacts.

3.4 Empirical Strategy

We develop an empirical approach combined with the bioeconomic model to test the hypothesis that adult snow crab stock and recruitment in the EGSL are positively impacted by water quality improvements. We employ a fixed effects panel analysis using data from the snow crab fishery in the EGSL for the nine fishing zones over the period 2001 to 2016. Our empirical model assumes water quality impacts are the same across all zones but differs at varying stages in snow crab lifecycle. We additionally use fixed effects to control for other individual zone characteristics. Our empirical strategy makes use of equation 3.1 and the linearized recruitment equation (equation 3.4), which assume that water quality (V_{it}) impacts both stock and recruitment. The water quality variable of interest for our analysis is dissolved oxygen. However, when estimating the impact of dissolve oxygen on species, the impact of other covarying environmental factors, such as temperature and salinity, must be isolated. An alternative to using dissolved oxygen as a control variable in estimation is to use oxygen saturation, which is adjusted for varying temperature, salinity and pressure. The equations used in our empirical estimation can be re-written as functions of oxygen saturation (O). In equations 3.1 and 3.4, we assume that oxygen saturation linearly impacts adult stock and recruitment, such that,

$$X_{it} = O_{it-1} + (1 - M)X_{it-1} + R_{it-1} - H_{it-1} \tag{3.10}$$

and

$$ln\frac{R_{it}}{S_{it-4}} = a - b * S_{it-4} + c * O_{it-4} \tag{3.11}$$

The variables in 3.10 transformed using logarithms to ensure that residuals are normally distributed and dummy variables are included to control for zone specific characteristics. The stock equation to be estimated is

$$ln(X_{it}) = \alpha_1 ln(O_{it-1}) + \alpha_2 ln(X_{it-1}) + \alpha_3 ln(R_{it-1}) - \alpha_4 ln(H_{it-1}) + D_z + \epsilon_{it} \tag{3.12}$$

where D_z is a zone-specific dummy variable and $z \in \{12A, 12B, 12C, 13, 14, 15, 16, 16A, 17\}$. ϵ is an error term that is assumed to a normally distributed i.i.d. term. Our parameter of interest in equation 3.12 is α_1. This gives the impact of variations in oxygen saturation on the adult snow crab stock. If the α_1 is non-positive, we reject the null hypothesis. If α_1 is positive, we fail to reject the null hypothesis that oxygen saturation positively impacts snow crab stock.

For our empirical estimations, we use a modified version of the Ricker stock-recruit equation. Based on the biological literature on snow crabs, we find evidence that the number of recruits in each period is not only determined by the abundance of female spawners but also the abundance of males. Male snow crabs reach a higher carapace length than female snow crabs and as such, make up a larger proportion of the fishable stock. Males help the females remove their shell during moult, protect them from other males and predators and even feeds them. Upon larval release, males assist females with dispersing the larvae. This implies that the abundance of male snow crabs is a significant determinant of larval production and survival. Our recruitment equation can be adjusted to represent this biological occurrence, where recruitment is a function of overall stock. The stock recruitment relationship described in equation 3.11 assumes that the ratio of current recruits to stock is impacted by oxygen saturation during spawning. However, the biological literature suggests that oxygen saturation may have a larger impact of snow crab during juvenile settlement than during the larval stages. We adjust equation 3.11 to reflect the possible impact of oxygen saturation on juveniles during the recruitment period. Dummy variables are also included to account for zone-specific effect, where $z \in \{12A, 12B, 12C, 13, 14, 15, 16, 16A, 17\}$

$$ln\frac{R_{it}}{X_{it-4}} = \beta_0 - \beta_1 X_{it-4} + \beta_2 O_{it} + \beta_3 O_{it-4} + D_z + \eta_{it} \tag{3.13}$$

η is an error term that is assumed to a normally distributed i.i.d. term. Our parameter of interest in equation 3.13 is β_2. If β_2 and β_3 are positive, we fail to reject the null hypothesis that oxygen saturation positively impacts the stock-recruitment relationship.

3.4.1 Data Description

To investigate the impact of oxygen saturation on snow crab stock in the EGSL, we utilize fishery and environmental data from 2001 - 2016. The key variables used to estimate equations 3.12 and 3.13 are listed in table 3.1summarized in Table 3.2.

Table 3.1: Variable Description

Variable	Name	Source
X_{it}	Total shrimp stock	Estimated
h_{it}	Catch per unit effort	Collected by biologist
S_{it}	Average oxygen saturation	Galbraith et al. (2019)
Q_{it}	Total allowable catch (tonnes)	*Notices to Fish Harvesters* (n.d.)
H_{it}	Total harvest (tonnes)	Collected by biologist
p_{it}	Ex-vessel price per kg	Strategic Services Division, DFO, Quebec
c_{it}	Cost per unit effort	Estimated
π	Profit	Calculated

While some data was available for all nine(9) zones being considered from 2001 - 2016, for some zones - namely, Zones 12C and 16A - there were numerous missing data points and these zones were dropped from our analysis. As such, a complete dataset was constructed for only 7 of the 9 fishing zones - zones 12A, 12B, 13, 14, 15, 16, 17. Data on the key variables was obtained directly from biologists who are currently conducting surveys of snow crab stocks in the EGSL zones, and preparing research documents that give a historical assessment of stock evolution. From equation 3.12, the outcome variable of interest is the natural log of adult snow crab stock. Data provided on adult stock was incomplete and instead catch per unit effort data was used to calculate adult stock. This required some assumption about the harvest function in equation 3.5. For our analysis, we assumed a Schaefer harvest function (J. Zhang & Smith, 2011) such that

$$H_{it} = H(E_{it}, X_{it}) = qE_{it}X_{it} \tag{3.14}$$

where q is the catchability coefficient and catch per unit effort $h_{it} = qX_{it}$. Estimates of the catchabiltiy coefficient was obtained was the literature on snow crab, though not specific to snow crab in the EGSL. Catchabiltiy for the eastern Bering Sea snow crab was estimated to be 0.48-0.78 for males and 0.74-1 for females (Szuwalski & Turnock, 2016). Given that only male snow crabs are fished in the EGSL due to fishery restriction, we used the range for male eastern Bering Sea snow crab as an estimate for catchability in the the EGSL. Additional data required for modeling includes harvest, quota and other environmental variable. Quota for total allowable catch was obtained from the DFO Notice to Fish Harvesters. The DFO sets the quotas for the next year based on the projected harvest levels obtained by applying the decision rules of the current precautionary approach. Observed harvest was very close or equal to the quota on total catch in period. Stock and harvest variables are measured in kilograms(kg).

Table 3.2 gives the summary statistics for key variables used in estimation. For each zone, there are 16 observations for each variable, except for Zone 13, for which there was a moratorium on fishing from 2003 to 2007. We find large variations in stock between Zones 12A, 12B, 13 and 14 and Zones 15, 16 and 17. Zone 15 recorded a significantly higher average stock from 2001 - 2016 than all other zones. A similar variation across zones is seen for recruitment. However, the ratio of recruits to adult stock is similar across all zones except Zone 15. Zone 15 has the highest average stock average recruitment is relatively lower than all zones. Harvest varied widely across zones. Harvest accounted for a large proportion of stock for zones with lower stocks. Interestingly, recorded harvest in Zone 12B was higher than the estimated stock. This may indicate some errors in data collection. For each zone, the quota on total allowable catch was very close to the actual recorded harvest.

The explanatory variable of interest is oxygen saturation. Data on oxygen saturation was obtained from the Department of Fisheries and Oceans' (DFO) Atlantic Zone Monitoring Program (AZMP) and other DFO surveys along with the 2018 report on the physical oceanographic conditions in the Gulf of St. Lawrence (Galbraith et al., 2019). The 2018 report also includes data on surface and deeper layers temperature, salinity and dissolved oxygen. In each year, oxygen saturation is recorded at different longitudes and latitudes on a monthly basis. Oxygen saturation levels account for fluctuations in temperature, salinity and pressure changes. For the period, 37 million observations were received, which were each assigned a fishing zone and then aggregated to an annual level. Average annual oxygen saturation was obtained from Galbraith et al. (2019). Other environmental data was collected even though were not used in empirical testing. Data on dissolved oxygen concentration, surface and bottom layer temperature, and salinity was retrieved from DFO surveys and may be useful for robustness checks. From Table 3.2 we see that average oxygen saturation varies across zones, with Zone 17 having the lowest average oxygen saturation over the period of study. This is consistent with reports that indicate that areas in the Lower Estuary and Gulf of St. Lawrence have experienced the largest declines in oxygen over the past 70 years. Zone 17 experienced moderate to severe hypoxia in some years. Zone 16 also experienced years that were moderately hypoxic but, on average was, not hypoxic. All other zones had oxygen saturation above the critical hypoxic level.

Other variables listed in Table 3.1 are necessary for policy simulations.

3.4.2 Estimation and Results

The model described above is estimated using panel fixed effects regression. The results are depicted in Table 3.3.

