<u>THREE ESSAYS ON NON-MARKET VALUATION:</u>
<u>VALUING THE BILLY FRANK JR. NISQUALLY NATIONAL WILDLIFE REFUGE</u>

by

Anthony Good
A Dissertation
Submitted to the
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of
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Committee:

Alex Tabarrok ______________________ Director

Bryan Caplan ______________________

____

______________________ Department Chairperson

______________________ Program Director

______________________ Dean, College of Humanities and Social Sciences

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George Mason University
Fairfax, VA

Three Essays on Non-Market Valuation: Valuing the Billy Frank Jr. Nisqually National Wildlife Refuge

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Anthony Good
Master of Arts
George Mason University, 2017
Bachelor of Science
Wright State University, 2015

Director: Alex Tabarrok, Professor
Department of Economics

Spring Semester 2020
George Mason University
Fairfax, VA

DEDICATION

This is dedicated to my beautiful wife Maggie, my supportive mother Brenda, and my two favorite siblings Ashley and Nick.

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I'll first thank my wife, Maggie Good, for her unwavering love and support. Second, I'll thank my mother, Brenda Hart, for always believing in me. Finally, I'll thank my mentor at the U.S. Geological Survey, Emily Pindilli, for introducing me to non-market valuation research and guiding my professional development.

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LIST OF ABBREVIATIONS

Billy Frank Jr. Nisqually National Wildlife Refuge .... Refuge
Ecosystem services conceptual model .... ESCM
Kilometer .... km
Year .... yr
Social cost of carbon .... SCC
United States dollar .... USD
Vedogbeton & Johnston (2020) .... V&J
Willingness to pay .... WTP
Household .... HH
Total economic value .... TEV
U.S Fish & Wildlife Service .... US FWS
National Wetlands Inventory .... NWI
Comprehensive Conservation Plan .... CCP
Hectares .... ha
Integrated assessment model .... IAM
The Interagency Working Group on the Social Cost of Greenhouse Gases .... IWG
Meta-regression model .... MRM
Ecological-economic model .... EEM
Marginal willingness to pay .... MWTP
United States of America .... US
Australia .... AU
Geographic information system .... GIS
Marginal implicit price .... MIP
Beverton-Holt .... B-H
Stock-recruitment .... SR
Marginal willingness to pay .... MWTP
Meters .... m
British Columbia .... BC
Habitat concerns index .... HCI
Habitat quality index .... Q

ABSTRACT

THREE ESSAYS ON NON-MARKET VALUATION: VALUING THE BILLY FRANK JR. NISQUALLY NATIONAL WILDLIFE REFUGE

Anthony Good, Ph.D.

George Mason University, 2020

Dissertation Director: Dr. Alex Tabarrok

The Nisqually Delta Restoration project began in 2009 with the removal of the Brown Farm Dike, inundating 308 ha of the Billing Frank Jr. Nisqually National Wildlife Refuge (Refuge) (Woo et al., 2011). In conjunction with the restoration of tribal lands by the Nisqually Indian Tribe in 1996, the Brown Farm Dike removal constitutes the single largest estuary restoration project in the Pacific Northwest (Woo et al., 2011). The sudden change in tidal flows resulting from the removal of the dike in 2009 augmented the landscape, which impacted the wetland's ecological functions (e.g. nursery for juvenile salmon, carbon sequestration, and flood and flow control) and associated ecosystem services (e.g. nursery support for commercial fishing, climate change mitigation, and flood protection). In this dissertation, I use a suite of non-market valuation methods (i.e. benefit transfer, hedonic price, and production function) to determine the effect of habitat change on the value of a subset of ecosystem services provided by the Refuge.

In Chapter 1, I provide an overview of the suit of ecosystem series provided by the Refuge using an ecosystem service conceptual model (ESCM). The ESCM maps the linkages between direct and indirect drivers of wetland ecosystem change, changes to wetland ecological functions, and changes in the production of ecosystem goods and services provided by the Refuge. I also estimate a baseline value of nursery support for commercial fishing and carbon sequestration ecosystem services provided by the Refuge. The value of nursery support for commercial fishing is calculated using a benefit function transferred from Vedogbeton and Johnston's novel commodity consistent meta-regression model (2020). I find that households in a 50 km radius of the Refuge are willing to pay roughly $1 for a one percentage point increase in fish harvest. The value of carbon sequestration is estimated using carbon flux data over reference and restoring tidal marsh sites and estimates of the social cost of carbon (SCC). My calculations suggest that the avoided monetized damages associated with an incremental increase in one metric ton of carbon emissions each year is $153,083 $tCO_2\ yr^{-1}$ in reference marsh and $39,003 $tCO_2\ yr^{-1}$ in restoring marsh (2018 USD).

In Chapter 2, I estimate the effect of the 2009 Nisqually River Delta Restoration project on property values in Thurston and Pierce Counties in Washington, U.S.A. Economic benefits of the restoration project are estimated by comparing the marginal implicit price to live near the Refuge before and after the removal of the Brown Farm Dike. To assess pre-removal and post-removal marginal willingness to pay, I use property sales data in Pierce and Thurston counties from 2005-2015 and a log-linear hedonic price regression model. The results indicate that the removal of the Brown Farm Dike improved services provided to local homeowners. The pre-removal marginal willingness to pay to live one foot closer to the Brown Farm Dike is -$0.69 (-$1,822 per ½ mile), while the post-removal willingness to pay to live a foot closer to the Brown Farm Dike site is -$0.55 (-$1,452 per 1/2 mile). This assessment indicates that the Nisqually Delta Restoration project increased the marginal willingness to live near the Brown Farm Dike site by $0.14 per foot ($370 per ½ mile). This analysis contributes to the growing body of literature by estimating the effects of a tidal marsh restoration project on housing prices and provides an indication of the ecosystem service value of natural resource management actions.

In Chapter 3, I employ a bioeconomic model to estimate the value of salmon habitat in the Nisqually River Delta. Wetland restoration projects have emerged as powerful tools for reinvigorating wetland productivity and mitigating climate change. The economic tradeoffs associated with wetland restoration are case dependent, which means an assessment of their economic viability needs to be conducted for each individual project. In the Nisqually River Delta, several tidal marsh restoration projects have been completed to improve ecosystem functionality, resulting in changes in salmon habitat. Changing habitat mosaics impacts the productivity of salmon by altering food availability, water characteristics, and opportunities to find shelter. By augmenting the bioeconomic model created by Knowler et al. (2003) and applying it to treaty coho salmon fishing in the Nisqually River Delta, I determine the direct use values attributed to coho salmon habitats in the Nisqually River Delta in the production of treaty commercial coho salmon fishing.

CHAPTER ONE – VALUING THE BILLING FRANK JR. NISQUALLY NATIONAL WILDLIFE REFUGE

1.1 Introduction

There is increasing research interest in ecosystem service valuation. In fact, the number of scientific ecosystem services publications from 2000 to 2008 grew by more than 600 percent (Vihervaara et al., 2010). Likely due to their ability to support vast quantities of biodiversity, wetlands have become one of the most widely studied ecosystems in the ecosystem services literature (Mitsch et al., 2015: Zedler & Kercher, 2005). Unlike many other wetlands in the Pacific Northwest, wetlands in the Refuge provide a unique set of ecosystem goods and services including groundwater recharge, water quality enhancement, carbon storage, and flood control. However, the provisioning of these valuable goods and services are threatened by extensive population growth and land use change.

This study is the first in a series of papers that form the Nisqually River Delta Ecosystem Services Assessment. The goal of the ecosystem services assessment is to estimate the quantity, quality, and value of various ecosystem services provided by the Billy Frank Jr. Nisqually National Wildlife Refuge (Refuge). Not only will I assess the stock of natural capital within the Refuge, I will also determine how the value of ecosystem services change in response to natural and anthropogenic interventions. Understanding the stock and flow of benefits provided by the Refuge improves the ability to understand the trade-offs of management decisions and conservation policies. To help inform policy and national resource management, I use a GIS-based integrated ecological-economic approach to determine the value of wetlands in the Refuge.

In this chapter, I employ the benefit function transfer method to value nursery support for commercial fisheries. Specifically, I rely on a novel meta-analysis conducted by Vedogbeton & Johnston (2020) (V&J). V&J's commodity consistent meta-regression model (MRM) estimates the willingness to pay (WTP) per household (HH) for marsh habitat changes using 139 observations from 23 stated preference studies. Many meta-regression analyses have been conducted to determine the value of wetlands (Brander et al., 2006; Borisova-Kidder, 2006; Brouwer et al., 1999; Chaikumbung et al., 2016; He et al., 2015; Ghermandi et al., 2010; Vedogbeton & Johnston, 2020; Woodward & Wui, 2001). Benefit function transfers, as estimated by meta-regressions, tend to outperform unit value transfers because the benefit function quantitatively accounts for policy and study site differences while unit value transfers only qualitatively accounts for site differences (Rosenberger & Loomis, 2003). By transferring the wetland benefit function in V&J to the Nisqually River Delta, I determine that households are willing to pay $1.28 (2018 USD) for a one percentage point increase in fish harvest supported by the Refuge.

Not only do wetlands provide support to commercial fisheries, they are also net carbon sinks and can sequester up to 2100 gCO_2 m^{-2} $year^{-1}$ (Mitsch et al. 2012). Carbon sequestration regulates the quantity of atmospheric CO_2 by preventing it from being absorbed into the atmosphere. The monetized damages associated with an incremental increase in one metric ton of carbon emissions each year is called the social cost of carbon (SCC or SC-CO_2). By reducing the atmospheric concentration of CO_2, wetlands supply an ecosystem service equal to the value of the SCC. I rely on estimates of the social cost of CO_2 produced by the Interagency Working Group on the Social Cost of Greenhouse Gases, United States Government, to determine the value of carbon sequestration by wetland habitat in the Refuge (2017). The value of the avoided monetized damages associated with an incremental increase in one metric ton of carbon emissions each year is \$153,083 $tCO_2\ yr^{-1}$ in reference marsh and \$39,003 $tCO_2\ yr^{-1}$ in restoring marsh (2018 USD).

Although the benefit transfer approach provides reliable baseline estimates for the WTP for changes in fish harvest, it does not define a direct relationship between wetland habitat change and change in fish harvest. Furthermore, the results for the value attributed to the avoided SCC are merely an annual estimate and do not include habitat change scenarios (e.g. sea level rise, raising of I-5) or natural wetland change associated with the restoring marsh. This paper calls for new primary non-market valuation research to obtain precise estimates of the value of ecosystem services under different land-use change scenarios in the Refuge.

This paper has the following sections: (1) Introduction provides a background on the Nisqually River Delta Ecosystem Services Assessment and the purpose of this paper; (2) Background reviews information on ecosystem services, ecological functions of wetlands, the Billy Frank Jr. Nisqually National Wildlife Refuge, and details the ecosystem services conceptual model; (3) Methods and Data explains the methodology and data employed in this analysis; (4) Results provides benefit transfer results; (5)

Discussion examines the results found in (4); (6) Conclusion summarizes key findings and determines the need for new primary ecosystem services valuations.

1.2 Background

1.2.1 Ecosystem Services

Ecosystem services are benefits humans obtain from nature. Although the term “ecosystem services” was coined in the 1970s, biologist have advocated for the conservation of ecosystems for centuries based on the grounds that ecosystem functions are vital to the production of goods and services (Ehrlich & Ehrlich, 1970; Study of Critical Environmental Problems, 1970). However, until recently, there was no clear consensus from biologist and economists on the appropriate methodology to employ in natural capital valuations. Economists supported the traditional neoclassical approach of cost-benefit analysis, while biologist aimed to devise an alternative system of holistically comparing the benefits humans receive from wildlife resources (Helliwell, 1969). To alleviate the methodological differences, the scientific community has built conceptual ecosystem services models to effectively standardized methodologies and improve the consistency in applying ecosystem service values (Mason et al. 2018).

The value of an ecosystem is derived from direct and indirect human use. Total economic value (TEV) is a measure of the utility gained or lost from the consumption and existence of ecosystem services and includes direct use-value (e.g. food, fuelwood, recreation), indirect-use value (e.g. flood mitigation, soil erosion protection, nutrient cycling), and non-use values (e.g. biodiversity and culture). These values can be further categorized into provisioning services, regulatory services, cultural services, and supporting services (Millennium Ecosystem Assessment, 2005). Table 1 provides a list of wetland specific ecological functions, ecosystem services, and ecosystem service value types.