Table 3.2: Summary statistics

Variable	**Mean**	**Std. Dev.**	**Min.**	**Max.**
		Zone 12A		
X_{it}	183,626	97,986	56,107	384,386
R_{it}	5.56	3.75	1.11	13.92
O_{it}	48.16	10.37	34.13	67.13
Q_{it}	153,509	52,662	72,560	235,820
H_{it}	143,236	53,176	66,211	235,139
		Zone 12B		
X_{it}	192,438	105,060	55,181	379,280
R_{it}	6.56	5.72	0.43	21.31
O_{it}	53.97	5.97	40.33	62.75
Q_{it}	271,601	71,314	194,098	424,476
H_{it}	254,390	80,971	153,283	415,406
		Zone 13		
X_{it}	328,013	280,330	36,670	941,756
R_{it}	8.95	16.56	0.64	59.12
O_{it}	66.40	11.75	44.61	86.50
Q_{it}	210,726	202,575	4,535	769,136
H_{it}	195,011	182,262	4,535	719,251
		Zone 14		
X_{it}	753,788	396,124	317,132	1,531,886
R_{it}	23.77	32.34	0.17	93.62
O_{it}	55.61	5.88	45.57	67.07
Q_{it}	453,021	113,258	318,357	691,224
H_{it}	450,835	112,891	318,357	691,134
		Zone 15		
X_{it}	8,851,775,571	5,415,487,249	1,495,443	17,091,166,208
R_{it}	47.62	65.59	0.73	237.61
O_{it}	57.88	7.58	37.90	73.11
Q_{it}	492,076	131,655	295,682	716,620
H_{it}	491,594	132,493	292,961	716,983
		Zone 16		
X_{it}	9,796,138	3,177,995	4,319,787	15,418,465
R_{it}	939	501	140	1,743.76
O_{it}	46.71	8.65	29.81	61.01
Q_{it}	3,523,102	881,663	1,965,469	5,035,691
H_{it}	3,501,161	863,137	1,957,306	4,971,267
		Zone 17		
X_{it}	6,633,745	2,344,138	3,763,224	11,012,495
R_{it}	549	406	89.09	1,392
O_{it}	35.50	5.92	23.69	47.61
Q_{it}	1,754,024	485,237	1,217,194	2,739,140
H_{it}	1,743,072	486,358	1,209,938	2,662,045

As discussed in the background description of the species, snow crabs experience natural recruitment troughs and waves over 8 - 12-year periods, which have no identified cause. To control for this natural occurrence, we use 8-year moving averages of stock and recruitment in equation 3.12. Given the one year lag and 8-year moving averages, the number of data points were reduced from 100 to 49. In our first estimation of equation 3.12, we included oxygen saturation as one of the regressors. The results are shown in the first column of Table 3.3. Saturation was found to be insignificant in determining snow crab stock in each period. This is supported by the biological literature, which suggests that adult snow crabs are tolerant to declining oxygen saturation. Previous stock and recruitment are significant determinants of current stock. The coefficient on harvest was found to have the correct sign but insignificant. Our fixed effects regression pools data from each zone as forces the coefficient on regressors to be equal. A Chow test on parameters associated with stock and oxygen saturation was performed and we failed to reject the null hypothesis that the parameters across zones were equal. The error term for the pooled regression was tested for heteroscedasticity, for which we found no evidence. These results are presented in Appendix C.

However, results from the Durbin-Watson test for autocorrelation revealed negative autocorrelation. This was corrected by assuming disturbances are AR(1) distributed. Results are reported in the second column of Table 3.3. The assumption of AR(1) disturbances resulted in a change in magnitude of regressors, however, the signs and significance of the regressors remained the same. Additionally, Zone 12B was dropped due to insufficient variation, reducing the number of zones from 7 to 6. Coefficients suggest that a 1% increase in previous stock and recruitment results in a 0.66% or 0.13% increase in current stock, respectively.

Equation 3.13 was estimated using a similar approach - using zone fixed effects. The results are depicted in column 3 of Table 3.3. The coefficient on period four spawning stock, assumed to determine recruitment, is negative, implying some density dependence. The coefficient was, however, found to be an insignificant determinant of snow crab recruitment. From our recruitment equation, current period saturation was the only significant determinant of the proportion of fourth period stock to current recruits. The coefficient implies that a 1% increase in oxygen saturation

Table 3.3: Estimation Results

VARIABLES	(1) Log(X_{it})	(2) Log(X_{it})	(3) ln$\frac{R_{it}}{X_{it-4}}$
Log(O_{it-1})	0.0862	0.0385	
	(0.0992)	(0.0854)	
Log(X_{it-1})	0.794***	0.660***	
	(0.0823)	(0.146)	
Log(R_{it-1})	0.115***	0.131**	
	(0.0387)	(0.0604)	
Log(H_{it-1})	-0.0474	-0.134	
	(0.0674)	(0.0999)	
X_{it-4}			-6.33e-11
			(1.34e-10)
O_{it}			0.0456**
			(0.021)
O_{it-4}			-0.0046
			(0.021)
Constant	3.331**	6.308***	-13.85***
	(1.475)	(0.805)	(1.539)
Zone FE	Yes	Yes	Yes
AR(1)	No	Yes	No
Observations	49	42	70
R-squared	0.798		0.085
Number of crab zones	7	6	6

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

would lead to a 0.045% increase in the ratio of recruits to stock. These results are are consistent with the biological literature on the impact of oxygen saturation on snow crabs. A Chow test for equality of parameters was carried out. No evidence was found to suggest that oxygen saturation had a different impact on each zone. Additional tests for heteroscedasticity and autocorrelation were carried out, for which we failed to reject the null hypotheses (see Appendix C).

While adult snow crabs and even snow crab larvae appear to be resistant to low oxygen saturation, snow crab during juvenile recruitment is most impacted by declining oxygen saturation. As such, we expect that oxygen saturation would be important in determining current period juvenile recruitment but insignificant for determining the adult stock abundance.

3.5 Welfare Effects

From the data provided on oxygen saturation in the seven of the nine commercial snow crab fishing zones in the EGSL (see table 3.2), we see that only one of the four zones experiences moderate to severe average hypoxia (oxygen concentration 40% and below). Zone 17 has oxygen concentration averaging 35% between 2001 and 2016, with some years having oxygen concentration as low as 24%. For the other zones, average oxygen concentration over the same period was not below 46%. Our policy simulation therefore focuses on the minimum increase in oxygen saturation required for Zone 17 to escape hypoxia. For Zone 17, an approximate 5%-point increase in average oxygen saturation would cause oxygen saturation to increase above the critical hypoxic level. However, increasing the oxygen concentration in Zone 17 by 5% cannot occur without also increasing oxygen saturation in all other zones, given that Zone 17 is located in the lowest part of the EGSL and downstream from the other zones. Additionally, the proximate cause for the decline in oxygen levels and hypoxic conditions in the EGSL is attributed to the combination of anthropogenic nutrient loads and underlying environmental factors, which makes it impossible to target a single fishing zone for an improvement in oxygen saturation. Without explicitly modelling the underlying causes for hypoxia in the EGSL, our counterfactual policy simulations will assume a 5%-point increase above average oxygen saturation across all zones from 2002 - 2016.

Using the parameters from Table 3.3, we simulate changes snow crab stock in the EGSL in response to exogenous shocks to oxygen saturation under cases outlined above. However, our empirical results suggest that oxygen saturation is not a significant of adult snow crab stock. Instead changes in oxygen saturation will only have a significant impact on juvenile recruitment and as such, we will only assume changes in equation 3.11 as oxygen saturation changes.

Along with the estimated parameters from equation 3.13 and assuming existing oxygen saturation levels, we created a base case for our simulations. The base case model was calibrated using actual data - initial stock and oxygen saturation were estimated to be their actual 2001 levels. Adult stock and juvenile recruitment from 2002 to 2016 were then estimated using the estimated parameters from equations 3.12 and 3.13. Total stock in each year is the sum of adults that escaped harvest and previous period juveniles recruited into the current period's adult stock. In each period, harvest equals the quota set by the regulator. With the quota regulation and the assumption that harvest equals the quota in each period, effort in the fishery depends on whether there are sufficient regulations to prevent entry.

In *Case 1* where regulations are insufficient to prevent entry, increases in stock biomass due to increases in oxygen saturation will lead to increase in quota and harvest. With a higher stock biomass, there is the potential for more profits, which attracts effort until resource rents are dissipated and profits are driven to zero. The rent dissipating level of effort is

$$E_{it} = \frac{p_{it}(c + bX_{it})}{w_{it}} \tag{3.15}$$

Based on our discussion in Section 3.2, the EGSL fishing industry is assumed to be price-taking, implying that changes in harvest will have no impact on prices. The result is that consumer surplus will be unaffected by changes in harvest. Given that profits are dissipated in each period and consumer surplus remains unchanged, welfare is not impacted by changes in oxygen saturation, making this case uninteresting. In *Case 2* outlined in section 3.3.1, regulations such as licensing prevents quick entry and allow nonzero profits, at least in the short run. Increases in stock biomass due to increases in oxygen saturation will lead to increase in quota and harvest. With a higher stock biomass, there is the potential for more rents, which attracts more effort. Restrictions prevent additional effort from entering the fishery, leading to a cap on effort and nonzero profits in the short run

$$\bar{E}_{it} = \frac{c + bX_{it}}{\bar{h}_{it}} \tag{3.16}$$

Profits will be determined residually by

$$\pi_{it} = p_{it}(c + bX_{it}) - w_{it}\frac{c + bX_{it}}{\bar{h}_{it}} \tag{3.17}$$

Where, per unit price of harvest, p_{it}, and per unit cost of effort, w_{it} vary annually and by zone. Profits are additionally a function of the regulator's instrument *b* as well as stock. Improvements in oxygen saturation will result in increases in stock, harvest and residual profits. Our assumption that the assumption industry is price-taking implies that while prices change annually, these changes in prices are out of the control of EGSL fishers. As such, changes from harvest in the EGSL fishery does not lead to changes in consumer surplus. Our policy simulations will be based on *Case 2*, in which changes in oxygen saturation leads to changes in fishery profits. Our final step was to simulate the impact of a 5% increase in oxygen saturation on juvenile recruitment and consequently, overall total stock to determine the consequent impacts on net present value profits from 2002 to 2016, using data on prices and cost. The results of our welfare analysis are presented in Table 3.4.

In the case where fishery regulations prevent profits from dissipating, across all six zones we find modest increases in profits under the scenario of a 5-percentage point increase in oxygen saturation in each year. The zones that gain the most from this improvement are the zones in the lowest part of the EGSL. Zones 16 and 17 are in the lowest part of the EGSL and experience the lowest oxygen saturation across all zones. This change in profits can be viewed as the value of increasing oxygen saturation in the EGSL. This is only a part of the value since increasing oxygen saturation is beneficial to other commercial species fished in the EGSL.

Table 3.4: Change in profits[a] from 2002 - 2016 when oxygen saturation is increased by 5-percentage points above actual saturation in each zone over the same period, assuming industry is price taking.

Zone		Total profits (CAN$)[b]
Zone 12A	Actual	2,343,150
	5%-point increase[c]	2,348,188
	difference	0.2%
Zone 13	Actual	2,832,691
	5%-point increase[d]	2,840,925
	difference	0.3%
Zone 14	Actual	6,344,600
	5%-point increase	6,360,832
	difference	0.3%
Zone 15	Actual	6,675,526
	5%-point increase	6,691,565
	difference	0.2%
Zone 16	Actual	49,306,064
	5%-point increase	49,548,791
	difference	0.5%
Zone 17	Actual	28,101,390
	5%-point increase	28,453,796
	difference	1.3%
All Zones	Actual	95,603,422
	5%-point increase	96,244,099
	difference	0.7%

[a] Profits in Canadian dollars (CAN$) is estimated as a present value from 2002 - 2016 using a 5% discount rate.

[b] In 2001, CAN$1 = US$ 0.65

[c] Discounted profits from 2002 - 2016 with a 5-percentage point increase in oxygen saturation.

[d] Discounted profits from 2002 - 2016 with a 5-percentage point increase in oxygen saturation, assuming moratorium was not implemented.

3.6 Discussion and Conclusion

The results of our empirical analysis partially supports our hypothesis that snow crab abundance is positively impacted by improvements in water quality, particularly, improvements in oxygen concentration. Consistent with biological literature on the relationship between oxygen saturation and snow crabs, we find that oxygen saturation does not significantly impact adult snow crab abundance but is a significant determinant of juvenile recruitment. Adult snow crabs are found to be more tolerant to low oxygen concentrations and even hypoxia. Snow crab larvae may be slightly less tolerant, however, the significant impact is found for juvenile snow crabs. In the EGSL, a 1% increase in oxygen results in an approximate 0.045% increase in the ratio of juvenile recruits to stock.

Our simulations further indicate that there are positive welfare effects of increasing oxygen saturation. If oxygen saturation is 5-percentage points higher each year, there is a net increase in fish stock above the baseline scenario. The change in fish stock resulting from changes in environmental conditions has welfare implications, and who loses and who gains depend on market conditions and institutions governing fishery management. One of the important implications of our results is that the welfare changes from an improvement in oxygen saturation vary significantly across zones. For example, we find the largest change in profits occurred in Zone 17, implying that this zone stands to gain the most from improvements in oxygen saturation. Zone 17 is the second most productive snow crab fishing zone in the EGSL, despite having relatively low oxygen saturation experiencing intermittent hypoxia over the period of analysis. This is contrary to what is observed in this portion of the EGSL for other fish species such as the northern shrimp and may be a further indicator of the tolerance of snow crabs to low oxygen saturation.

Our empirical estimation and numerical simulations indicated that there are positive welfare benefits to be obtained from improving water quality through increasing oxygen saturation in the EGSL. The analysis, however, is not without its limitations. The most obvious limitation is the number of years of analysis. Due to the unavailability of data on snow crab abundance and harvest in the EGSL, we were only able to analyze 16 data points for each zone. This is insufficient data for each zone given that snow crabs experience 8 to 12-year natural waves and troughs that are unexplained. This implies that stock will decline despite favourable environmental conditions. We controlled for this by using 8-year moving averages, which significantly reduced the number of observations per zone. The use of fixed effects along with the assumption of AR(1) distributed errors further reduced the number of zones that were analyzed. The final results for our estimation of equation3.12 were based on 42 observations across 6 zones and and 70 observations across 6 zones for equation 3.13. This raises questions about the use of our results for generalizations. However, the model, estimation and welfare analysis presented here creates the foundation for analysis once more data is made available. In addition to the lack of data, there is little confidence in the data that was available as it appeared that the data was not based on actual annual observations from the fishery but rather aggregations. This further affects the reliability of the results from estimations and welfare analysis.

Another limitation is related to policy simulations. In our policy simulation, we assumed a 5-percentage point increase in oxygen saturation because it would take the critical Zone 17 above the hypoxic threshold, and this was the simulated increase across all zones. However, water loses oxygen as it flows between zones, so higher a larger increase would be needed in zones in the upper portions of the river in order to achieve a 5-percentage point increase in the lower EGSL (see Figure 3.1). It may be more intuitive to model the pathway though which a policy intervention would lead to an increase in oxygen saturation across the different zones. This would require data on the source of pollution in the EGSL and the biological and hydrological mechanisms through which pollution leads to hypoxia. This is an area for future research.

References

Abdel-Tawwab, M., Monier, M., Hoseinifar, S., & Faggio, C. (2019). Fish response to hypoxia stress: growth, physiological, and immunological biomarkers. *Fish Physiology Biochemistry*, *45*, 997 - 1013.

Asche, F., Bjøorndal, T., & Gordon, D. V. (2009). Resource rent in individual quota fisheries. *Land Economics*, *85*(2), 279–291. Retrieved from `http://www.jstor.org/stable/27759674`

Bao, J., Li, X., Xing, Y., Feng, C., & Jiang, H. (2020). Effects of hypoxia on immune responses and carbohydrate metabolism in the chinese mitten crab, eriocheir sinensis. *Aquaculture Research*, *51*(7), 2735-2744. Retrieved from `https://onlinelibrary.wiley.com/doi/abs/10.1111/are.14612` doi: https://doi.org/10.1111/are.14612

Barbier, E. (2019). Valuing Coastal Habitat–Fishery Linkages under Regulated Open Access. *Water*, *11*(847).

Barbier, E. B. (2019). Valuing coastal habitat–fishery linkages under regulated open access. *Water*, *11*(4), 847.

Barbier, E. B. (2022). The policy implications of the Dasgupta review: Land use change and biodiversity. *Environmental and Resource Economics*, 1–25.

Barica, J. (1990). Seasonal variability of n: P ratios in eutrophic lakes. *Hydrobiologia*, *191*(1), 97–103.

Basher, Z., & Costello, M. J. (2020). World maps of ocean environment variables. In M. I. Goldstein & D. A. DellaSala (Eds.), *Encyclopedia of the world's biomes* (p. 479-493). Oxford: Elsevier. Retrieved from `https://www.sciencedirect.com/science/article/pii/B9780124095489120767` doi: https://doi.org/10.1016/B978-0-12-409548-9.12076-7

Beiras, R., & Beiras, R. (2018). Nonpersistent inorganic pollution. *Marine pollution. España: Elsevier. Doi*, *10*.

Beverton, R., & Holt, S. (2012). *On the dynamics of exploited fish populations* (Vol. 11). Springer Science & Business Media.

Beverton, R. J. H. (1954). *Notes on the use of theoretical models in the study of the dynamics of exploited fish populations*. US Fishery Laboratory.

Birch, C. P. D. (1999, 06). A New Generalized Logistic Sigmoid Growth Equation Compared with the Richards Growth Equation. *Annals of Botany*, *83*(6), 713-723. Retrieved from `https://doi.org/10.1006/anbo.1999.0877` doi: 10.1006/anbo.1999.0877

Bisping, S., Alfermann, T., & Strickland, P. (2019). Population characteristics of yellow perch in dead lake, florida. *Journal of Fish and Wildlife Management*, *10*(2), 296-303. Retrieved from `https://doi.org/10.3996/082018-JFWM-068` doi: 10.3996/082018-JFWM-068

Björn, L. O., Papageorgiou, G. C., & Blankenship, R. E. (2009). A viewpoint: why chlorophyll a? *Photosynthesis research*, *99*, 85–98.

Board of Governors of the Federal Reserve System (US), u. h. a. . a., publisher =Federal Reserve Bank of St. Louis. (n.d.). *Canadian dollars to u.s. dollar spot exchange rate [excaus].*

Bourdages, H., Brassard, C., Galbraith, P., Gauthier, J., Nozères, C., Senay, C., … Senay, C. (2018). Preliminary results from the groundfish and shrimp multidisciplinary survey in august 2017 in the estuary and northern gulf of st. lawrence.

Bourdages, H., Marquis, M.-C., Ouellette-Plante, J., Chabot, D., Galbraith, P., & Isabel, L. (2020). Assessment of northern shrimp stocks in the estuary and gulf of st. lawrence in 2019: commercial fishery and research survey data. *DFO Canadian Science Advisory Secretariat Research Document*, *2020/012*.

Breitburg, D. (2002). Effects of Hypoxia, and the Balance between Hypoxia and Enrichment, on Coastal Fishes and Fisheries. *Estuaries*, *25*(4), 767-781.

Brennan, C., Blanchard, H., & Fennel, K. (2016). Putting temperature and oxygen thresholds of marine animals in context of environmental change: A regional perspective for the scotian shelf and gulf of st. lawrence. *PLOS ONE*, *11*(12), 1-27. Retrieved from `https://doi.org/10.1371/journal.pone.0167411` doi: 10.1371/journal.pone.0167411

Chen, S., Xie, S., Chen, M., Mi, Z., He, Q., Yang, F., ... Tian, L. (2018, 12). Hypoxia-induced changes in survival, immune response and antioxidant status of the pacific white shrimp (litopenaeus vannamei) fed with graded levels of dietary myo-inositol. *Aquaculture Nutrition*, *25*. doi: 10.1111/anu.12877

Claret, M., Galbraith, E., Palter, J., Bianchi, D., Fennel, K., Gilbert, D., & Dunne, J. (2018). Rapid coastal deoxygenation due to ocean circulation shift in the northwest atlantic. *Nature climate change*, *8*(10), 868–872.

Cochran, R. E., & Burnett, L. E. (1996). Respiratory responses of the salt marsh animals, fundulus heteroclitus, leiostomus xanthurus, and palaemonetes pugio to environmental hypoxia and hypercapnia and to the organophosphate pesticide, azinphosmethyl. *Journal of Experimental Marine Biology and Ecology*, *195*(1), 125–144.

Coiro, L. L., Poucher, S. L., & Miller, D. C. (2000). Hypoxic effects on growth of palaemonetes vulgaris larvae and other species:: Using constant exposure data to estimate cyclic exposure response. *Journal of experimental marine biology and ecology*, *247*(2), 243–255.

Council, N. R. (n.d.). *Clean coastal waters: Understanding and reducing the effects of nutrient pollution* (Tech. Rep.).

Craig, K., & Bosman, S. (2013). Small spatial scale variation in fish assemblage structure in the vicinity of the northwestern gulf of mexico hypoxic zone. *Estuaries and Coasts*, *36*(2), 268–285.

Craig, K., & Crowder, L. (2005). Hypoxia-induced habitat shifts and energetic consequences in Atlantic croaker and brown shrimp on the Gulf of Mexico shelf. *Marine Ecology-progress Series*, *294*, 79–94.

Crutchfield, J. A. (1961). An economic evaluation of alternative methods of fishery regulation. *The Journal of Law and Economics*, *4*, 131–143.