Table 1: Wetland Ecosystem Services

Ecological Function	Ecosystem Service	Value Type
Nursery for juvenile salmon	Commerical fishing	Direct-use
Nursery for juvenile salmon	Tribal fishing	Direct-use
Productive yield	Timber	Direct-use
Groudwater recharge & discharge	Water supply	Direct-use
Productive yield	Genetic resources	Direct-use
Carbon storage & sequestration	Climate regulation	Indirect-use
Biological productivity and diversity	Disease regulation	Indirect-use
Retention, transformation, & removal of nutrients	Water purification	Indirect-use
Flood & flow control	Flood mitigation	Indirect-use
All ecosystem functions	Aesthetics	Direct-use
All ecosystem functions	Education	Non-use
Association with spiritual and historic information	Tribal heritage	Non-use
Habitat & nursery for plant and animal species	Recreation	Direct-use
Nutrient transformations	Nutrient cycling	Indirect-use
Soil production	Soil formation	Indirect-use

Source : adpated from Brander et al. (2006)

1.2.2 Ecological Functions of Wetlands

The exact definition for wetlands has been in contention in regulatory and nonregulatory circles for nearly a century (Tiner, 1999). Britannica describes wetlands as “complex ecosystems characterized by flooding or saturation of the soil” (Crandell, 2019). The U.S. Fish and Wildlife Service (US FWS) defines wetlands as "lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water" (Tiner, 1999, p. 6). Definitions by other government agencies and research institutes offer similarly broad descriptions. However, recent efforts to create a well-defined classification system has enabled researchers to compare different types of wetlands based on differences in hydrology, vegetation, and soils (Cowardin & Golat, 1995).

The Ramsar Wetland Classification System has become the foundation for international identification of wetland habitats. It classifies wetlands into three main types, i.e. marine/coastal, inland, and man-made. The three broad categories have over 40 total subcategories, e.g. coral reefs, estuarine waters, permanent inland deltas, ponds, etc. Subcategories are broken down even further in the US FWS's National Wetlands Inventory (NWI) database. Greater granularity allows researchers to pinpoint changes in habitat and determine drivers of habitat change.

Wetland functions (e.g. primary productivity, nutrient cycling, floodwater storage, etc.) vary based on wetland type, location, and human intervention. For example, Acremon and Holden (2013) conclude that landscape, topology, soil characteristics, soil moisture, and management largely determine wetlands' influence on flooding. These factors also influence the degree to which wetlands can keep pace with sea level rise. Thorne et al. (2012) determines that wetlands in the Pacific Northwest will fail to keep pace with mid to high sea level rise, resulting in a conversion of coastal wetlands to unvegetated tidal flats. Additionally, this process is estimated to contribute to a decrease in coastal wetlands' soil carbon reservoir, thereby degrading their carbon sequestration capacity.

Other natural processes, e.g. sedimentation and natural filtration, enable wetlands to rid water of pollutants. This valuable ecological function has been a major impetus for the construction of man-made wetlands (Hammer & Bastian, 1989). Not only do local, state, and federal governments acquire and manage wetlands to protect the nation's water supply, they also acquire wetlands to preserve their rich biodiversity. Wetlands are extremely biologically productive and support myriad biodiversity, including almost 5,000 species of plants, 190 species of amphibians, and many species of birds and fish (USDA, n.d.).

1.2.3 Billy Frank Jr. Nisqually National Wildlife Refuge

The Refuge is one of 560 wildlife refuges in the US FWS National Wildlife Refuge System. Located at the mouth of the Nisqually River in the southern end of the Puget Sound, the Refuge provides a sanctuary for many estuarine-dependent species. The roughly 3,114-acre Refuge was established in 1974 to protect fish and wildlife biodiversity. The 2009 Nisqually River Delta restoration project restored 308 ha of estuarine habitat and oversaw the construction of the Nisqually Estuary Boardwalk Trail. The Refuge attracts nearly 300,000 visitors annually by providing opportunities for wildlife watching, environmental education, photography, and hiking.

As an area rich in Native American history, the Nisqually River delta is a significant source of spiritual and tribal heritage. Nisqually Indians have lived along the Nisqually river for thousands of years. However, on December 26, 1854, the Medicine

Creek Treaty was signed, which relinquished tribal ownership rights around the Nisqually River to the U.S. government in exchange for permanent fishing and hunting rights (among other things) (historylink.org, 2003). The treaty enabled European settlers to dike and farm the fertile delta soil. However, the ecological changes caused by diking eventually lead to overgrowth of invasive freshwater vegetation and adversely affected estuarine-dependent salmon habitat.

Furthermore, by the mid-1900s, Washington state failed to uphold the 1854 treaty. With Billy Frank Jr. as the leading advocate for Indian treaty rights, the tribes won back their rights to fish in United States v. Washington, 384 F. Supp (1974). As of 2020, the Refuge and nearby salmon fisheries are co-managed by the US FWS and the Nisqually Indian Tribe. Joint management practices are defined in the Nisqually NWR Comprehensive Conservation Plan (CCP) for 15 years (2005-2020). The CCP lays out the conservation planning process, provides details about habitat use, defines the Refuge boundary expansion, and guides implementation and monitoring. This project coincides with the CCP's partnership opportunities by estimating the ecosystem services provided by the Refuge under different management plans.

1.2.4 Ecosystem Services Conceptual Model

An ecosystem services conceptual model (ESCM) is used to map connections between exogenous factors of wetland change, ecological functions, and socio-economic outcomes at single or multiple study sites (Olander et al., 2018). Exogenous factors or interventions are comprised of direct and indirect drivers of ecosystem change, e.g. management actions, climate change, and development. Ecological functions correspond to the capacity of an ecosystem to provide goods and services. Ecosystem services represent the ESCM's endpoints and reflect changes in social value. ESCM models have been developed for a variety of ecosystems around the world (Kelble et al., 2013; Olander et al. 2018; Schröter et al., 2014). This ECSM maps the linkages between direct and indirect drivers of wetland ecosystem change, changes to wetland ecological functions, and changes in the production of ecosystem goods and services provided by the Refuge (Figure 1).

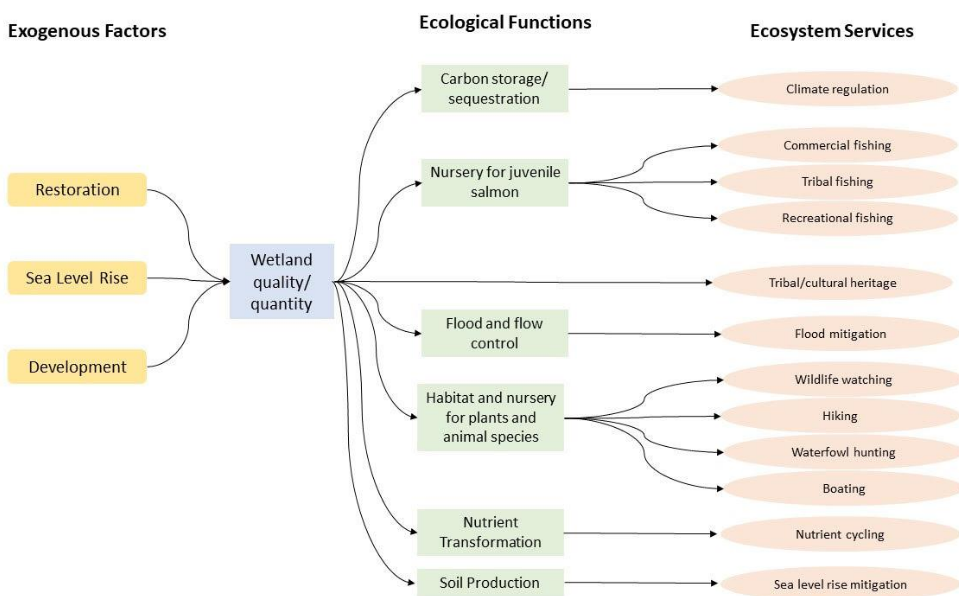

Figure 1: Ecosystem Services Conceptual Model

1.3 Methods and Data

1.3.1 Benefit Transfer Approach

The benefit transfer technique is defined as the use of ecosystem service values from preexisting primary non-market valuation studies conducted at one or many study sites to estimate or predict the value of ecosystem services at one or several unstudied sites (also referred to as policy sites). The benefit transfer method emerged as a useful valuation tool in the 1980s and became very popular in the 1990s due to the relative ease of implementation and low cost compared to other non-market valuation methods (e.g. contingent-valuation survey, travel-cost method, hedonic price method, etc.). Policymakers continue to employ the method to deliver timely monetary estimates of economic value (Richardson et al. 2013). However, a contrast exists between actual implementation in the policy arena and proper implementation defined by academics (Johnston & Rosenberger, 2013). To improve this studies credibility, reliability and replicability, I draw on a host of literature guides (Boyle & Parmeter, 2017; Johnston & Rosenberger, 2015; Richardson et al., 2015; Rosenberger & Loomis, 2017).

There are two types of benefit transfer, namely unit value transfers and benefit function transfers (Johnston et al., 2015). Unit value transfers are defined as the transfer of value from a single study site to one or more policy sites. Alternatively, benefit function transfers estimate the economic value of environmental goods and services at one or more policy sites by using a benefit function that estimates economic value while adjusting for site-specific characteristics. Unit value transfers stem from hedonic price studies, contingent valuation surveys, discrete choice experiments, or any other non-market valuation method. Decision-support tools, such as the Ecosystem Valuation Toolkit and EcoServ, are continuously updated to improve and standardize the valuation of ecosystem services (Bagstad et al. 2013). Standardization is important to improve the overall efficacy of unit transfers.

Unit transfers take less time and resources to implement than benefit function transfers. However, unit transfers can be misleading unless the attributes of the study site are very similar to the policy site. Meta-regression models are commonly used for defining benefit functions for two or more study sites (Johnston et al., 2015). For example, Woodland & Wui (2001) estimate sources of variation in the economic value of wetlands using bivariate and multi-variate regression models. The parameterized regression model represents a wetland valuation function that can be applied to policy sites.

In this study, I use the benefit function transfer approach to estimate the value of changes in commercial fishing harvest attributed to salmon habitat in the Refuge. Specifically, I employ the benefit function estimated in Vedogbeton & Johnston (2020). In the following subsections, I identify ecological functions and ecosystem services in the Refuge, explore exogenous factors of ecosystem change that affect the Refuge, and describe the benefit transfer function used in this valuation.

1.3.2 Ecological Functions and Ecosystem Services

The ESCM (Figure 1) depicts a list of relevant ecological functions and affiliated ecosystem services provided by the Refuge's wetland ecosystem. I identify ecological functions using the *Nisqually National Wildlife Refuge: Final Comprehensive Conservation Plan* (CCP) (US FWS, 2005). The CCP outlines the Refuge's physical environment, vegetation, habitat resources and describes wildlife special uses, recreational opportunities, and cultural resources. Furthermore, I incorporate the information in the CCP with expert opinion from Refuge management to compile a list of the Refuge's ecosystem services (Figure 1).

On August 28, 2019, the U.S. Geological Survey held a stakeholder meeting in Olympia, Washington to identify priority ecosystem services provided by coastal ecosystems in the Nisqually River Delta. Background was provided on the project and stakeholders were asked to provide insight into which services might matter most to their respective communities. A wide range of organizations were represented, including the Nisqually Indian Tribe, US FWS, and City of Lacy (Table 2).

Table 2: Stakeholder Meeting Organizations

Organization
Nisqually Tribe
U.S. Fish and Wildlife Service
Puget Sound Partnership
Washington Dept. of Fish and Wildlife
NRC Citizens Advisory Committee
Nisqually Land Trust
City of Lacey
City of Dupont
Olympia-Lacey-Tumwater Visitor and Convention Bureau
Ducks Unlimited
Nisqually River Foundation
Olympia Coalition of Ecosystem Preservation
Saint Martin's University
Tahoma Audubon Society
Capital Land Trust
Evergreen State College

To determine priority ecosystem services, stakeholders were given five sticky dots to place on one or more posters that represented each of the ecosystem services listed in

Figure 1. Each sticky dot represented a single vote. The activity produced a rank ordering of ecosystem services. Results indicate that the subset of stakeholders in attendance at the stakeholder meeting deem wildlife watching as the most important ecosystem service, followed by education, carbon sequestration, and tribal fishing (Figure 2).

Figure 2: Stakeholder Meeting Priority Ecosystem Services

1.3.3 Wetland Habitat and Exogenous Factors of Ecosystem Change

Ballanti et al. (2017) use an object-oriented hierarchical classification method to classify wetland habitat in the Refuge in 1957, 1980, and 2015. They find that wetland area decreased from 1957 to 1980 by 15 ha (-15 percent) and increased from 1980 to 2015 by 120.5 ha (120 percent). Additionally, forest area declined from 1957 to 2015 by roughly 200 ha. Although the goal of the study was not to determine the causes of wetland area change in the Refuge, they do suggest that wetland change from 1980 to 2015 was influenced by wetland restoration in 2009. Furthermore, they contend that watershed-scale changes in forest habitat may be a response to “both anthropogenic and natural drivers” of habitat change (pp. 22). More specifically, agricultural growth, grassland expansion, and conversion of upland forest to open space as a result of housing development all attribute to the loss of forest habitat. Restricted sediment supply to the Nisqually River Delta also poses concerns about future bank erosion.

To get a better understanding of the natural and anthropogenic ecosystem stressors affecting the Refuge, the U.S. Geological Survey held an ecosystem services scenario planning meeting in Olympia, Washington on August 29, 2019. The meeting addressed manager questions and coordinated scenarios with ongoing research efforts in the Nisqually River Delta. Refuge management determined that sea level rise, urbanization, population growth, and I5 road usage pose the greatest risk to the ecological productivity of tidal marsh. Although I don’t specifically model changes in wetland land types based on risks pointed out by Refuge management in this paper, I do assess the value of ecosystem services affiliated with wetland change.