Daldorph, P. W. G., & Thomas, J. D. (1991). The effect of nutrient enrichment on a freshwater community dominated by macrophytes and molluscs and its relevance to snail control. *Journal of Applied Ecology*, *28*(2), 685–702. Retrieved from `http://www.jstor.org/stable/2404576`

Daoud, D., Chabot, D., Audet, C., & Lambert, Y. (2007). Temperature induced variation in oxygen consumption of juvenile and adult stages of the northern shrimp, pandalus borealis. *Journal of Experimental Marine Biology and Ecology*, *347*(1-2), 30–40.

Daoud, D., Chabot, D., Audet, C., & Lambert, Y. (2008). Energy budget of the northern shrimp, pandalus borealis, as a function of size and temperature. *Comparative Biochemistry and Physiology Part A: Molecular Integrative Physiology*, *150*(3, Supplement), S114. Retrieved from `https://www.sciencedirect.com/science/article/pii/S1095643308009823` (Abstracts of the Annual Main Meeting of the Society of Experimental Biology, 6th - 10th July 2008, Marseille, France) doi: https://doi.org/10.1016/j.cbpa.2008.04.622

Daoud, D., Lambert, Y., Audet, C., & Chabot, D. (2010). Size and temperature-dependent variations in intermolt duration and size increment at molt of northern shrimp, *Pandalus borealis*. *Marine Biology*, *157*, 2655-2666.

Deacon, R. T., Finnoff, D., & Tschirhart, J. (2011). Restricted capacity and rent dissipation in a regulated open access fishery. *Resource and Energy Economics*, *33*(2), 366–380.

de La Chenelière, V., Brodeur, P., & Mingelbier, M. (2014). Restauration des habitats du lac saint-pierre: un prérequis au rétablissement de la perchaude. *Le naturaliste canadien*, *138*(2), 50–61.

Dennison, W., Orth, R., Moore, K., Stevenson, J., Carter, V., Kollar, S., … Batiuk, R. (1993). Assessing Water Quality with Submersed Aquatic Vegetation. *BioScience*, *43*(2), 86–94.

Desprez, M., Rybarczyk, H., Wilson, J., Ducrotoy, J., Sueur, F., Olivesi, R., & Elkaim, B. (1992). Biological impact of eutrophication in the bay of somme and the induction and impact of anoxia. *Netherlands Journal of Sea Research*, *30*.

DFO. (n.d.-a). *Canada's stateoftheoceans report, 2012.* Department of Fisheries and Oceans, Canada. Retrieved from `chrome-extension://efaidnbmnnnibpcajpcglclefindmkaj/https://waves-vagues.dfo-mpo.gc.ca/library-bibliotheque/346701.pdf`

DFO. (n.d.-b). Notice to fish harvesters. Retrieved from `https://inter-l01.dfo-mpo.gc.ca/applications/opti-opei/notices-species-avis-especes-eng.php?type=2&display_option=1&region_id=4&sub_type_id=5&species=702&area=1858`

DFO. (n.d.-c). *Snow crab.* Department of Fisheries and Oceans, Canada. Retrieved from `https://www.dfo-mpo.gc.ca/species-especes/profiles-profils/snow-crab-crabe-neiges-atl-eng.html`

DFO. (2006a). *2004 Costs and Earnings Survey, Atlantic Region.* Department of Fisheries and Oceans, Canada. Retrieved from `https://www.dfo-mpo.gc.ca/stats/commercial/ces/content-eng.htm`

DFO. (2006b). Cost and Earnings Survey, 2004: Non-traditional shrimp fishers, Québec region.

DFO. (2012). *Canada's state of the oceans report.* Centre of Expertise on the State of the Oceans, Canada. Retrieved from `https://www.dfo-mpo.gc.ca/oceans/publications/soto-rceo/2012/page03-eng.html`

DFO. (2014). *Northern shrimp – Areas 8, 9, 10 and 12 (Estuary and Gulf of St. Lawrence).* Department of Fisheries and Oceans, Canada. Retrieved from `https://www.dfo-mpo.gc.ca/fisheries-peches/ifmp-gmp/shrimp-crevette/shrimp-crevette-2018-eng.html#4`

DFO. (2016). *Commercial fisheries and aquaculture.* Department of Fisheries and Oceans, Canada. Retrieved from `https://dfo-mpo.gc.ca/stats/commercial/cfs/2012/section2-eng.htm`

DFO. (2019). *Snow crab - estuary and northern gulf of st. lawrence inshore areas (12a, 12b, 12c, 13, 14, 15, 16, 16a and 17).* Department of Fisheries and Oceans, Canada. Retrieved from `https://www.dfo-mpo.gc.ca/stats/commercial/land-debarq/sea-maritimes/s2019av-eng.htm`

DFO. (2020a). *Seafisheries landed value by region, 2019.* Department of Fisheries and Oceans, Canada. Retrieved from `https://www.dfo-mpo.gc.ca/stats/commercial/land-debarq/sea-maritimes/s2019av-eng.htm`

DFO. (2020b). *Assessment of northern shrimp stocks in the estuary and gulf of st. lawrence in 2019.* Department of Fisheries and Oceans, Canada. Retrieved from `https://publications.gc.ca/collections/collection_2020/mpo-dfo/fs70-6/Fs70-6-2020-010-eng.pdf`

Diaz, R. J., & Solow, A. (1999). Ecological and economic consequences of hypoxia, topic 2. Gulf of Mexico hypoxia assessment. *National Oceanic and Atmospheric Administration, Silver Spring, Maryland.*

Dodds, W., Bouska, W. W., Eitzmann, J. L., Pilger, T. J., Pitts, K. L., Riley, A., … Thornbrugh, D. J. (2009). Eutrophication of u.s. freshwaters: Analysis of potential economic damages. *Environmental Science & Technology*, *43*(1), 12-19. Retrieved from `https://doi.org/10.1021/es801217q` doi: 10.1021/es801217q

Domenici, P., Lefrançois, C., & Shingles, A. (2007). Hypoxia and the antipredator behaviours of fishes. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *362*(1487), 2105–2121.

Dupont-Prinet, A., Pillet, M., Chabot, D., Hansen, T., Tremblay, R., & Audet, C. (2013). Northern shrimp (*Pandalus borealis*) oxygen consumption and metabolic enzyme activities are severely constrained by hypoxia in the estuary and gulf of st. lawrence. *Journal of Experimental Marine Biology and Ecology*, *448*, 298-307. Retrieved from `https://www.sciencedirect.com/science/article/pii/S0022098113002906` doi: https://doi.org/10.1016/j.jembe.2013.07.019

Environment and Climate Change Canada. (2018). *Canadian Environmental Sustainability Indicators: Nutrients in the St. Lawrence River. Consulted on Month day, year.* Retrieved from `www.canada.ca/en/environment-climate-change/services/environmentalindicators/nutrients-st-lawrence-river.html.`

EPA. (2006). *Voluntary Estuary Monitoring Manual.* Retrieved from `https://www.epa.gov/sites/production/files/2015-09/documents/2009_03_13_estuaries_monitor_chap18.pdf`

Foubert, A., Lecomte, F., Brodeur, P., Le Pichon, C., & Mingelbier, M. (2020). How intensive agricultural practices and flow regulation are threatening fish spawning habitats and their connectivity in the st. lawrence river floodplain, canada. *Landscape Ecology*, *35*(5), 1229–1247.

Frenette, J., Arts, M., Morin, J., Gratton, D., & Martin, C. (2006). Hydrodynamic control of the underwater light climate in fluvial lac saint-pierre. *Liminology and Oceanography*, *51*(6), 2632–2645.

Galbraith, P., Chassé, J., Caverhill, C., Nicot, P., Gilbert, D., & Lafleur, C. (2019). Physical oceanographic conditions in the gulf of st. lawrence in 2018. *DFO Canadian Science Advisory Secretariat Research Document*, *2019/046*.

Gannon, D., Berens McCabe, E., Camilleri, S., Gannon, J., Brueggen, M., Barleycorn, A., … Wells, R. (2009). Effects of *Karenia brevis* harmful algal blooms on nearshore fish communities in southwest Florida. *Marine Ecology Progress Series*, *378*, 171 - 186.

Giacomazzo, M., Bertolo, A., Brodeur, P., & Magnan, P. (2023). Relationship between submerged aquatic vegetation, turbidity, and fish distribution in a large shallow fluvial lake. *Environmental Biology of Fishes*, *106*(1), 1–17.

Giacomazzo, M., Bertolo, A., Brodeur, P., Massicotte, P., Goyette, J.-O., & Magnan, P. (2020). Linking fisheries to land use: How anthropogenic inputs from the watershed shape fish habitat quality. *Science of The Total Environment*, *717*, 135377.

Gilbert, D., Chabot, D., Archambault, P., Rondeau, B., & Hébert, S. (2007). Appauvrissement en oxygène dans les eaux profondes du saint-laurent marin: causes possibles et impacts écologiques. *Le naturaliste canadien*, *31*(1), 67–75.

Gilbert, D., Sundby, B., Gobeil, C., Mucci, A., & Tremblay, G.-H. (2005). A seventy-two-year record of diminishing deep-water oxygen in the st. lawrence estuary: The northwest atlantic connection. *Limnology and Oceanography*, *50*(5), 1654-1666. Retrieved from `https://aslopubs.onlinelibrary.wiley.com/doi/abs/10.4319/lo.2005.50.5.1654` doi: https://doi.org/10.4319/lo.2005.50.5.1654

Gravinese, P. M., Munley, M. K., Kahmann, G., Cole, C., Lovko, V., Blum, P., & Pierce, R. (2020). The effects of prolonged exposure to hypoxia and florida red tide (karenia brevis) on the survival and activity of stone crabs. *Harmful Algae*, *98*, 101897. Retrieved from `https://www.sciencedirect.com/science/article/pii/S1568988320301761` doi: https://doi.org/10.1016/j.hal.2020.101897

Hardin, G. (2019). The tragedy of the commons. In *Green planet blues* (pp. 41–49). Routledge.

Hardy, D., Munro, J., & Dutil, J.-D. (1994). Temperature and salinity tolerance of the soft-shell and hard-shell male snow crab, chionoecetes opilio. *Aquaculture*, *122*(2-3), 249–265.

Heisler, J., Glibert, P., Burkholder, J., Anderson, D., Cochlan, W., Dennison, W., … Suddleson, M. (2008). Eutrophication and harmful algal blooms: A scientific consensus. *Harmful Algae*, *8*(1), 3-13. Retrieved from `https://www.sciencedirect.com/science/article/pii/S1568988308001066`

Hernandez-Llamas, A., & Ratkowsky, D. A. (2004). Growth of fishes, crustaceans and molluscs: estimation of the von bertalanffy, logistic, gompertz and richards curves and a new growth model. *Marine Ecology Progress Series*, *282*, 237–244.

Hilborn, R., Orensanz, J., & Parma, A. (2005). Institutions, incentives and the future of fisheries. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *360*(1453), 47–57.

Hoff, A., & Frost, H. (2008). Modelling combined harvest and effort regulations: the case of the dutch beam trawl fishery for plaice and sole in the north sea. *ICES Journal of Marine Science*, *65*(6), 822–831.

Holland, D. S. (2011). Optimal intra-annual exploitation of the maine lobster fishery. *Land Economics*, *87*(4), 699–711.

Homans, F., & Wilen, J. (1997). A Model of Regulated Open Access Resource Use. *Journal of Environmental Economics and Management*, *32*(1).

Howarth, R., Anderson, D., Cloern, J., Elfring, C., Hopkinson, C., Lapointe, B., … Walker, D. (2000). Nutrient pollution of coastal rivers, bays, and seas. *Issues in Ecology*(7).

Huang, L., Nichols, L., Craig, J., & Smith, M. (2012). Measuring welfare losses from hypoxia: The case of North Carolina brown shrimp. *Marine Resource Economics*, *27*.

Huang, L., & Smith, M. (2011). Management of an annual fishery in the presence of ecological stress: The case of shrimp and hypoxia. *Ecological Economics*, *70*.

Huang, L., Smith, M., & Craig, K. (2010). Quantifying the Economic Effects of Hypoxia on a Fishery for Brown Shrimp *Farfantepenaeus aztecus*. *Marine and Coastal Fisheries: Dynamica, Management, and Ecosystem Science*, *2*, 232–248.

Hudon, C., & Carignan, R. (2008). Cumulative impacts of hydrology and human activities on water quality in the st. lawrence river (lake saint-pierre, quebec, canada). *Canadian Journal of Fisheries and Aquatic Sciences*, *65*, 1165–1180. doi: 10.1139/F08-069

Hudon, C., Cattaneo, A., Tourville Poirier, A., Brodeur, P., Dumont, P., Mailhot, Y., … de Lafontaine, Y. (2012). Oligotrophication from wetland epuration alters the riverine trophic network and carrying capacity for fish. *Aquatic Science*, *74*, 495-511.

Hudon, C., Jean, M., & Létourneau, G. (2018). Temporal (1970–2016) changes in human pressures and wetland response in the St. Lawrence River (Québec, Canada). *Science of The Total Environment*, *643*, 1137–1151.

Hvingel, C., & Kingsley, M. C. (2006). A framework to model shrimp (Pandalus borealis) stock dynamics and to quantify the risk associated with alternative management options, using Bayesian methods. *ICES Journal of Marine Science*, *63*(1), 68-82. Retrieved from `https://doi.org/10.1016/j.icesjms.2005.09.002` doi: 10.1016/j.icesjms.2005.09.002

Islam, M. S., & Tanaka, M. (2004). Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: a review and synthesis. *Marine Pollution Bulletin*, *48*(7 - 8), 624-649.

Istvánovics, V. (2010). Eutrophication of lakes and reservoirs. *Lake Ecosystem Ecology; Elsevier: San Diego, CA, USA*, 47–55.

Jacobson, P., Hansen, G., Bethke, B., & Cross, T. (2017). Disentangling the effects of a century of eutrophication and climate warming on freshwater lake fish assemblages. *PLoS ONE*, *12*(8). Retrieved from `https://doi.org/10.1371/journal.pone.0182667`

Jin, D., Thunberg, E., & Hoagland, P. (2008). Economic impact of the 2005 red tide event on commercial shellfish fisheries in new england. *Ocean & Coastal Management*, *51*(5), 420-429. doi: https://doi.org/10.1016/j.ocecoaman.2008.01.004

Kahn, J. (1988). Measuring The Economic Effects of Brown Tides. *Journal of Shellfish Research*, *7*.

Kahn, J., & Kemp, W. (1985). Economic losses associated with the degradation of an ecosystem: The case of submerged aquatic vegetation in chesapeake bay. *Journal of Environmental Economics and Management*, *12*(3), 246-263. Retrieved from `https://www.sciencedirect.com/science/article/pii/0095069685900336` doi: https://doi.org/10.1016/0095-0696(85)90033-6

Kar, T., & Chakraborty, K. (2011). A bioeconomic assessment of the bangladesh shrimp fishery. *World Journal of Modelling and Simulation*, *7*(1), 58–69.

Karpoff, J. M. (1985). Time, capital intensity, and the cost of fishing effort. *Western Journal of Agricultural Economics*, 254–258.

Katsev, S., Chaillou, G., Sundby, B., & Mucci, A. (2007). Effects of progressive oxygen depletion on sediment diagenesis and fluxes: A model for the lower st. lawrence river estuary. *Limnology and Oceanography*, *52*(6), 2555–2568.

Kinnunen, R. E. (2003). Great lakes commercial fisheries. *Michigan Sea Grant Extension, Marquette*, *19*.

Knowler, D., & Barbier, E. (2005). Managing the Black Sea Anchovy Fishery with Nutrient Enrichment and a Biological Invader. *Marine Resource Economics*, *20*(3), 263–285.

Knowler, D., Barbier, E., & Strand, I. (2001). An Open-Access Model of Fisheries and Nutrient Enrichment in the Black Sea. *Marine Resource Economics*, *16*(3).

Kolka, R. K., Riggs, C. E., Nater, E. A., Wickman, T. R., Witt, E. L., & Butcher, J. T. (2019). Temporal fluctuations in young-of-the-year yellow perch mercury bioaccumulation in lakes of northeastern minnesota. *Science of The Total Environment*, *656*, 475-481. Retrieved from `https://www.sciencedirect.com/science/article/pii/S0048969718346369` doi: https://doi.org/10.1016/j.scitotenv.2018.11.280

Kungvankij, P., Chua, T.-E., Pudadera Jr, B., Corre, K., Borlongan, E., Tiro Jr, L., ... others (1986). Shrimp culture: pond design, operation and management.

Kutty, M. (1987). *Chemical features of water.* United Nations Development Programme Food and Agriculture Organization of the United Nations. Retrieved from `https://www.fao.org/3/ac183e/AC183E00.htm#TOC`

Lavaud, R., Thomas, Y., Pecquerie, L., Benoît, H. P., Guyondet, T., Flye-Sainte-Marie, J., & Chabot, D. (2019). Modeling the impact of hypoxia on the energy budget of atlantic cod in two populations of the gulf of saint-lawrence, canada. *Journal of Sea Research*, *143*, 243–253.

Magnan, P., Paquin, É., Brodeur, P., Paradis, Y., Vachon, N., Dumont, P., & Mailhot, Y. (2022). *État du stock de perchaudes du lac saint-pierre et du secteur pont laviolette–saint-pierre-les-becquets en 2021. comité scientifique sur la gestion de la perchaude du lac saint-pierre.* Ministère des Forêts, de la Faune et des Parcs.

Magnoni, L. J., Eding, E., Leguen, I., Prunet, P., Geurden, I., Ozório, R. O., & Schrama, J. W. (2018). Hypoxia, but not an electrolyte-imbalanced diet, reduces feed intake, growth and oxygen consumption in rainbow trout (oncorhynchus mykiss). *Scientific reports*, *8*(1), 1–14.

Mahmood, R., Pielke, R. A., & McAlpine, C. A. (2016). Climate-relevant land use and land cover change policies. *Bulletin of the American Meteorological Society*, *97*(2), 195 - 202.

Retrieved from `https://journals.ametsoc.org/view/journals/bams/97/2/bams-d-14-00221.1.xml` doi: 10.1175/BAMS-D-14-00221.1

Mailhot, Y., Dumont, P., Paradis, Y., Brodeur, P., Vachon, N., Mingelbier, M., … Magnan, P. (2015). Yellow perch (perca flavescens) in the st. lawrence river (québec, canada): population dynamics and management in a river with contrasting pressures. In *Biology of perch* (pp. 101–147). Boca Raton, FL: CRC Press.

Mao, X., Guo, X., Wang, Y., & Takayama, K. (2019). Influences of global warming on the larval survival and transport of snow crab (chionoecetes opilio) in the sea of japan. *Sustainability*, *11*(8), 2198.

Marcello, L. A., Mueter, F. J., Dawe, E. G., & Moriyasu, M. (2012). Effects of temperature and gadid predation on snow crab recruitment: comparisons between the bering sea and atlantic canada. *Marine Ecology Progress Series*, *469*, 249–261.

Massey, D. M., Newbold, S., , & Gentner, B. (2006). Valuing water quality changes using a bioeconomic model of a coastal recreational fishery. *Journal of Environmental Economics and Management*, *52*.

Maunder, M. N., & Piner, K. R. (2014). Contemporary fisheries stock assessment: many issues still remain. *ICES Journal of Marine Science*, *72*(1), 7-18. Retrieved from `https://doi.org/10.1093/icesjms/fsu015` doi: 10.1093/icesjms/fsu015

McDowell, R., Noble, A., Pletnyakov, P., Haggard, B., & Mosley, L. (2020). Global mapping of freshwater nutrient enrichment and periphyton growth potential. *Scientific reports*, *10*(1), 1–13.

McLeese, D., & Watson, J. (1968). Oxygen consumption of the spider crab (chionoecetes opilio) and the american lobster (homarus americanus) at a low temperature. *Journal of the Fisheries Board of Canada*, *25*(8), 1729–1732.