1.3.4 Fish Harvest and Benefit Function

The Refuge supports recreational, tribal subsistence, and tribal commercial fishing by functioning as a nursery and habitat for juvenile fish species. Recreational Chinook salmon fishing is permitted outside the Refuge and is popular among boat fishermen from July to September. The Nisqually Indian Tribe exercises their legal right to partake in ceremonial and subsistence fishing in and outside of the Refuge. To reduce the risk over overexploitation, tribal fishermen abide by the rules and regulations set out by the Nisqually Fish Commission. Also, selective fishing regulations are used to target hatchery produced Chinook salmon and preserve the population of naturally produced Chinook salmon. Figure 3 shows treaty Chinook, coho, chum, and pink salmon catch in the Nisqually River from 2000 to 2018. Harvests have steadily declined from 2011 to 2018 due to changes in water temperature, water salinity, habitat loss, and other anthropogenic activities (Kendell et al. 2015, Ruff et al. 2017, Zimmerman et al. 2015). It should be noted that I do not rely on specific fish population and harvest data in this study. Instead, I estimate the economic value of a percentage point improvement in fish harvest based solely on a hypothetical change. For more information on fish harvest and populations in the Puget Sound, see Chapter 3 of this dissertation.

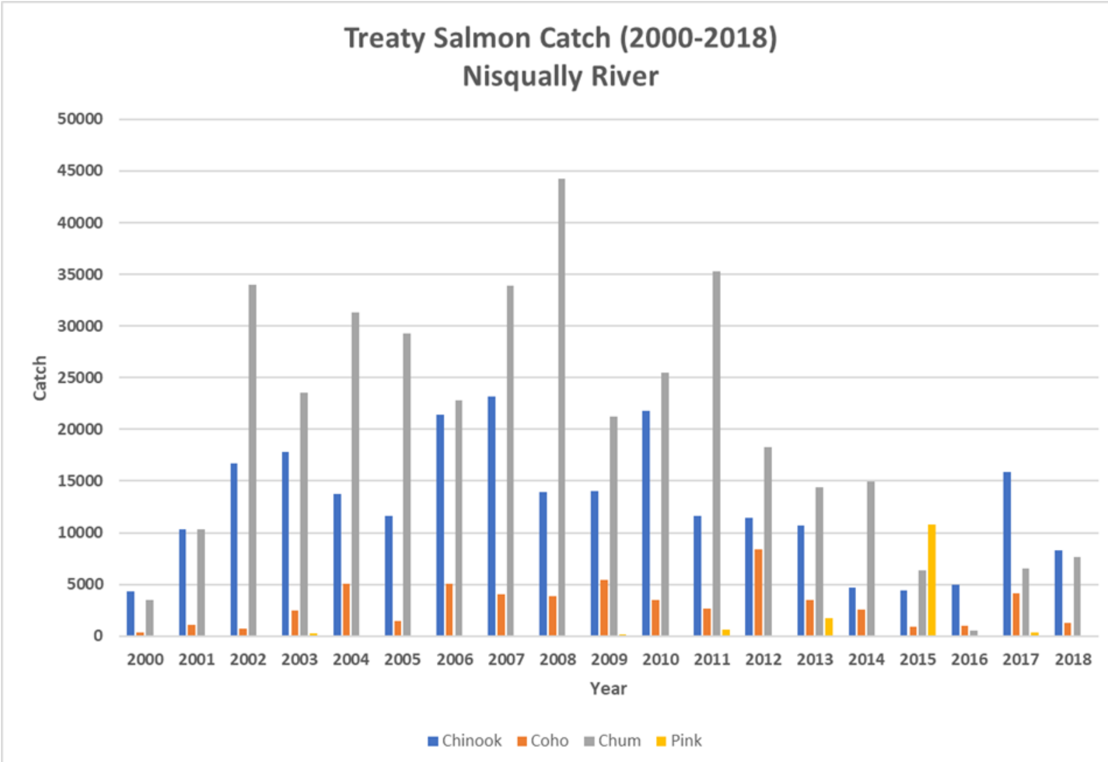

Figure 3: Nisqually River Treaty Salmon Catch

I use the results from the meta-analysis regression in Vedogbeton and Johnston (2020) as the wetland benefit function. V&J estimate the WTP per household for habitat change using 23 stated preference studies conducted from 1990 to 2016 in the US and Canada[1]. They control for differences in income, sampled area, type of change (absolute or relative), commodity change unit (fish harvest, population, or survival), and wetland type (among other things). V&J tested multiple Meta-regression models (MRM) functional forms and compared a commodity consistent MRM with a commodity inconsistent MRM. They determined that the commodity consistent MRM has improved

[1] To view a description of the meta-regression variables, please refer to Tables 15 and 16.

statistical accuracy and benefit transfer performance over the commodity inconsistent MRM.

Keeping the differences in statistical performance in mind, I use the commodity consistent MRM2 functional form where the dependent variable is transformed to willingness to pay (WTP) per percentage point change in habitat (WTP/unit)[2]. The benefit function is specified to estimate WTP per household (HH) per one percent point change in fish harvest. Regressors in the benefit function are calculated in several ways. For example, the size of the wetland area affected by the fish harvest change is determined to be equal to the total area of wetlands in the Refuge in 2015, which, according to Ballanti et al. (2017), is 846.83 acres. V&J incorporate the sampled area of each of the stated preference studies into the meta-analysis. I assume a sample area of roughly 2.18 million acres, which equates to a 50 km radius. I estimate the 2015 weighted median HH income to be $62,755 (Table 3)). Similarly, I assume the peer review regressor is equal to one. The WTP payment is assumed to be paid annually, as opposed to a lump-sum. To stay time consistent, the yearindex is equal to 30 (2015 minus 1985). The mean value of the binary dichotomous variable in V&J's dataset is equal to 0.32. Taking the average into account, I assume the dichotomous regressor is equal to zero. This analysis does concern fish habitat, making the fish habitat regressor equal to one. The Refuge is comprised of salt marsh combined with other wetland habitat, which corresponds to a value of one for the salt other habitat regressor. Finally, the regressor for

[2] See Table 17 for MRM2 results

the intercept is equal to one. A list of the regressors can be found in the benefit function transfer results table in the Appendix (Table 15 and Table 16).

Table 3: Median Household Income Within 50 km of the Refuge (2018 USD)

	2015 Median HH Income	Total Area (km^2)	Area within 50 km of Refuge (km^2)	Area within 50km of Refuge/ Sum of Total Area	2015 Weighted Median HH Income
Mason County	56,851	2,720	1,572	0.18	10,129
Pierce County	63,140	4,680	2,678	0.30	19,163
Kitsap County	69,065	1,470	445	0.05	3,481
Grays Harbor County	46,536	5,760	444	0.05	2,339
Thurston County	65,377	2,000	2,004	0.23	14,849
Lewis County	49,972	6,310	877	0.10	4,964
King County	85,858	5,980	805	0.09	7,830
Total	436,798	28,920	8,824	1.00	62,755

1.3.5 Carbon Flux and Social Cost of Carbon

The value of carbon sequestered by tidal marsh in the Refuge is calculated using carbon flux data from reference and restoring marsh and estimates of the social cost of carbon. Stuart-Haentjens and Windham-Myers (2020) provide carbon flux data for two separate carbon flux towers stationed in reference and restoring marsh in the Refuge. The reference marsh was unaffected by the 2009 Nisqually River Delta Restoration project while the restoring marsh inundated after the removal of the Brown Farm Dike. Both the carbon flux towers in the restoring and reference marsh measure CO_2, CH_4, solar radiation, evapotranspiration, and heat exchange. This is the only instance where carbon flux towers continuously measure CO_2 and CH_4 in resorting and reference marshes.

The SCC is a metric produced to estimate monetized damages associated with an incremental increase in carbon emissions each year (Pearce, 2003). The metric includes "changes in net agricultural productivity, human health, property damages from increased flood risks, and the value of ecosystem services due to climate change" (Interagency Working Group on the Social Cost of Greenhouse Gases, 2016, pp. 3). Estimating these effects involves forecasting macroeconomics and biophysical phenomena using Integrated Assessment Models (IAMs). The Interagency Working Group on the Social Cost of Greenhouse Gases (IWG) employs the DICE, PAGE and FUND IAMs. Their purpose is to provide valid estimates of the SCC to enable government "agencies to incorporate the social benefits of reducing carbon dioxide (CO_2) emission into cost-benefit analyses of regulatory actions" (Interagency Working Group on the Social Cost of Greenhouse Gases, 2016, pp. 3).

Although various IAM estimates have been widely used in academic and regulatory analyses, they have not been barred from scrutiny. Most criticisms of IAMs are related to the assumed degree of climate sensitivity, the ad hoc nature of damage functions, and the uncertainty of catastrophic climate outcomes (Pindyck, 2013). Given that forecasting future greenhouse gas emissions (GHG) and their effect on biological and economic outcomes is necessary to determine the consequences of CO_2 emissions, it's important to analyze underlying model assumptions to determine the ability of SCC estimates to inform regulatory policy decisions (Greenstone et al., 2013). Recent endeavors have been attempted to reconcile and improve SCC estimates (Greenstone et al., 2013; Howard & Sterner, 2017; Kotchen, 2018, Wang et al., 2019). In a meta-analysis by Wang et al., estimates for the SCC range from -$13.36 per ton of CO_2 to $2386.91 per ton of CO_2, with an average of $54.70 per ton of CO_2 (2019). Average values tend to be higher in peer-reviewed journals and when the estimates are made by DICE, RICE, or PAGE models (Wang et al. 2019).

I incorporate SCC estimates from the IWG to determine the economic value of carbon sequestered by reference and restoring marsh in the Refuge. Table 4 gives a summary of the IWG's average SCC estimates at a two and a half, three, and five percent discount rate (2018 USD).

Table 4: Social Cost of CO_2

Year	5% Average	3% Average	2.5% average
2010	12.10	37.51	60.50
2015	13.31	43.56	67.76
2020	14.52	50.82	75.02
2025	16.94	55.66	82.28
2030	19.36	60.50	88.33
2035	21.78	66.55	94.38
2040	25.41	72.60	101.64
2045	27.83	77.44	107.69
2050	31.46	83.49	114.95

Adapted from IWG (2016)

Data on acres of wetlands and carbon flux are incorporated into a simple ecological-economic model (EEM) (Figure 4). The EEM summarizes the workflow for estimating the value of carbon sequestered by wetlands in the Refuge. In 2015, the Refuge consisted of 962 acres of reference marsh and 734 acres of restoring marsh, as mapped by Ballanti et al. (2017). The reference marsh sequesters 2680 g CO_2 m^{-2} yr^{-1} and the restoring marsh sequesters 895 g CO_2 m^{-2} yr^{-1} (Stuart-Haentjens & Windham-Myers, 2020).

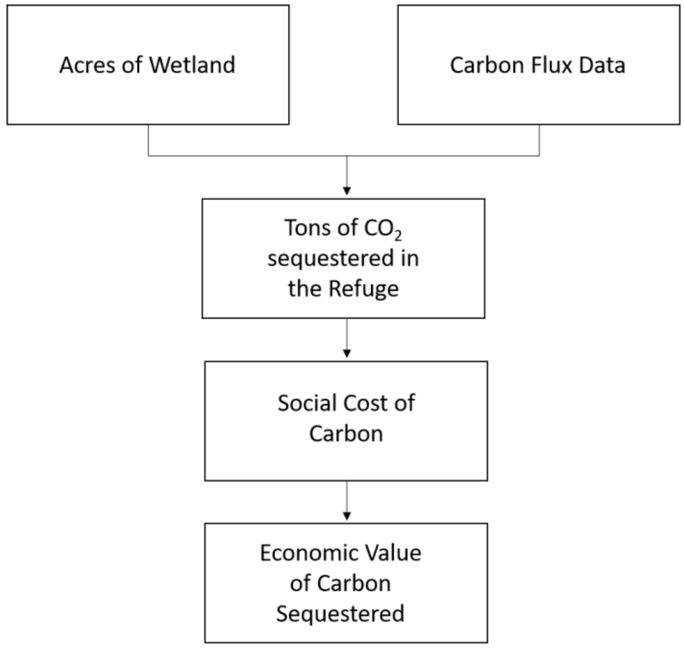

Figure 4: Ecological-Economic Model

1.4 Results

1.4.1 Habitat and Nursery

I calculate the value of habitat and nursery support provided by wetlands in the Refuge using the benefit function in V&J (2020) (Table 5). The calculation is performed by implanting Refuge specific variables into the benefit function. Once the variables have been introduced in the benefit function, they are multiplied by the corresponding benefit function parameters. The sum of the multiplication procedure is equal to the value of wetland habitat in the Refuge. Using the V&J (2020) meta-regression benefit function, I estimate the WTP per HH for a one percentage point increase in fish harvest to be $1.28. WTP per HH for a one percent, five percent, and ten percent increase in fish harvest are presented in Table 6. Because wetlands exhibit diminishing marginal returns to scale, the WTP per HH for a five percent and ten percent increase in fish harvest are additively less than a one percent change (Woodward & Wui, 2001).

By estimating the total number of households in a 50 km range of the Refuge, I can calculate the WTP per hectare for a percentage point change in fish harvest. According to the 2010 census, there are roughly 421,000 HHs in a 50 km radius of the Refuge. Considering the 846.83 ha of affected wetland area, I estimate the WTP per ha of wetlands for a one percent increase in fish harvest to be $637 per ha. Table 6 also provides values for the WTP per ha for five percent and ten percent changes in fish harvest.

Table 5: Benefit Function Transfer Results

Vedogbeton & Johnston 2020			
Variable	Coefficient (β)	Regressors (X)	Xβ
WTP per percentage point habitat change (2018 USD)			1.28
WTP per percentage point habitat change (2016 USD)			1.22
ln(WTP)			0.20
ln(absolute change)	-0.189	0	0
ln(relative change)	0.231	1.00	0.23
ln(sampled area)	-0.158	14.60	-2.31
ln(income)	4.385	11.05	48.44
ln(affected area)	0.138	6.74	0.93
change harvest	-1.354	1	-1.35
change population	-1.244	0	0
change survivial	-0.701	0	0.00
riparian marsh	-1.146	0	0
annual wtp	-2.284	0	0
habitat fish	-2.8	1	-2.80
habitat multiple	-1.502	0	0
dichotomous	-0.318	0	0
peer review	0.716	1	0.72
year index	-0.124	30	-3.72
salt other habitat	-0.0733	1	0
change size		0	0
intercept	-39.87	1	-39.87

Table 6: WTP per Household for Fish Harvest Change

Change in fish harvest	1%	5%	10%
WTP per HH	1.28	3.21	10.21
WTP per hectare of wetlands	637	1599	5079

1.4.2 Carbon Sequestration

In 2015, reference marsh sequestered nearly 12 tCO_2 $acre^{-1}$ and restoring marsh sequestered almost 4 tCO_2 $acre^{-1}$. While considering the total area of reference and restoring marsh (962 acres and 734 acres, respectively) and the IWG's estimate for the SCC, I estimate the value of carbon sequestration to be \$153,083 yr^{-1} for reference marsh and \$39,003 yr^{-1} for restoring marsh. Table 7 provides annual estimates for the economic value of carbon sequestration by wetlands in the Refuge**.**

Table 7: Social Cost of Carbon Avoided

Marsh	Acres	tCO_2 sequestered $acre^{-1}$ $year^{-1}$	tCO_2 sequestered $year^{-1}$	Avoided SCC (2.5%)	Avoided SCC (3%)	Avoided SCC (5%)
Reference	962	12	11,501	\$779,333	\$501,000	\$153,083
Restoring	734	4	2,930	\$198,559	\$127,645	\$39,003

1.5 Discussion

In this section, I compare the results with the literature, address the weaknesses of the benefit function transfer approach, and make suggestions for future research. The amount of CO_2 sequestered by wetlands in the Refuge is lower than the global average. For example, Chmura et al. (2003) estimates the global average for carbon sequestration by tidal saline wetlands to be 220 gC m^{-2} yr^{-1} (roughly 8081 gCO_2 m^{-2} yr^{-1}), which is greater than both the reference marsh (2680 gCO_2 m^{-2} yr^{-1}) and restoring marsh (895 gCO_2 m^{-2} yr^{-1}). However, results are consistent with sequestration in nearby salt marshes. Callaway et al. (2012) determine that tidal marsh in the San Francisco Bay sequesters 111 gC m^{-2} yr^{-1} (roughly 407 gCO_2 m^{-2} yr^{-1}).

Using the benefit function from Brander et al. (2006), Patton et al. (2012) value two national wildlife refuges' ability as a nursery to support commercial fishing at \$24 ha^{-1} yr^{-1}. Schmidt et al. (2011) perform a "rapid ecosystem service valuation" of the Skykomish watershed. Using the Benefit transfer approach, they provide a low value (\$14.38 ha^{-1} yr^{-1}) and a high value (\$5539.72 ha^{-1} yr^{-1}) for habitat and nursery support by wetlands within the Skykomish watershed. Although many papers use the benefit transfer method to calculate the value of nursery support to commercial fishing of wetlands, very few primary papers exist that value wetland salmon habitat in the Pacific Northwest. Knowler et al. (2003) estimate the value of salmon fisheries habitat in the Strait of Georgia to range from $0.93 to $4.95 per ha of watershed. The estimates produced in this paper are consistent with the range estimates in the literature.

The estimate in this study for nursery support for commercial fishing should be regarded as general, rather than a precise, estimate for a couple of reasons. To my knowledge, V&J do not include wetland valuation studies from the Pacific Northwest, which means that my estimates are out of sample. Out-of-sample benefit function transfers are less accurate than in-sample estimates and can lead to misleading valuations. V&J's commodity consistent (i.e. restricted ecosystem type, single ecosystem service, and single valuation method) benefit function reduces transfer error but it does not negate it entirely. V&J conclude that the commodity consistent median out-of-sample transfer error is 41.48 percent. Given the presence of positively skewed transfer error, the results in this paper should be regarded as an upper bound.

I did not devise a wetland habitat change scenario that would contribute to a one, five, or ten percent change in fish harvest. Moreover, the amount of carbon sequestered by wetland habitat in the Refuge will change as the restoring marsh matures and other natural and anthropogenic factors influence marsh productivity. Future research should explicitly model changes in marsh quantity and quality under realistic scenarios (e.g. sea level rise, raising I-5, tidal marsh restoration, etc.). Explicit ecological-economic modelling over a 50 year or more time horizon will allow researchers to provide a more holistic assessment by estimating the present value of a flow of ecosystem services under various habitat change scenarios. In future research, the U.S. Geological Survey aims to develop a suite of modeled spatially-explicit scenarios for the Delta to test how management actions and future stressors will affect coastal ecosystems and ecosystem services.

1.6 Conclusion

I estimate the value of carbon sequestration and nursery support for commercial fishing provided by the Refuge to gain insight about the relationship between changing ecosystem functions and ecosystem service values. Using the benefit function from V&J (2020), I determine that HHs are willing to pay $1.28 (2018 USD) for a one percent increase in fish harvest. Adjusting the WTP for the total number of households in a 50 km radius of the Refuge and 846.83 of affected wetland area, the WTP for a one percent change in fish harvest is $637 per hectare. The value of carbon sequestration in marsh unaffected by the 2009 restoration project is \$153,083 yr^{-1} (2018 USD). Similarly, restoring marsh provides a value of \$39,003 yr^{-1} (2018 USD).

The results found in this paper may be applied to qualitatively support manager decision-making when intervention causes changes in wetland size, nearby population growth, and urban development. However, to better understand the impact of natural and anthropogenic ecosystem stressors on ecosystem services provided by the Refuge, new primary studies must be undertaken. Future research should include connections between changes in ecological functions and their accompanying ecosystem services. Batker et al.

has drawn similar conclusions regarding future research in the Puget Sound. “Dynamic modeling that translates ecosystem change in changes in the delivery and value of ecosystem services would provide a very powerful tool for testing different restoration, land use change, and climate change scenarios” (n.d., pp. 62). Furthermore, it is imperative that cultural ecosystem services are not disregarded and are considered in future research. Cultural services linked to the Refuge are undeniably valuable.

CHAPTER TWO – ESTIMATING THE BENEFITS OF TIDAL MARSH RESTORATION IN THE NISQUALLY RIVER DELTA

2.1 Introduction

The Nisqually Delta Restoration project began in 2009 with the removal of the Brown Farm Dike, inundating 308 ha of the Nisqually National Wildlife Refuge (Woo et al., 2011). In conjunction with the restoration of tribal lands by the Nisqually Indian Tribe in 1996, the Brown Farm Dike removal constitutes the single largest estuary restoration project in the Pacific Northwest (Woo et al., 2011). The sudden change in tidal flows resulting from the removal of the dike in 2009 augmented the landscape, which impacted the wetland's ecological functions (e.g. nursery for juvenile salmon, carbon sequestration, and flood and flow control) and associated ecosystem services (e.g. recreational fishing, climate change mitigation, and flood protection). This paper is part of an ecosystem assessment designed to estimate current and future potential ecosystem services associated with restoration of the Nisqually River Delta as well as determine the impact of various Nisqually National Wildlife management decisions on ecosystem services.

The Refuge was established in 1974 and renamed the Billy Frank Jr. Nisqually National Wildlife Refuge in 2015 after Native American civil rights activist Billy Frank Junior. Located along the southern end of the Puget Sound, the Refuge provides a home to many estuarine dependent animals, including bald eagles, wood ducks, beavers, Chinook salmon, coho salmon, pink salmon, and peregrine falcons (US FWS, 2019). The unique estuarine landscape and diverse group of animals provides opportunities for wildlife watching, environmental education, photography, hiking, and fishing. In 2017, nearly 300,000 individuals visited the Refuge, contributing to regional economic impacts of 111 jobs, $4 million in employment income, and $15 million in total economic output (Banking on Nature, 2019).

The Nisqually Estuary Boardwalk Trail is popular among recreationists and starts at the visitor center and weaves through miles of wetlands until it reaches the 360-degree Puget Sound Viewing Platform, where visitors can see McAllister Creek, Mount Rainer, the Olympics, and Puget Sound Islands (WTA.org). Prior to the 2009 restoration, visitors walked along the Brown Farm Dike Trail to access the Boardwalk. However, the Brown Farm Dike Trail was removed prior to the removal of the Brown Farm Dike in 2009. In 2011, a new boardwalk trail was completed that runs atop the estuary and provides visitors with remarkable opportunities to view tidal changes and spot active wildlife.

This study aims to estimate the impact of the 308 ha Nisqually River Delta Restoration project on nearby property values using the hedonic price method. The hedonic price method is a non-market valuation technique commonly used to estimate the value of environmental goods and services, including air quality, water quality, and damn removal (Greenstone & Chay, 2005; Lewis et al., 2008; Walsh, 2011). Unlike ecosystem service studies that estimate the value of individual environmental amenities, I estimate the value of a suite of environmental goods and services provided by the Refuge. Specifically, I estimate the effects of the Brown Farm Dike removal on the marginal willingness to pay (MWTP) to the live near the Refuge, but do not attempt to define the relationship between changes in the MWTP with changes in individual ecosystem services provided by the Refuge.

To estimate the impact of the removal of the dike on the MWTP to live near the Refuge, I compile a dataset of property sales in the two counties directly adjacent to the Refuge from 2005-2015. The dataset includes property characteristics (lot size, house size, etc.), characteristics of a property's neighborhood (county, school district, nearest metropolitan area), and nearby environmental amenities (parks, trails, lakes, etc.). Additionally, the dataset includes the number of foreclosure sales within a one-mile radius of a property. Failure to account for foreclosures may lead to misinterpretation of the coefficients of the hedonic price regression, which will result in erroneous MWTP estimates (Coulson & Zabel, 2012).

Using a semilogarithmic hedonic price regression model, I estimate the effects of the presence and removal of the Brown Farm Dike on nearby property values. I find that the marginal implicit price at the mean to live a foot closer to the dike is -$0.69 before removal and -$0.55 after dike removal. These results provide evidence that the Nisqually River Delta Restoration project increased the MWTP to live near the Refuge.

This paper has the following sections: (1) Introduction provides general information on the Refuge and the purpose of this paper; (2) Background reviews relevant hedonic price studies regarding various types of wetlands; (3) Methods and Data explains the methodology and data employed in this analysis; (4) Results provides regression results;(5) Discussion expands on the results found in (4); (6) Conclusion summarizes key findings and discusses how this project may be utilized in future studies.

2.2 Background

Wetlands are inherently difficult to value because many of the goods and services they provide are rarely sold in markets and there is limited knowledge about the link between wetland ecosystem changes and human-related benefits (Barbier, 2013). Wetlands ecosystem services include recreation, carbon sequestration, cultural value and biodiversity; however, wetlands also provide disservices, such as odor and insect nuisances. The utility gain (or loss) from the entire set of ecosystem services is called total economic value (TEV). TEV consists of direct use-values, indirect use-values, and non-use values. Direct use-values are derived from consumptive goods, including fresh water, fish, timber, and non-timber forest products. Indirect use-values consist of ecosystem goods and services that are non-consumptive by nature and indirectly contribute to the production or consumption of other goods, e.g. water purification, flood protection, shoreline stabilization, and recreation. Ecosystem goods and services that are not directly or indirectly used may contribute to non-use values. Non-use values are the most difficult to valuate because it includes value non-users place on the present and future existence of the ecosystem (e.g. existence value and bequest value).

Climate change is expected to increase flood risks and reduce global biodiversity (Hirabayashi et al., 2013; Lovejoy & Hannah, 2005). As one of the most productive ecosystems, wetlands play an important role in mitigating climate change (Erwin et al., 2009). Coastal wetlands are valuable buffers against storm surges and support rich biodiversity. Restoration and conservation projects lead to greater ecosystem functionality, which directly impacts the value of ecosystem services and the TEV of restored sites. Although restored wetlands provide less supporting and regulating ecosystem services when compared to natural wetlands, restored wetlands provide 36% more provision, regulation, and supporting ecosystem services than degraded wetlands (Meli et al., 2014). Because maximizing net benefits is typically not the main concern of restoration, pre- and post-restoration costs and benefits are not always estimated, leaving the impact of restoration on TEV unknown. Additionally, methods for estimating TEV (e.g. cost-benefit analysis) can be expensive and may not be economical to perform. Thus, many researchers conduct ecosystem service assessments using a variety of methods including contingent valuation surveys, travel cost models, and hedonic price models to estimate the benefits of wetlands. To capture pre- and post-restoration changes in ecosystem services, this study employs the hedonic price method. This is the first hedonic price study that I am aware of to estimate the impact of the presence (and removal) of a dike on housing prices. There are, however, studies that attempt to estimate the benefits of wetland restoration and the impact of wetland proximity on property values.