Meyfroidt, P., de Bremond, A., Ryan, C. M., Archer, E., Aspinall, R., Chhabra, A., … zu Ermgassen, E. K. H. J. (2022). Ten facts about land systems for sustainability. *Proceedings of the*

National Academy of Sciences, *119*(7), e2109217118. Retrieved from `https://www.pnas.org/doi/abs/10.1073/pnas.2109217118` doi: 10.1073/pnas.2109217118

Mississippi River/Gulf of Mexico Watershed Nutrient Task Force. (2020b). *Gulf hypoxia action plan 2008 for reducing, mitigating and controlling hypoxia in the northern gulf of mexico and improving water quality in the mississippi river basin.* United States Environmental Protection Agency. Retrieved from `https://www.epa.gov/sites/default/files/2015-03/documents/2008_8_28_msbasin_ghap2008_update082608.pdf`

Nguyen, T., Ravn-Jonsen, L., & Vestergaard, N. (2016). Marginal damage cost of nutrient enrichment: The case of the baltic sea. *Environmental Resource Economics*, *64*, 109 - 129. Retrieved from `https://doi.org/10.1007/s10640-014-9859-8`

Notices to fish harvesters. (n.d.). Retrieved 2023-04-24, from `https://www.qc.dfo-mpo.gc.ca/en/Snow-crab-crabe-des-neiges`

O'Connor, T., & Whitall, D. (2007). Linking hypoxia to shrimp catch in the northern gulf of mexico. *Marine Pollution Bulletin*, *54*, 460-3. doi: 10.1016/j.marpolbul.2007.01.017

O'Hare, M. T., Baattrup-Pedersen, A., Baumgarte, I., Freeman, A., Gunn, I. D. M., Lázár, A. N., … J., B. M. (2018). Responses of Aquatic Plants to Eutrophication in Rivers: A Revised Conceptual Model. *Frontiers in Plant Science*, *9*, 451.

Parrack, M. L. (1978). Aspects of brown shrimp, penaeus aztecus, growth. *Fishery Bulletin*, *76*(4), 827.

Pick, F. (2016). Blooming algae: a canadian perspective on the rise of toxic cyanobacteria. *Canadian Journal of Fisheries and Aquatic Science*, *73*, 246-263. Retrieved from `https://cdnsciencepub.com/doi/pdf/10.1139/cjfas-2015-0470`

Pillet, M., Dupont-Prinet, A., Chabot, D., Tremblay, R., & Audet, C. (2016). Effects of exposure to hypoxia on metabolic pathways in northern shrimp (pandalus borealis) and greenland halibut (reinhardtius hippoglossoides). *Journal of Experimental Marine Biology and Ecology*, *483*, 88-96.

Power, M., & van den Heuvel, M. (1999, 07). Age-0 yellow perch growth and its relationship to temperature. *Transactions of The American Fisheries Society*, *128*, 687-700. doi: 10.1577/1548-8659(1999)128<0687:AYPGAI>2.0.CO;2

Purcell, K., Craig, J., Nance, J., Smith, M., & Bennear, L. (2017). Fleet behavior is responsive to a large-scale environmental disturbance: Hypoxia effects on the spatial dynamics of the northern Gulf of Mexico shrimp fishery. *PLOS ONE*, *12*(8), 135377.

Pérez-Castañeda, R., & Defeo, O. (2005). Growth and mortality of transient shrimp populations (Farfantepenaeus spp.) in a coastal lagoon of Mexico: role of the environment and density-dependence. *ICES Journal of Marine Science*, *62*(1), 14-24. Retrieved from `https://doi.org/10.1016/j.icesjms.2004.10.005` doi: 10.1016/j.icesjms.2004.10.005

Quinn, T., & Deriso, R. (1999). *Quantitative fish dynamics*. oxford university Press.

Qyli, M., Aliko, V., et al. (2017). The impact of short-term exposure to hypoxia on mediterranean crab carcinus aestuarii. *Albanian Journal of Agricultural Sciences*, 399–405.

Rabotyagov, S. S., Campbell, T. D., White, M., Arnold, J. G., Atwood, J., Norfleet, M. L., … others (2014). Cost-effective targeting of conservation investments to reduce the northern gulf of mexico hypoxic zone. *Proceedings of the National Academy of Sciences*, *111*(52), 18530–18535.

Rahman, A., Dabrowski, J., & McCulloch, J. (2020). Dissolved oxygen prediction in prawn ponds from a group of one step predictors. *Information Processing in Agriculture*, *7*(2), 307–317.

Reimer, M., & Wilen, J. (2013). Regulated open access and regulated restricted access fisheries. In J. F. Shogren (Ed.), *Encyclopedia of energy, natural resource, and environmental economics* (p. 215-223). Waltham: Elsevier. Retrieved from `https://www.sciencedirect.com/`

`science/article/pii/B9780123750679000450` doi: https://doi.org/10.1016/B978-0-12-375067-9.00045-0

Ricker, W. E. (1954). Stock and recruitment. *Journal of the Fisheries Board of Canada*, *11*(5), 559–623.

Roman, M., Brandt, S., Houde, E., & Pierson, J. (2019). Interactive effects of hypoxia and temperature on coastal pelagic zooplankton and fish. *Frontiers in Marine Science*, *6*(139), 1-18.

Rounds, S., Wilde, F., & Ritz, G. (2013). Dissolved oxygen. In (3.0 ed.). U.S. Geological Survey Techniques of Water-Resources Investigations.

Sampat, A., Hicks, A., G.J., R.-M., & Zavala, V. (2021). Valuing economic impact reductions of nutrient pollution from livestock waste. *Resources, Conservation and Recycling*, *164*, 105199. Retrieved from `https://www.sciencedirect.com/science/article/pii/S0921344920305164` doi: https://doi.org/10.1016/j.resconrec.2020.105199

Sandström, A., & Karås, P. (2002). Effects of eutrophication on young-of-the-year freshwater fish communities in coastal areas of the baltic. *Environmental Biology of Fishes*, *63*(1), 89-101.

Simonit, S., & Perrings, C. (2005). Indirect economic indicators in bio-economic fishery models: agricultural price indicators and fish stocks in lake victoria. *ICES Journal of Marine Science*, *62*(3), 483-492. Retrieved from `https://doi.org/10.1016/j.icesjms.2005.01.012` doi: 10.1016/j.icesjms.2005.01.012

Small, K., Kopf, R. K., Watts, R. J., & Howitt, J. (2014). Hypoxia, blackwater and fish kills: experimental lethal oxygen thresholds in juvenile predatory lowland river fishes. *Plos One*, *9*(4), e94524.

Smith, M. (2007). Generating value in habitat-dependent fisheries: The importance of fishery management institutions. *Land Economics*, *83*(1), 59–73.

Smith, M., Asche, F., Bennear, L., & Oglend, A. (2014). Spatial-dynamics of hypoxia and fisheries: The case of Gulf of Mexico Brown Shrimp. *Marine Resource Economics*, *29*(2).

Smith, M., & Crowder, L. (2011). Valuing Ecosystem Services with Fishery Rents: A Lumped-Parameter Approach to Hypoxia in the Neuse River Estuary. *Sustainability*, *3*, 2229–2267.

Spigarelli, S. A., Jensen, A. J., & Thommes, M. M. (1981). An assessment of the impacts of water intakes on alewife, rainbow smelt and yellow perch populations in lake michigan. Retrieved from `https://www.osti.gov/biblio/6484628`

Statscan. (n.d.). *Statistics canada producer price indexes.* Retrieved from `https://www150.statcan.gc.ca/n1/en/subjects/prices_and_price_indexes/producer_price_indexes`

Stockner, J., Rydin, E., & Hyenstrand, P. (2000). Cultural oligotrophication: Causes and consequences for fisheries resources. *Fisheries*, *25*, 7-14.

Stollery, K. (1986). A short-run model of capital stuffing in the pacific halibut fishery. *Marine Resource Economics*, *3*(2), 137–153.

Storebakken, T., Shearer, K., Roem, A., et al. (2000). Growth, uptake and retention of nitrogen and phosphorus, and absorption of other minerals in atlantic salmon salmo salar fed diets with fish meal and soy-protein concentrate as the main sources of protein. *Aquaculture Nutrition*, *6*(2), 103–108.

Subbey, S., Devine, J. A., Schaarschmidt, U., & Nash, R. D. (2014). Modelling and forecasting stock–recruitment: current and future perspectives. *ICES Journal of Marine Science*, *71*(8), 2307-2322. Retrieved from `https://doi.org/10.1093/icesjms/fsu148` doi: 10.1093/icesjms/fsu148

Svendsen, M., Andersen, N., Hansen, P., & Steffensen, J. (2018). Effects of Harmful Algal Blooms on Fish: Insights from *Prymnesium parvum*. *Fishes*, *3*(1), 11.

Szuwalski, C., & Turnock, J. (2016). A stock assessment for eastern bering sea snow crab. *Stock Assessment and Fishery Evaluation Report for the King and Tanner Crab Fisheries of the Bering Sea and Aleutian Islands Regions*, 167–251.

Tankersley, R. A., & Wieber, M. G. (2000). Physiological responses of postlarval and juvenile blue crabs callinectes sapidus to hypoxia and anoxia. *Marine Ecology Progress Series*, *194*, 179–191.

Thanh, N. (2013). Bioeconomic model of eastern baltic cod under the influence of nutrient enrichment. *Natural Resource Modeling*, *26*(2), 259-280. Retrieved from `https://onlinelibrary.wiley.com/doi/abs/10.1111/j.1939-7445.2012.00137.x` doi: https://doi.org/10.1111/j.1939-7445.2012.00137.x

Thibodeau, B., de Vernal, A., & Mucci, A. (2006). Recent eutrophication and consequent hypoxia in the bottom waters of the lower st. lawrence estuary: Micropaleontological and geochemical evidence. *Marine Geology*, *231*(1), 37-50. Retrieved from `https://www.sciencedirect.com/science/article/pii/S0025322706001277` doi: https://doi.org/10.1016/j.margeo.2006.05.010

Tian, X., Leung, P., & Hochman, E. (1993). Shrimp growth functions and their economic implications. *Aquacultural Engineering*, *12*(2), 81-96. Retrieved from `https://www.sciencedirect.com/science/article/pii/0144860993900187` doi: https://doi.org/10.1016/0144-8609(93)90018-7

Todd, P. A., Ong, X., & Chou, L. M. (2010). Impacts of pollution on marine life in southeast asia. *Biodiversity and conservation*, *19*(4), 1063–1082.