Prior hedonic studies suggest that wetlands influence residential property values (Boyer & Polasky, 2004). Lupi et al. (1991) analyze over 18,000 residential properties sold in Ramsey County, Minnesota and determine that total protected wetland acres have a significantly positive effect on prices of residential property. Doss and Taff (1996) determine whether the type of wetland, i.e. forested, scrub-shrub, emergent vegetation and open water, has an impact on residential housing prices in Ramsey County, Minnesota. Their results suggest home buyers prefer scrub-shrub and open-water wetlands compared to forested and emergent-vegetation wetlands. Mahan et al. (2000) find that individuals in Portland, Oregon are willing to pay US$436 to live 1,000 feet closer to wetlands and that increasing the size of the nearest wetland by one acre increases home values by US$24. However, they did not find a significant impact of wetland types on property values.

Tapsuwan et al. (2009) conducted a study in Perth, Western Australia and conclude that the price of a property 943 m away from a wetland increases by AU$42.40 by reducing the distance to the wetland by one meter. Furthermore, they determine that the quantity of nearby wetland is also relevant in the sale price of a home. The sale price will increase by AU$6,976 with the addition of a wetland within 1.5 km of a home. In a study by Frey et al. (2013), the authors establish that Long Beach residents value living close to the Colorado Lagoon. Using a semi-log model and average house prices, they estimate that moving a home 10 percent closer to the lagoon (300 ft) increases property sale prices by US$3,471. A similar study uses the hedonic price method to estimate marginal willingness to pay to be close to a dam (Lewis et al. 2008). Using the mean residential sales price, Lewis et al. estimates the marginal willingness to pay to be farther from the dam to be $2.43 per meter prior to the removal but close to 0 ($0.16) after removal. In other words, the willingness to pay to be near the dam site is negative but becomes less negative after the dam is removed.

2.3 Study Area

Figure 5 depicts the location of the study area within Washington State. The study area is comprised of the total area of Pierce and Thurston Counties. Thurston County has a total area of 2,000 square km and is home to 252,264 residents, with an average income of $32,410 (2017 USD) per capita (US Census Bureau, 2010). There are a few large bodies of water in Thurston, including Long Lake, Lake Lawrence, and Black Lake, as well as several rivers, including the Chehalis River, Skookumchuck River, Deschutes River, and Nisqually River. There are six major cities, with Olympia (the capital of Washington state) being the largest. Pierce County has a total area of 4,680 square kilometers and has a population of 795,225 with an average income of $31,157 (2017 USD) per capita (US Census Bureau, 2010). The largest city in Pierce County is Tacoma, which sits in the Seattle Metropolitan area and is home to Mt. Rainer, the highest point in the state of Washington. Mt, Rainer can be easily seen from the end of Boardwalk Trail at the Refuge.

Figure 5: Study Area

Figure 6 shows the area of interest within the study area including the locations of the Refuge, Brown Farm Dike, and a subset of properties sold during 2005-2015.

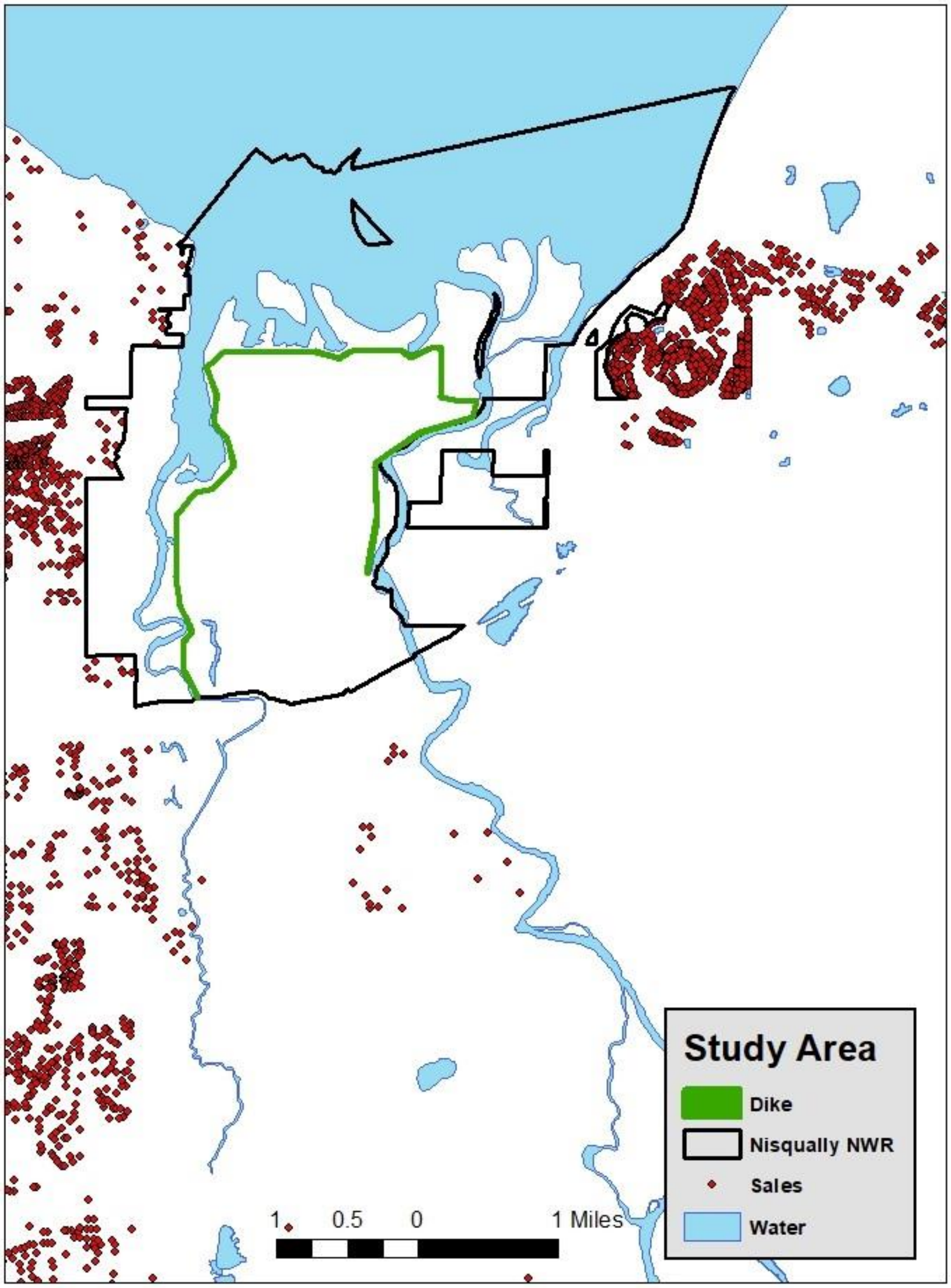

Figure 6: Area of Interest

The Nisqually Indian Tribe has lived for thousands of years in present day Olympia, Tenino and Dupont, Washington. Due largely to the loss of wetlands in the 1990s, the Tribe and surrounding communities risked losing their accumulated stock of cultural and environmental capital tied to salmon populations. Salmon populations in Puget Sound plummeted in the 1990s (Lane & Taylor, 1996; White, 1997), and Chinook Salmon were deemed threatened under the Endangered Species Act of 1973. To combat this loss, the Tribe created a host of natural resources programs to administer and evaluate the effectiveness of habitat restoration and wildlife recovery efforts in the Nisqually Basin, including the Brown Farm Dike restoration project. In 1996, the Tribe began removing dikes on tribal lands. The largest habitat restoration project in the Pacific Northwest was completed in 2009 with the removal of the Brown Farm Dike in the Refuge. Figure 7 provides a spatial overview of the Nisqually estuary restoration program (Cutler, 2010). The Brown Farm Dike was located along the edge of the Refuge and is depicted as the outermost solid red line.

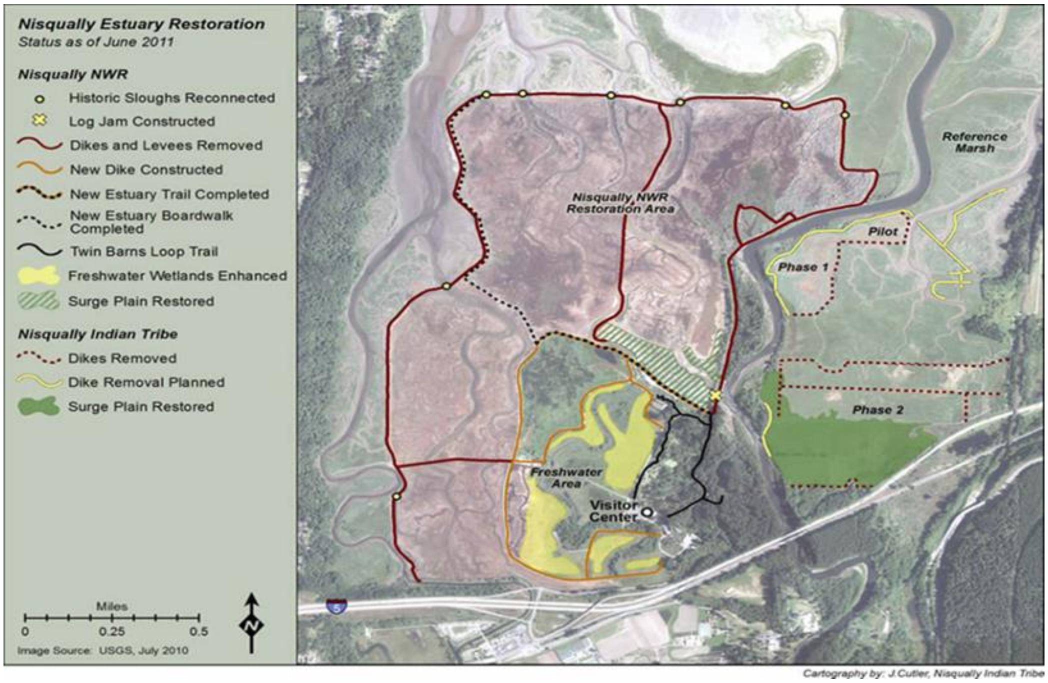

Figure 7: Nisqually Estuary Restoration[3]

The 4-mile long Brown Farm Dike was built in 1904 by Alson Brown to impede tidal flows and cultivate land suitable for agriculture (Nisquallydelatrestoration.org, 2019). Although farming on the tidelands ceased well before the farmland was absorbed into the Refuge in 1974, the Brown Farm Dike prohibited natural tidal flow and fields became dominated by invasive freshwater species, such as red canary grass and common cattail. These species tend to increase sedimentation, creating a shallow wetland that inhibits native vegetation growth (US FWS, 2019).

[3] *Figure 7*. Nisqually Delta Restoration Plan Map. Adapted from "Nisqually Delta Restoration," by J. Cutler, Retrieved December 19, 2019, from Nisquallydeltarestoration.org.

Restoration converted much of the area to salt marsh, killing most of the invasive species population. However, many restored sites have not yet fully recovered and are surrounded by mudflats (Ellings et al., 2016). Figure 8 is a depiction of the change over time of land cover in the Refuge impacted by the removal of the dike. While only a portion of the total impacted area has seen a revitalization of natural vegetation, restoration has increased major channel area by 42 percent delta-wide and 580 percent in restored areas from 2005 to 2011 (Ellings et al., 2016). Additionally, salinity and water temperature have improved among restored and non-restored sites, which are important factors in salmon bioenergetics. Salmon have been seen utilizing restored habitat as early as 1-year post-restoration (Ellings et al., 2016).

Figure 8: July 2009 – March 2010 Land Cover Change[4]

[4] *Figure 4. July 2009 – March 2010 Land Cover Change. Adapted from "Science: Aerial Photography and Remote Sensing by J. Cutler, Retrieved December 19, 2019, from Nisquallydeltarestoration.org.*

2.4 Method

The hedonic price method is a revealed preference valuation technique used to determine the value of nonmarket goods and services, including open space, air quality, water quality, aesthetic value, and cultural heritage. Rosen (1974) built a theoretical hedonic price model based on the consumer theory laid out by Lancaster (1966) in which the utility gained from consuming a good can be derived by the properties of the good itself. In Rosen's model, the price of a differentiated good (e.g. house) can be described by a vector of its characteristics (e.g. structural characteristics, neighborhood attributes, and environmental amenities). The market price for the *h*th house can be written as:

Equation 1: Market Price

$$P_h = P_h(\mathbf{S},\mathbf{N},\mathbf{E})$$

where P_h is the price of house h, **S** is a vector of structural characteristics of house h, $\mathbf{S} = (s_1, s_2,\ldots,s_j)$, **N** is a vector of neighborhood characteristics associated with house h, $\mathbf{N} = (n_1, n_2,\ldots n_n)$ and **E** is a vector of environmental amenities near house h, $\mathbf{E} = (e_1, e_2,\ldots,e_m)$. The Marginal implicit price (MIP) of a characteristic can be found by taking the partial derivative of P_h with respect to that characteristic $\frac{\partial P_h}{\partial e_m}$. The marginal implicit price represents the amount of money needed to purchase a house with one more unit of e_m, all else equal.

In the theoretical hedonic price model, the marginal implicit price is determined by the interaction between consumers and suppliers in a competitive market. Consumers seek to maximize utility

Equation 2: Consumer Utility

$$U = U(X,\mathbf{S},\mathbf{E},\mathbf{N})$$

subject to a budget constraint

Equation 3: Budget Constraint

$$Y = X + p_h$$

where X is a numeraire composite commodity representing all goods other than housing, p_h is the price of a house and Y is income. The first order condition describing the optimal level of the *m*th environmental amenity is satisfied when

Equation 4: Optimality Condition

$$\frac{\partial U_{e_m}}{\partial U_x} = \frac{\partial p_h}{\partial e_m}$$

Optimality is attained when the marginal rate of substitution between environmental amenity e_m and all other goods X equals the marginal willingness to pay for environmental amenity e_m.

The hedonic price function reveals changes in the MWTP for a set of consumers, given marginal changes in house characteristics. However, because the hedonic price function does not directly reveal the inverse demand function for e_m, it cannot be used to reveal changes in MWTP for nonmarginal changes in house characteristics. To solve this problem, Rosen developed a two-step procedure to identify the inverse demand function for e_m. Estimating the inverse demand function relies on consistent estimates of the hedonic price function. Due to the many difficulties associated with estimating the inverse demand function, as described in Bartik (1987), I focus on recovering estimates of the hedonic price function to value marginal changes in house characteristics.

I use a semilogarithmic time dummy hedonic model with an interactive term to estimate the hedonic price function. The function assumes that the price p_{it} of property *i* at time *t* is a function of the structural attributes of the property, characteristics of the neighborhood in which the property is located, and environmental amenities near the property. The standard semilogarithmic hedonic price model is written as:

Equation 5: Standard Semilogarithmic Hedonic Price Model

$$\ln\,(p_{it}) = \beta_0 + \sum_{j=1}^{j} \beta_j S_{ijt} + \sum_{k=1}^{k} \beta_k E_{ikt} + \sum_{n=1}^{n} \beta_i N_{int} + \varepsilon_{it}$$

where β_0 is the intercept, S_{ijt} is the *j*th structural attribute of property *i* at time *t*, E_{ikt} is the *k*th environmental amenity near property *i* at time *t*, N_{int} is the *n*th neighborhood characteristic of property *i* at time *t*, and $\varepsilon_{it}$is the random error term. Many environmental amenities and neighborhood characteristics are represented as distance variables. Each distance variable is measured in feet. Square footage and acreage are specified to be quadratically related to sales price.

I employ an interactive dummy variable to measure the impact of the Brown Farm Dike removal on property sale prices. D_{it} (DDIKE) measures the distance to the Brown Farm Dike of property *i* at time *t* and P_{it} (PDIKE) is a dummy variable equal to 1 for years after dike removal (post 2009) and 0 for years prior to dike removal (pre 2009). The interactive term $P_{it}D_{it}$ is equal to DDIKE × PDIKE. The time dummy variable hedonic model with an interactive term is written as:

Equation 6: Hedonic Model with Interactive Term

$$\ln(p_{it}) = \beta_0 + \sum_{j=1}^{j} \beta_j S_{ijt}$$

$$+ \sum_{k=1}^{k} \beta_k E_{ikt} + \sum_{n=1}^{n} \beta_i N_{int} + \delta_1 D_{it} + \delta_2 P_{it} D_{it} + \varepsilon_{it}$$

The (MIP) of the Refuge prior to the completion of the restoration project is:

Equation 7: MIP Prior to Restoration

$$\frac{\partial p_{it}}{\partial D_{it}} = \delta_1 * p_{it}$$

The (MIP) of the Refuge after the completion of the restoration project is:

Equation 8: MIP After Restoration

$$\frac{\partial p_{it}}{\partial D_{it}} = (\delta_1 + \delta_2) * p_{it}$$

The interpretation of marginal implicit prices as accurate measures of MWTP to live near the Refuge is heavily reliant on Rosen's assumption that the housing market is in equilibrium. Large changes in vacancy rates are an indication that a housing market may not be in equilibrium (Campbell et al., 2011). Following Campbell et al. (2011), I control for the number of foreclosure sales in a 1-mile radius of each property sale in the hedonic price regression to account for otherwise unobserved effects of the 2007-2008 financial crisis. Coulson and Zabel (2013) present two alternative methods for accounting for foreclosures. The first is to include foreclosure sales and the second is to spatially account for the effect of foreclosures on the sales price of nearby homes. I do not include foreclosure sales because there is evidence that the nontrivial number of foreclosures in Thurston County and Pierce County precludes the existence of a single housing market.

Given the existence of two housing submarkets (foreclosed properties and non-foreclosed properties), including foreclosure sales and non-foreclosure sales in a single hedonic price regression results in selectivity bias. Selectivity bias can be avoided by creating two hedonic price regressions, thereby segmenting the housing market into two submarkets. It can also be avoided by explicitly accounting for the impact of foreclosed properties on the sale of non-foreclosed properties in a hedonic price regression. To avoid market segmentation and improve interpretability of the coefficients in the hedonic price regression, I include variable d_i in the hedonic price regression. Variable d_i represents the number of foreclosures in a 1-mile radius of property i at time t.

2.5 Data

I obtained sales data for properties sold in Pierce County and Thurston County from 2005 to 2015. The dataset includes 78,928 property sales with 31,963 sales in Thurston County and 46,965 sales in Pierce County. Sales data for Pierce County was accessed and downloaded from the Pierce County Open Data Portal (https://gisdata-piercecowa.opendata.arcgis.com) and sales data for Thurston County was provided by the Thurston County Assessor's Office (2019). Each sale has a unique identifier and parcel ID. Each parcel ID corresponds to a specific tax parcel number, which I used in conjunction with tax parcel GIS files to geocode each sale. Geocoding each sale is necessary to examine the spatial effects of surrounding environmental and neighborhood characteristics. Table 8 provides a description of the structural, environmental, and neighborhood variables used in this analysis.

Table 8: Variable Descriptions

Variable	Description
Structural Variables	
PRICE18	Sale price of property sold in $2018
YEARSALE	Property sale year
ACRES	Lot size in acres
SQFT	House size in square feet
AGE	Age of home in years
BEDROOM	Number of bedrooms
BATHROOM	Number of bathrooms
STORIES	Number of stories
Neighborhood Variables	
THURSTON	1 = Thurston County, 0 = Pierce County
DSEATTLE	Distance to Seattle, WA in feet
FORECLOSURE	Number of foreclosure sales in a 1 mile radius
Submarket Dummy Variables (ref. location is Steilacoom Historical School District)	
YELMTHURSTON	1 = in district, 0 = otherwise
PUYALLUP	1 = in district, 0 = otherwise
CARBONADO	1 = in district, 0 = otherwise
UNIVERSITYPLACE	1 = in district, 0 = otherwise
SUMNER	1 = in district, 0 = otherwise
DIERINGER	1 = in district, 0 = otherwise
ORTING	1 = in district, 0 = otherwise
CLOVERPARK	1 = in district, 0 = otherwise
PENINSULA	1 = in district, 0 = otherwise
FRANKLINPIERCE	1 = in district, 0 = otherwise
BETHEL	1 = in district, 0 = otherwise
EATONVILLE	1 = in district, 0 = otherwise
AUBURN	1 = in district, 0 = otherwise
WHITERIVER	1 = in district, 0 = otherwise
FIFE	1 = in district, 0 = otherwise
CAPITAL	1 = in district, 0 = otherwise
NORTHTHURSTON	1 = in district, 0 = otherwise
OLYMPIA	1 = in district, 0 = otherwise
RIVERRIDGE	1 = in district, 0 = otherwise
BLACKHILLS	1 = in district, 0 = otherwise
TIMBERLINE	1 = in district, 0 = otherwise
YELM	1 = in district, 0 = otherwise
TUMWATER	1 = in district, 0 = otherwise
TENINO	1 = in district, 0 = otherwise
ROCHESTERHIGH	1 = in district, 0 = otherwise
Environmental Variables	
DWATER	Distance to nearest water in feet
DROAD	Distance to nearest road in feet
DTRAIL	Distance to nearest trail in feet
DPARK	Distance to nearest park in feet
DDIKE	Distance to Brown Farm dike in feet
PDIKE09	1 = post-removal, 0 = pre-removal

ACRES, SQFT, AGE, BEDROOM, BATHROOM, and STORIES are all structural amenities associated with individual properties. Environmental amenities, including DWATER, DPARK, DTRAIL and DROAD, are calculated by measuring the distance from the nearest body of water, park, trail, or road (respectively) to the centroid of each property parcel sold. DROAD includes all major highways and freeways, DWATER includes all ponds, lakes, rivers and streams, and DPARK includes all local, state, and federally owned parks. THURSTON is a dummy variable used to control for county level effects. Zip codes were not included in the dataset, and although the submarket variables are not as spatially granular as zip codes (26 submarkets vs. 100 zip codes), the submarket dummy variables sufficiently replace zip codes as neighborhood fixed effects. PRICE18 is the sales price (2018 USD) of single-family dwellings sold between 2005 and 2015 in Thurston County and Pierce County, Washington, US. The average home sits on 0.47 acres of land, is 1,914 square feet, has roughly 3 bedrooms and 2 bathrooms, and a property value of $335,444 (Table 9).

All single-family dwellings are included in the dataset, including A-Frames, cabins, cottages, colonial, custom, log homes, mansions, split-entry, split-level, 1-story, multiple-story, and Victorian homes. All other building types (commercial, townhomes, apartments, etc.) were removed from the dataset. Additionally, sales without information on the number of bathrooms, bedrooms, square feet, or acres were also removed from the dataset. Summary statistics can be found in Table 9. Property sales may only be geocoded if they are spatially represented as a tax parcel. The property sales without valid parcel numbers were eliminated from the dataset. The centroid of each valid property was calculated, and Thurston and Pierce County shapefiles of waterbodies, parks, and roads were merged to determine near distances. Near distances are calculated using ArcGIS's Generate Near Table (Analysis) tool with the GEODESIC method to ensure all distance estimates are spatially and geodetically accurate. All GIS shapefiles are projected in NAD83_HARN_Washington_South_ftUS.

Table 9: Descriptive Statistics

Variable	Obs.	Mean	Std.Dev.	Min	Max
Structural Variables					
PRICE18	78928	335444.43	1617449.87	13789.14	10929335.14
YEARSALE	78928	2009.36	3.48	2005.00	2015.00
ACRES	78928	0.47	1.27	0.01	49.98
SQFT	78928	1913.61	682.80	144.00	9027.00
AGE	78928	19.97	25.55	0.00	155.00
BEDROOM	78928	3.20	0.90	0.00	9.00
BATHROOM	78928	2.06	0.68	0.00	12.00
STORIES	78928	1.58	0.50	1.00	4.00
Neighborhood Variables					
THURSTON	78928	0.41	0.49	0.00	1.00
DSEATTLE	78928	46282.77	15679.61	14015.80	96260.65
FORECLOSOURE	78928	1.22	3.12	0.00	31.00
Environmental Variables					
DWATER	78928	776.86	570.20	0.00	4704.85
DROAD	78928	654.19	676.96	2.80	7020.89
DTRAIL	78928	4712.88	4059.83	14.16	41182.18
DPARK	78928	1086.06	1042.87	0.00	18032.05
DDIKE	78928	73988.20	37176.60	1330.10	312291.20
PDIKE09	78928	0.51	0.50	0.00	1.00

Figure 9 provides an overlay of foreclosure sales on property sales from 2005-2015. Pierce County had considerably more foreclosure sales (1,966) than Thurston County (254). There is no Thurston County foreclosure data for 2005 and 2006.

Additionally, there are no recorded foreclosures sales in Pierce County in 2005. Foreclosure data was received by request from the Thurston County Treasurer's Office and through the Pierce County Assessor Treasurer website. Because foreclosure data is spatially dependent, I do not interpolate the number of foreclosures in Thurston County during 2005 and 2006. However, I expect the number of foreclosures in Thurston County to be similar to the number of foreclosures in Pierce County during those two years. Thus, failure to include Thurston county foreclosure sales in 2005-2006 likely has a negligible impact on the results. Figure 10 is a line graph that shows the dramatic increase in foreclosures following the 2007-2008 financial crisis, particularly in Pierce County.