Tomasetti, S. J., Morrell, B. K., Merlo, L. R., & Gobler, C. J. (2018, 12). Individual and combined effects of low dissolved oxygen and low ph on survival of early stage larval blue crabs, callinectes sapidus. *PLOS ONE*, *13*(12), 1-16. Retrieved from `https://doi.org/10.1371/journal.pone.0208629` doi: 10.1371/journal.pone.0208629

Vert-Pre, K. A., Amoroso, R. O., Jensen, O. P., & Hilborn, R. (2013). Frequency and intensity of productivity regime shifts in marine fish stocks. *Proceedings of the National Academy of Sciences*, *110*(5), 1779–1784.

Vis, C., Cattaneo, A., & Hudon, C. (2008). Shift from chlorophytes to cyanobacteria in benthic macroalgae along a gradient of nitrate depletion. *Journal of Phycology*, *44*, 38 - 44. doi: 10.1111/j.1529-8817.2007.00429.x

Wei, L., Zhang, X., Huang, G., & Li, J. (2009). Effects of limited dissolved oxygen supply on the growth and energy allocation of juvenile chinese shrimp, fenneropenaeus chinensis. *Journal of the World Aquaculture Society*, *40*(4), 483–492.

Whittaker, G., Barnhart, B. L., Srinivasan, R., & Arnold, J. G. (2015). Cost of areal reduction of gulf hypoxia through agricultural practice. *Science of the Total Environment*, *505*, 149–153.

Xia, Y., Zhang, M., Tsang, D. C., Geng, N., Lu, D., Zhu, L., … others (2020). Recent advances in control technologies for non-point source pollution with nitrogen and phosphorous from agricultural runoff: current practices and future prospects. *Applied Biological Chemistry*, *63*(1), 1–13.

Zhang, J., & Smith, M. D. (2011). Estimation of a generalized fishery model: A two-stage approach. *Review of Economics and Statistics*, *93*(2), 690–699.

Zhang, Y., Liu, X., Qin, B., Shi, K., Deng, J., & Zhou, Y. (2016). Aquatic vegetation in response to increased eutrophication and degraded light climate in eastern lake taihu: Implications for lake ecological restoration. *Scientific reports*, *6*(1), 23867.

Zheng, J., & Kruse, G. H. (2006). Recruitment variation of eastern bering sea crabs: climate-forcing or top-down effects? *Progress in Oceanography*, *68*(2-4), 184–204.

Zimmerman, R., & Nance, J. (2001). Effects of hypoxia on the shrimp fishery of louisiana and texas. , 293-310. Retrieved from `https://agupubs.onlinelibrary.wiley.com/doi/abs/10.1029/CE058p0293` doi: https://doi.org/10.1029/CE058p0293

Zisserson, B., & Cook, A. (2017). Impact of bottom water temperature change on the southernmost snow crab fishery in the atlantic ocean. *Fisheries Research*, *195*, 12–18.

Appendix A

Chapter 1

A.1 Model assumptions

A.1.1 Chow tests

Chow tests for for equality of coefficients across zones.

H_0: (1) $\beta_2^{zone8} - \beta_2^{zone9} = 0$

(2) $\beta_2^{zone8} - \beta_2^{zone10} = 0$

(3) $\beta_2^{zone8} - \beta_2^{zone12} = 0$

H_1: At least one group has different slope parameters

F(3, 102) = 0.55

Prob $>$ F = 0.6514

Result: Fail to reject null that coefficient on saturation is equal across zones

H_0: (1) $\beta_3^{zone8} - \beta_3^{zone9} = 0$

(2) $\beta_3^{zone8} - \beta_3^{zone10} = 0$

(3) $\beta_3^{zone8} - \beta_3^{zone8} = 0$

H_1: At least one group has different slope parameters

F(3, 99) = 3.87

Prob $>$ F = 0.0115

Result: Reject null and conclude that at least one group has different stock effects

H_0: $\beta_3^{zone8} - \beta_3^{zone9} = 0$

H_1: $\beta_3^{zone8} - \beta_3^{zone9} \neq 0$

F(1, 99) = 0.80

Prob $>$ F = 0.3746

Result: Fail to reject null and conclude that there is no difference in the stock effect between zones 8 and 9

H_0: $\beta_3^{zone8} - \beta_3^{zone10} = 0$

H_1: $\beta_3^{zone8} - \beta_3^{zone10} \neq 0$

F(1, 99) = 2.16

Prob $>$ F = 0.1449

Result: Fail to reject null and conclude that there is no difference in the stock effect between zones 8 and 10

H_0: $\beta_3^{zone8} - \beta_3^{zone12} = 0$

H_1: $\beta_3^{zone8} - \beta_3^{zone12} \neq 0$

F(1, 99) = 8.76

Prob $>$ F = 0.0038

Result: Reject null and conclude that stock effect is different for Zone 12.

A.1.2 Autocorrelation

Durbin-Watson test

H_0: no first order autocorrelation

H_1: positive or negative autocorrelation

Critical region: $1.284 < d < 1.567$ Result: $d > 1.567$ for each zone. Fail to reject the null.

A.1.3 Heteroscedasticity

Bruesh-Pagan test

H_0: Constant variance (homoscedastic error)

H_1: No constant variance (Heteroscedasticity is present)

Result: The null hypothesis was rejected for zones 8 and 12. To correct this, robust standard errors were used.

A.2 Robustness Check

Table A.1: Fixed effects regression using Dissolved Oxygen (DO) instead of oxygen saturation (S). Temperature (T) and salinity (V) are used as other environmental controls that impact the amount of oxygen that dissolves in a water body.

VARIABLES	(1) Growth Rate
DO_{t-1}	0.338***
	(0.116)
T_{t-1}	-12.55
	(11.58)
V_{t-1}	41.86*
	(24.88)
X_{t-1}	-0.000557**
	(0.000233)
$\frac{Q_{it-1}}{X_{t-1}}$	-1.401
	(17.91)
D_8	-112.7***
	(31.99)
D_9	-108.8***
	(39.62)
D_{10}	-91.32**
	(40.28)
$D_{12} * X_{t-1}$	-0.0251***
	(0.00837)
Constant	-1,283
	(815.0)
Observations	112
R-squared	0.237

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

VARIABLES	Zone 8 Growth Rate	Zone 9 Growth Rate	Zone 10 Growth Rate	Zone 12 Growth Rate
S_{it-1}	1.264	1.339	0.485	0.0602
	(0.789)	(0.813)	(0.611)	(1.870)
X_{it-1}	-0.000205	-0.000680	-0.000439	-0.0276**
	(0.000714)	(0.000550)	(0.000356)	(0.0108)
$\frac{Q_{it-1}}{X_{it-1}}$	169.0**	16.08	-76.13	-14.88
	(75.09)	(150.6)	(117.0)	(23.57)
Constant	-111.1	-39.90	23.31	75.43
	(69.18)	(73.03)	(55.62)	(67.17)
Observations	28	28	28	28
R-squared	0.387	0.246	0.101	0.218

Standard errors in parentheses

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

A.3 Simulations

A.3.1 Parameter estimates

A.3.2 Alternate policy simulations

Table A.2: Fixed effects regression using logarithm of Saturation (S). The coefficient on saturation can be viewed as the percentage change.

VARIABLES	Growth Rate
$Log(S_{t-1})$	36.57
	(23.37)
X_{t-1}	-0.000629***
	(0.000231)
$\frac{Q_{it-1}}{X_{t-1}}$	-7.692
	(17.65)
D_8	-68.88**
	(30.07)
D_9	-53.36*
	(30.76)
D_{10}	-40.69
	(30.01)
$D_{12} * X_{t-1}$	-0.0266***
	(0.00834)
Constant	-54.95
	(84.07)
Observations	112
R-squared	0.171

Standard errors in parentheses

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

Table A.3: Fixed effects regression including saturation and a squared saturation term.

	(1)
VARIABLES	Growth Rate
Sat_{t-1}	-0.885
	(1.924)
Sat_{t-1}^2	0.0141
	(0.0154)
X_{t-1}	-0.000600**
	(0.000231)
$\frac{Q_{it-1}}{X_{it-1}}$	-5.164
	(17.64)
D_8	-57.76*
	(31.26)
D_9	-42.61
	(31.73)
D_{10}	-32.16
	(30.43)
$D_{12} * X_{t-1}$	-0.0257***
	(0.00832)
Constant	82.69
	(51.51)
Observations	112
R-squared	0.185

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

Table A.4: Fixed effects regression including an interaction term between stock (X) and saturation (S).

	(1)
VARIABLES	Growth Rate
Sat_{t-1}	2.046**
	(0.783)
$X_{t-1} * Sat_{t-1}$	-3.01e-05*
	(1.60e-05)
X_{t-1}	0.000896
	(0.000830)
$\frac{Q_{it-1}}{X_{it-1}}$	-9.685
	(17.42)
D_8	-95.87***
	(31.70)
D_9	-75.80**
	(31.49)
D_{10}	-66.08**
	(31.94)
$D_{12} * X_{t-1}$	-0.0278***
	(0.00823)
Constant	6.887
	(34.82)
Observations	112
R-squared	0.206

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

Table A.5: Regression of current and previous stock on quota. Parameters are used in simulations to determine harvest, given the assumption that harvest equals quota in each period, as well as in determining price using the isoelastic demand curve.

VARIABLES	(1) Q_{it}	(2) Q_{it}
X_{it}	0.0579***	
	(0.00993)	
X_{it-1}		0.0758***
		(0.00899)
Constant	4,681***	4,070***
	(417.4)	(382.5)
Observations	116	112
R-squared	0.234	0.399
Number of zone	4	4

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

Table A.6: *Case 1* change in consumer surplus with a 5-percentage point increase in oxygen saturation above actual saturation, 1991 - 2018.

Zone	Total change in CS (CAN$)	Average annual change in CS (CAN$)
Esquiman (Zone 8)	7,733,402	276,193
Anticosti (Zone 9)	38,279,557	1,367,127
Sept-$\hat{\imath}$les (Zone 10)	84,707,569	3,025,270
Estuary (Zone 12)	90,110	3,218

Table A.7: *Case 2* welfare change with a 5-percentage point increase in oxygen saturation above actual saturation, 1991 -2018.