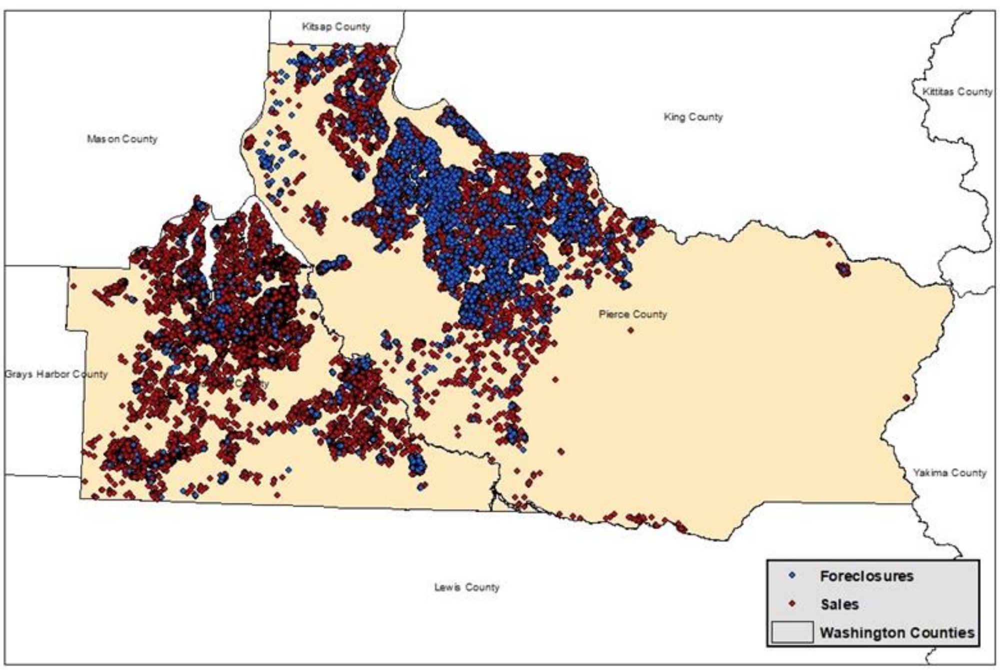

Figure 9: Foreclosures 1

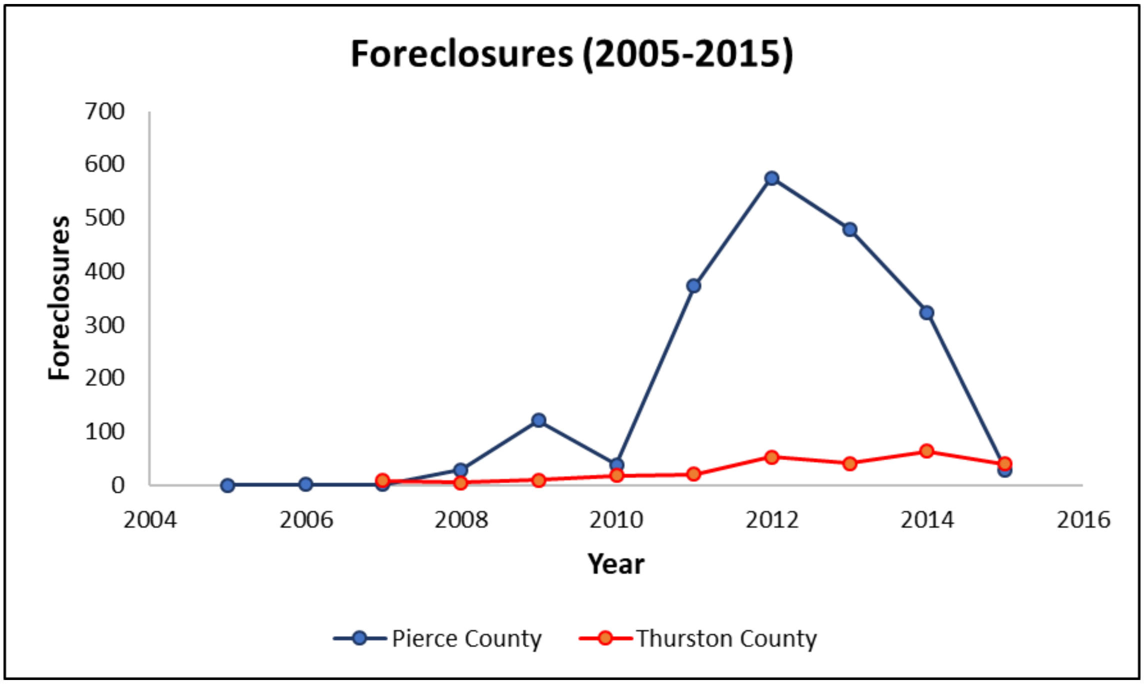

Figure 10: Foreclosures 2

2.6 Results

This section displays and describes the results from four hedonic price regressions and provides the MIP for select property characteristics using regression (4).

2.6.1 Hedonic Price

I run four hedonic price regressions. Regression (1) includes submarket fixed-effects and time fixed-effects. There are *m* submarket dummies (sub) for each property *i* in time period *t*. There are *l* time dummies (time) for each property *i* in time period *t.* Regression (2) includes submarket fixed-effects, time fixed-effects, and the variables of interest (DDIKE and DDIKE*PDIKE09). Regression (3) includes submarket fixed-effects, time fixed-effects, and property characteristics. Regression (4) includes all variables in the dataset, including submarket fixed-effects, time fixed-effects, property characteristics, neighborhood characteristics, environmental amenities, and the variables of interest (DDIKE and DDIKE*PDIKE09). The first four regression equations are listed below.

Equation 9: Regression 1

$$\ln(p_{it}) = \beta_0 + \sum_{m=1}^{25} \beta_m sub_{imt} + \sum_{l=1}^{10} \beta_l time_{ilt} + \varepsilon_{it}$$

Equation 10: Regression 2

$$\ln(p_{it}) = \beta_0 + \delta_1 D_{it} + \delta_2 P_{it} D_{it} + \sum_{m=1}^{25} \beta_m sub_{imt} + \sum_{l=1}^{10} \beta_l time_{ilt} + \varepsilon_{it}$$

Equation 11: Regression 3

$$\ln(p_{it}) = \beta_0 + \sum_{j=1}^{j} \beta_j S_{ijt} + \sum_{m=1}^{25} \beta_m sub_{imt} + \sum_{l=1}^{10} \beta_l time_{ilt} + \varepsilon_{it}$$

Equation 12: Regression 4

$$\ln(p_{it}) = \beta_0 + \sum_{j=1}^{j} \beta_j S_{ijt} + \sum_{k=1}^{k} \beta_k E_{ikt} + \sum_{n=1}^{n} \beta_i N_{int} + \delta_1 D_{it} + \delta_2 P_{it} D_{it} +$$

$$\sum_{m=1}^{25} \beta_m sub_{imt} + \sum_{l=1}^{10} \beta_l time_{ilt} + \varepsilon_{it}$$

Results for the four hedonic price regressions are displayed in Table 10. All results can be found in Tables 20, 21, and 22 in the Appendix. I correct for heteroscedasticity using weighted least squares.

Table 10: Regression Results

	Dependent Varible:			
	log(PRICE18)			
	(1)	(2)	(3)	(4)
DDIKE		3.18E-06***		2.07E-06***
		(9.07E-08)		(5.53E-08)
DDIKE*PDIKE09		-8.45E-07***		-4.18E-07***
		(6.07E-08)		(3.3E-08)
FORECLOSURE				-5.46E-03***
				(2.47E-04)
Structural	No	No	Yes	Yes
Neighborhood	No	No	No	Yes
Environmental	No	No	No	Yes
Submarket FE	Yes	Yes	Yes	Yes
Time FE	Yes	Yes	Yes	Yes
Constant	12.65234***	12.60411***	11.77601***	12.07774***
	(0.0076)	(0.0084)	(0.0067	(0.06)
Observations	78,928	78,928	78,928	78,928
R^2	0.23	0.24	0.76	0.79
Note:		*p<0.1; **p<0.05; ***p<0.01		

Hedonic price regression (1) regresses the sale price in 2018 dollars of each property sold in Thurston and Pierce Counties from 2005-2015 on submarket fixed-effects and time fixed-effects. Submarket fixed effects are represented as twenty-five school districts with Steilacoom Historical School District as the reference location. Time fixed-effects are represented as year dummy variables (1=sale year, 0=otherwise). These variables explain a small portion of the variation of sale price around its mean (R^2 = 0.23). The results of (1) indicate that property buyers consider more than just locational and time fixed-effects when purchasing a home. Six of the 24 listed school districts have a statistically significant positive impact on property sale prices; however, the signs change once neighborhood characteristics and structural attributes are included in the regression. As expected, the housing boom years have a positive impact on sale prices;

however, years after the collapse indicate a slow housing market recovery in Thurston and Pierce Counties. Regression (2) includes submarket fixed-effects, time fixed-effects, and the variables of interest (DDIKE and DDIKE*PDIKE09). DDIKE and PDIKE09 are both statistically significant and suggest the dike removal positively impacts property sale prices.

Regression (3) includes the submarket fixed-effect, time fixed-effects, and structural characteristics of the houses purchased from 2005-2015. Structural characteristics (size of the lot, size of the house, age of the house, number of bedrooms, number of bathrooms, and number of stories), time fixed-effects, and submarket fixed-effects explain nearly 76 percent of the variation of the sale price around its mean. According to regression (3), acreage and square footage have diminishing marginal value, and homebuyers prefer older homes to newer homes, less bedrooms, more bathrooms and fewer stories. The signs on the coefficients for bedrooms and age are likely due to omitted variable bias. Regression (4) corrects for omitted variable bias in (3) by including all variables found in (1), (2), and (3) and by also including county level income, county fixed-effects, distances to the nearest body of water, road, trail, and park.

Regression (4) explains 79 percent of the variation of sale price around its mean. The sign and statistical significance of several coefficients in (1), (2), and (3) are different in (4), eluding to the strong possibility of omitted variable bias in (1), (2), and (3). Of the 24 school districts listed in Table 3, nine are positive and statistically significant at the 0.01 confidence level, including Capital, North Thurston, Olympia, River Ridge, Black Hills, Timberline, Tumwater, Tenino, and Rochester High. The remaining 15 are negatively related with sales price. Like (1) and (2), (4) suggests the years during the housing market boom had a positive impact on sale prices while subsequent time fixed-effects had a negative impact. As expected, ACRES and SQFT are statistically significant and exhibit diminishing marginal value. Homes tend to lose value with age and number of stories, suggesting homebuyers prefer newer ranch style houses. Homebuyers are willing to pay a positive sum for additional bathrooms; however, they are not willing to pay more for additional bedrooms.

Unsurprisingly, individuals prefer to live closer to Seattle. However, homes in Pierce County exhibit a significant price penalty even after controlling for foreclosures, which were more abundant from 2005 to 2015 in Pierce County. Foreclosures negatively impact property prices, suggesting homebuyers discount properties in areas with foreclosures. Walking trails and water bodies positively impact home values. Local, state, and federally owned parks do positively influence property sales prices. The variables of interest, DDIKE and DDIKE*PDIKE09, are both statistically significant. The positive coefficient for DDIKE suggests that the presence of the dike is a disservice. The negative coefficient on DDIKE*PDDIKE09 suggests that the removal of the dike removed some of the penalty on property values, but because the absolute value of the DDIKE coefficient is large than the absolute value of the DDIKE*PDIKE09 coefficient, the removal of the dike did not remove all the disservices associated with the presence of the dike.

2.6.2 Marginal Implicit Price (MIP)

The MIP of a property characteristic is defined as the partial derivate of the hedonic function with respect to that characteristic, all else equal. It represents the additional amount a homebuyer must pay for a home with a marginal improvement in a characteristic. In the case of a log-linear functional form, the coefficient has a semi-elasticity interpretation. By multiplying the coefficient of a variable in the log-linear hedonic price regression by the average sale price of properties in the dataset, I can estimate the hedonic price for that characteristic at the mean. The marginal implicit prices for select variables are presented in Table 11.

Table 11: Marginal Implicit Price of Select Characteristics

Variable	MIP at the Mean
ACRES	25239.42
DWATER	-2.93
DEATTLE	-1.45
FORECLOSURE	-1814.34
DDIKE	0.69
DDIKE*PDIKE09	-0.14

Properties in this dataset are, on average, 74,000 feet from the Brown Farm Dike and the average sale price of properties is $335,444. Each MIP is estimated based on the mean property price and the average distance from the Brown Farm Dike. Thurston and Pierce County homebuyers prefer to live near water and Seattle. In fact, they are willing to pay $2.93 to live one foot closer to streams, rivers, or lakes and $1.45 to live one foot closer to Seattle. Not only do they prefer to live near water and the largest metropolitan area in the Pacific Northwest, they also are willing to pay nearly $25,000 for an additional acre of land. Homebuyers discount the value of properties by $1,814 with the addition of a single foreclosure sale in a 1-mile radius.

Furthermore, according to Thurston and Pierce County residents, the presence of the Brown Farm Dike is a disservice. Prior to the removal of the dike, a property that is situated 74,000 feet from the dike will experience a higher sale price of $0.69 if it were one foot farther away from the dike. After the removal, a property that sits 74,000 feet from the dike will experience a higher sale price of $0.55 if it were one foot farther away from the dike site. These results indicate that the removal of the dike positively influenced the MWTP to live near the Refuge.