Zone	Total change in CS[a] (CAN$)	Total change in Profit[b] (CAN$)	Total change in Welfare[c] (CAN$)	Average annual change in Welfare[d] (CAN$)
Esquiman (Zone 8)	7,733,402	- 1,095,625	6,637,777	237,063
Anticosti (Zone 9)	38,279,557	-5,196,416	33,083,140	1,181,540
Sept-$\hat{\imath}$les (Zone 10)	84,707,569	-8,020,992	76,686,576	2,738,806
Estuary (Zone 12)	90,110	-26,973	63,137	2,255
All Zones	130,810,639	-14,340,007	116,470,631	4,159,665

[a] Total change in consumer surplus (CS) in Canadian dollars (CAN$) is estimated as a present value from 1991 - 2018 using a 5% discount rate.

[b] Total change in profits or producer surplus (PS) in Canadian dollars (CAN$) is estimated as a present value from 1991 - 2018 using a 5% discount rate. Total revenues are calculated using demand elasticity of -0.5.

[c] Total change in welfare in Canadian dollars (CAN$) is estimated as the sum of present value profits and consumer surplus from 1991 - 2018.

[d] Annual average change in consumer surplus (CS) is the present value total, averaged over 28 years (1991 -2018).

Table A.8: Change in profits from 1991 - 2018 when oxygen saturation is increased by 5-percentage points above mean saturation in each zone over the same period, assuming firms are price takers.

Zone		Total discounted profits (CAN$)
Esquiman	Actual	51,605,174
(Zone 8)	5%-point increase	48,532,704
	difference	-6%
Anticosti	Actual	57,219,639
(Zone 9)	5%-point increase	59,015,663
	difference	3%
Sept-$\hat{\imath}$les	Actual	63,418,217
(Zone 10)	5%-point increase	65,456,529
	difference	3%
Estuary	Actual	5,174,620
(Zone 12)	5%-point increase	6,008,202
	difference	16%

Table A.9: Change in profits from 1991 - 2018 when oxygen saturation is increased by 5-percentage points above actual saturation in each zone over the same period, assuming firms are price takers.

Zone		Total discounted profits (CAN$)
Esquiman	Actual	51,605,174
(Zone 8)	5%-point increase	48,413,337
	difference	-6%
Anticosti	Actual	57,219,639
(Zone 9)	5%-point increase	57,715,422
	difference	1%
Sept-îles	Actual	63,418,217
(Zone 10)	5%-point increase	64,763,017
	difference	2%
Estuary	Actual	5,174,620
(Zone 12)	5%-point increase	6,004,039
	difference	16%

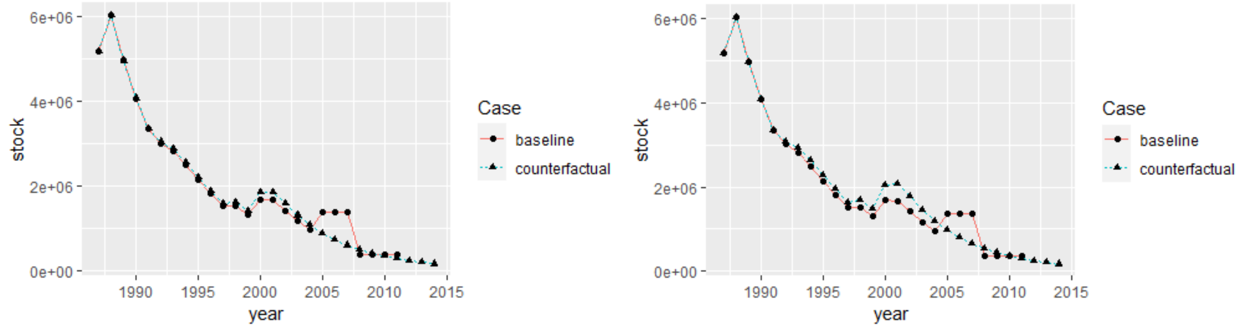
Appendix B

Chapter 2

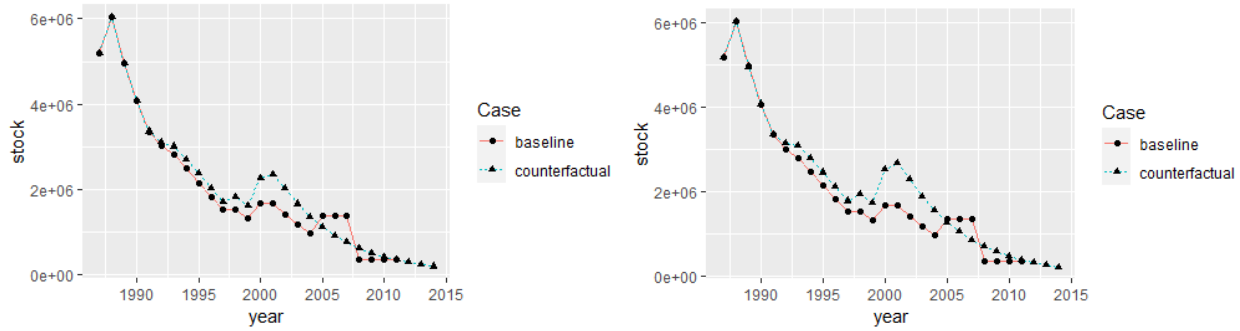
Table B.1: Parameter estimations for SAV and nutrients

	Model 1	Model 2
(Intercept)	1.89^{**}	1.50^{**}
	(0.63)	(0.53)
$logS_{t-1}$	0.18	0.21
	(0.19)	(0.19)
N_{t-1}	-0.28^{**}	
	(0.09)	
P_{t-1}	2.75^{*}	
	(1.31)	
TSS_{t-1}	-0.06^{**}	-0.05^{*}
	(0.02)	(0.02)
NP_{t-1}		-0.01^{***}
		(0.00)
R^2	0.85	0.84
Adj. R^2	0.81	0.81
Num. obs.	27	27

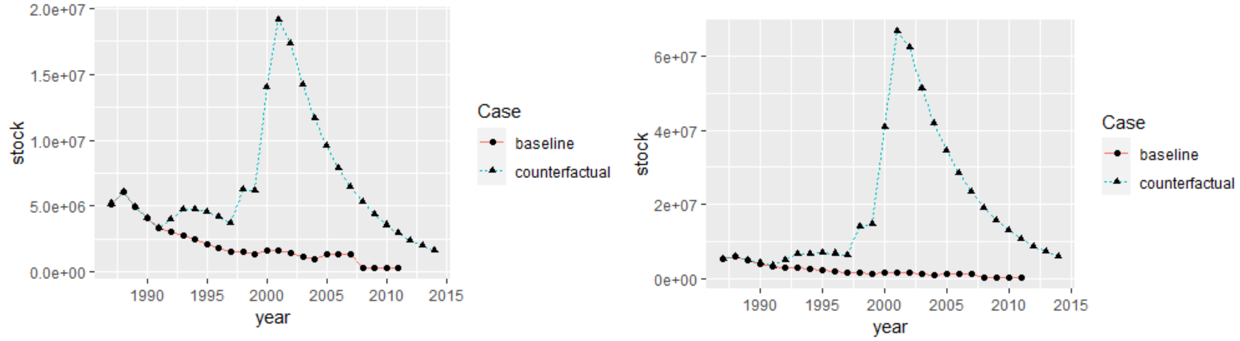
$^{***}p < 0.001$; $^{**}p < 0.01$; $^{*}p < 0.05$



(a) 1% reduction in N:P ratio **(b)** 2% reduction in N:P ratio



(c) 3% reduction in N:P ratio **(d)** 4% reduction in N:P ratio



(e) 15% reduction in N:P ratio **(f)** 20% reduction in N:P ratio

Figure B.1: Comparison of actual Yellow Perch stock against stock under different policy simulations. Note: The figures depict total stock under the baseline scenario using actual N:P ratio versus stock with under various counterfactual policy scenarios.

Appendix C

Chapter 3

C.1 Model assumptions

C.1.1 Chow tests

Chow tests for for equality of coefficients across zones for equations 3.12 and 3.13.

Equation 3.12

H_0: (1) $\alpha_1^{zone12A} - \alpha_1^{zone12B} = 0$

(2) $\alpha_1^{zone12A} - \alpha_1^{zone13} = 0$

(3) $\alpha_1^{zone12A} - \alpha_1^{zone14} = 0$

(4) $\alpha_1^{zone12A} - \alpha_1^{zone15} = 0$

(5) $\alpha_1^{zone12A} - \alpha_1^{zone16} = 0$

(6) $\alpha_1^{zone12A} - \alpha_1^{zone12A} = 0$

H_1: At least one group has different slope parameters

F(6, 33) = 0.80

Prob > F = 0.58

Result: Fail to reject null that coefficient on saturation is equal across zones

Equation 3.13

H_0: (1) $\beta_2^{zone12A} - \beta_2^{zone12B} = 0$

(2) $\beta_2^{zone12A} - \beta_2^{zone13} = 0$

(3) $\beta_2^{zone12A} - \beta_2^{zone14} = 0$

(4) $\beta_2^{zone12A} - \beta_2^{zone15} = 0$

(5) $\beta_2^{zone12A} - \beta_2^{zone16} = 0$

(6) $\beta_2^{zone12A} - \beta_2^{zone12A} = 0$

H_1: At least one group has different slope parameters

F(6, 31) = 0.98

Prob $>$ F = 0.4552

Result: Fail to reject null that coefficient on saturation is equal across zones

C.1.2 Autocorrelation

Durbin-Watson test

Durbin-Watson test for autocorrelation for equation 3.12 and equation3.13.

Equation 3.12

H_0: no first order autocorrelation

H_1: positive or negative autocorrelation

Critical region: $1.284 < d < 1.567$ Result: $d > 1.567$ for each zone. Reject the null of zero autocorrelation. Assume error term is AR(1) distributed.

Equation 3.13

H_0: no first order autocorrelation

H_1: positive or negative autocorrelation

Critical region: $1.284 < d < 1.567$ Result: $d < 1.567$ for each zone. Fail to reject null hypothesis.

C.1.3 Heteroscedasticity

Bruesh-Pagan test

Bruesh-Pagan test for equation 3.12 and 3.13.

Equation 3.12

H_0: Constant variance (homoscedastic error)

H_1: No constant variance (Heteroscedasticity is present)

Result: Fail to reject null hypothesis.

Equation 3.13

H_0: Constant variance (homoscedastic error)

H_1: No constant variance (Heteroscedasticity is present)

Result: Fail to reject null hypothesis.