2.6 Discussion

The Nisqually Delta restoration project is the largest tidal marsh restoration project in the Pacific Northwest. Extensive post-restoration monitoring has been conducted by the U.S. Geological Survey, US FWS, and Nisqually Indian Tribe (Ballanti, 2017; Ellings, 2009; Woo et al., 2011; Woo et al., 2018). From 1980 to 2015, the Refuge experienced a 54 percent net increase in emergent marsh, resulting in an increase of 120 ha of total wetland area. Much of the change in wetland area during 1980-2015 is due to the removal of the Brown Farm Dike (Ballanti et al., 2017). The revitalized tidal marsh has improved salmon and migratory bird populations (Ellings et al., 2016) and the Refuge has experienced increases in recreational visitation following the Brown Farm Dike removal.

The results provide evidence that the removal of the dike reflects positively on property values in surrounding communities. The validity of the evidence is dependent on the assumptions regarding the hedonic price function holding true. Particularly, it is imperative that the housing market is in equilibrium during the study period and that the property sales occur in a single, non-segmented, housing market. If these assumptions hold true, the marginal implicit prices can be considered accurate estimates of the MWTP for structural attributes, neighborhood characteristics, and environmental amenities.

First, it is not inconceivable that the differences in tax codes, buyer attributes, and other unobservable characterizes of Pierce and Thurston Counties results in two separately heterogonous housing markets. To account for county level differences, I use a county dummy variable (THURSTON). I also control for submarkets within each county using dummy variables. Furthermore, the time-fixed effects may also account for time varying unobservable differences. I do not create two separate hedonic price functions for Pierce County and Thurston County because I have effectively controlled for differences in the two counties and the Refuge resides on the border of both counties.

This model assumes equilibrium in the housing market. Divergences from equilibrium will results in additional random errors, which may lead to incorrect interpretation of the marginal implicit prices as marginal willingness to pay estimates. Specifically, the financial crisis of 2007-2008 may have resulted in systematic biases about the prices of houses. Systematically biased beliefs may result in systematic errors in the estimates of marginal willingness to pay, resulting in the incorrect interpretation of the dike removal on property prices. I account for effects of the financial crisis on home prices by incorporating a variable in the hedonic price regression that represents the number of foreclosed homes in a 1-mile radius of a home sale.

Although incorporating foreclosures in the hedonic price function does not validate or invalidate the equilibrium assumption, it does work toward accounting for the effects of the financial crisis on the underlying health of the housing market. Figure 11 provides evidence that the price of houses in Thurston and Pierce Counties began recovering in 2010. Unaccounted for changes in the housing market may shift housing demand and cause estimates of MWTP to be smaller in recessionary periods and larger in expansionary periods. For this reason, I interpret the change in the MIP to live near the Refuge as a lower bound for a change in the MWTP to live near the Refuge. Additionally, these estimations do not prove causation but do suggest a very strong and statistically significant positive relationship between the removal of the dike and home values in Thurston and Pierce Counties.

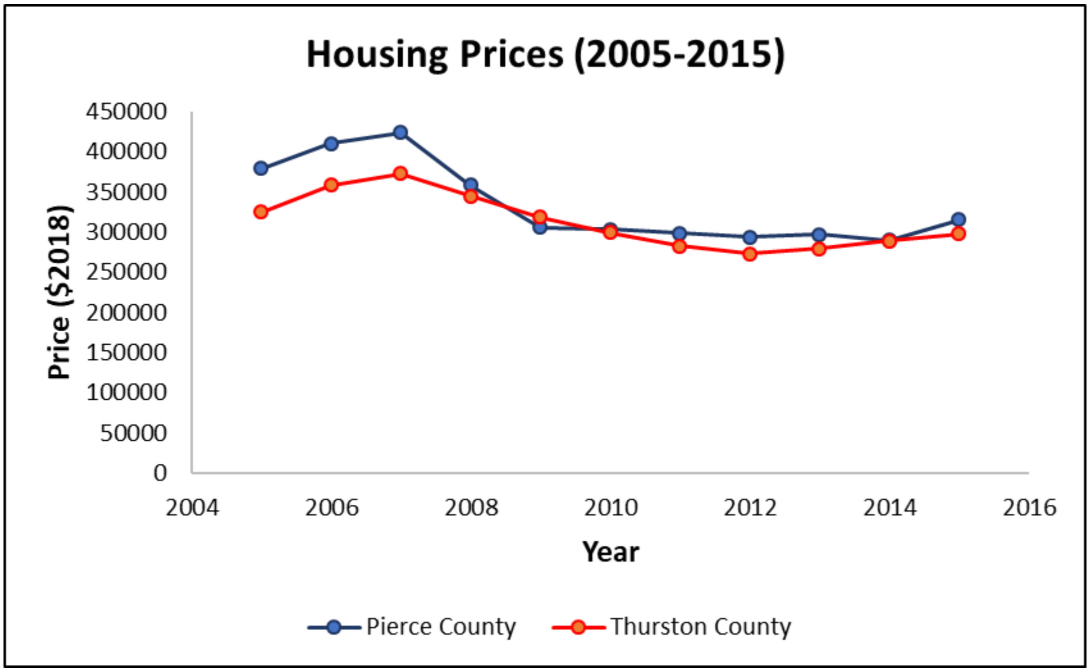

Figure 11: Housing Prices

2.7 Conclusion

Wetlands provide food, water, water filtration, timber, cultural resources, and many other services. They can also reduce sea level rise and mitigate storm flooding. Over half of the global wetland area has been lost in recent decades (Zedler & Kercher, 2005). Wetland restoration projects have emerged as powerful tools for reinvigorating wetland productivity and mitigating climate change. However, because of the heterogeneity in ownership structures, ecosystem types, and cause of degradation, the benefits of restoration cannot be easily transferred from one restored site to another. The economic tradeoffs associated with wetland restoration are case dependent, which means an assessment of their economic viability needs to be conducted for each individual project. The Nisqually River Delta Restoration project has led to fewer nonnative invasive species, greater abundance of migratory birds, and improved salmon prey resources (among other benefits) (Nisquallydeltarestoration.org, 2019). I assess the change in perceived environmental services and disservices associated with the restoration project via property values.

Hedonic price studies that attempt to estimate the impact of estuary restoration on property values face challenges untangling the relationship between changes in ecological functions and the provision of nonmarket ecosystem goods and services. I do not attempt to quantify changes in specific ecological functions or the MWTP for individual ecosystem services. Instead, I assess property values associated with the dike and its removal, which indicates if the restoration project provides benefits to property owners in Pierce and Thurston Counties. The results suggest that the Brown Farm Dike was a disservice to local homeowners and its removal improved nearby home values. Prior to the removal of the dike, a property would fall in value by $0.69 if it were one foot closer to the dike, all else equal. After the dike was removed a property would fall in value by $0.55 if it were one foot closer to the dike, all else equal.

The benefits and costs of restoration is dependent on ecosystem type, e.g. tidal or non-tidal, swamp, marsh, and bog. Ample evidence suggests that the effectiveness of restoration is habitat specific, and more research needs to be conducted to determine the best methods for restoring degraded habitats and assessing the economic implications of restoration (Meli et al., 2014). This paper adds to a growing body of literature on the benefits of wetland restoration by estimating the impact of the removal of a 4-mile-long dike in the Billy Frank Jr. Nisqually National Wildlife Refuge on housing prices in Thurston and Pierce Counties.

CHAPTER THREE – VALUING SALMON HABITAT IN THE NISQUALLY RIVER DELTA

3.1 Introduction

Wetlands are extremely productive ecosystems that provide flood protection, improve water quality, and cultivate biodiversity. The substantial 35 percent decline in global wetland area from 1970 to 2015 has reduced the overall productive capacity of these ecosystems (Ramsar Convention on Wetlands, 2018). Not only has the overall productive capacity of wetlands diminished, many existing wetlands are marginally less productive due to changes in habitat mosaics, water temperature, water flow, and water salinity caused by human activities, e.g. urbanization, industrial development, agriculture, and recreation. Human activities have also led to global sea level rise, which further threatens the ecological production of wetland ecosystems.

Wetlands can overcome sea level rise through accretion, a process where vegetation traps sediment and increases the elevation of the wetland, making them useful tools for mitigating and adapting to climate change. However, they are vulnerable to sea level rise if they are unable to migrate upland or if the sea level rises faster than the natural accretion rate. Failure to adapt to natural or anthropogenic shocks will result in habitat loss and degradation. A successful response will still lead to changes in the distribution of land types within the wetland but is unlikely to affect the total size of the wetland**.** The size, quality, and complexity of wetland land types influences the population dynamics of estuarine-dependent animals, such as fish, birds, and shellfish. Many fish species are particularly sensitive to changes in the size and quality of estuarine habitat. Chinook salmon in the Nisqually River Delta were found to forage in multiple habitat types, suggesting that the "timing, productivity, and diversity" of prey resources in various habitat types influences the productive capacity of the wetlands (Woo et al. 2018).

Despite changing habitat mosaics, the Puget Sound has experienced a steady increase in the total abundance of spawning adult salmon. This improvement is directly linked to the rapid rise in the number of naturally- and hatchery-produced pink and chum salmon, and hatchery-produced Chinook salmon (Losee et al. 2019). Although the abundance of the aforementioned species of naturally-produced salmon have increased, the abundance, survival, and productivity of naturally-produced Chinook and coho salmon have drastically declined over the last 40 years. Scientists attribute the decline to changes in water temperature, water salinity, habitat loss, overexploitation, and other anthropogenic activities (Kendell et al. 2017; Ruff et al. 2017; Zimmerman et al. 2015).

Due to the Nisqually River Delta's geographical significance as migration and rearing habitat for salmonids, several tidal marsh restoration projects have been undertaken to restore salmon habitat and improve ecosystem functionality. The most recent tidal marsh restoration project was completed in 2009 with the removal of the Brown Farm Dike in the Billy Frank Jr. Nisqually National Wildlife Refuge. As the result of the restoration projects, emergent marsh wetland increased by 79 percent from 1957 to

2015 (Ballanti et al., 2017). However, during that same time period, 35 percent of the total marsh acreage was lost (Ballanti et al., 2017)

As the largest tidal marsh restoration project in the Pacific Northwest, it's a challenge for resource managers to assess the economic tradeoffs without the knowledge of how specific decisions impact the value of goods and services provided by the wetland. To better understand how restoration projects and climate change impact the economic value of estuarine habitat, I employ a bioeconomic model developed by Knowler et al. (2003) to determine the economic value of coho salmon habitat in the Nisqually River delta under various habitat quality scenarios. The biology segment of this model is made of a Beverton-Holt (BH) stock-recruitment (SR) relationship and the economic segment is built upon a fisher cost function. The model results suggest that the economic value of treaty commercial coho salmon fishing in the Nisqually River Delta is $5 million under pristine habitat conditions and can fall as low as $2 million with substantial habitat deterioration.

This paper has the following sections: (1) Introduction provides a brief description of changing habitat in the Nisqually River Delta and the purpose of this paper; (2) Background reviews literature on salmon productivity and habitat; (3) Method explains the methodology used in this analysis; (4) Model summarizes the bioeconomic model developed in Knowler et al. (2003); (5) Data provides an overview of the data used in this analysis; (6) Results provides the results of the economic valuation; (7) Discussion examines the results found in (6); (8) Conclusion summarizes key findings and suggests a course for future research.

3.2 Background

The economic value of wetlands is derived from human usage and includes the utility gained or lost from the consumption and existence of goods and services provided by the ecosystem. TEV is a measure of total direct use-values (e.g. food, timber, recreation), indirect-use value (e.g. inundation mitigation, soil erosion protection, nutrient cycling), and non-use values (e.g. biodiversity and culture). Although indirect non-use values and non-use values are tremendously important, especially with regards to climate adaptation and culture significance, respectively, I chose to focus on wetland's direct use-value. Specifically, I estimate the direct-use value of the contribution of the Nisqually River Delta in the production of coho salmon available to treaty commercial fishers.

In order to assess the economic value of the contribution of wetland ecosystems in the production of salmon, I must first establish a relationship between habitat land types, habitat size, habitat capacity, and salmon productivity. Salmon productivity is measured as abundance per unit input (e.g. g/m^2, $adults/m^2$, smolts/km, fry/km, and $parr/m^2$). Productivity measurements are determined based on data availability, research agendas, and management goals and values. Habitat quality and complexity are often measured as indices known as habitat indicators. Habitat indicators can be grouped into five categories: streamflow, water temperature, water chemistry, physical habitat quality, and other habitat quantity and quality indicators (Lewis & Ganshorn, 2007). Additionally, there are several anthropogenic habitat pressures, such as land development, water use, and riparian and foreshore development, that impact the production of salmon (Lewis & Ganshorn, 2007).

Salmon habitat indicators are created via direct measurement using stream gauges, water quality testing equipment, and GIS tools. Salmon productivity, however, is statistically inferred using data on standing stocks. Not only do the methods for estimating salmon productivity differ between studies, models designed to estimate the impact of habitat indicators on salmon productivity also differ. Some models attempt to estimate the spatial and temporal distribution of salmon based on natural biological and ecological processes, e.g. forms of the B-H SR equation and Ricker SR equation. Other models incorporate biological, ecological, and economic processes to determine habitat and population interactions, e.g. bioeconomic models and individual-based models. To better understand the relationship between salmon productivity, habitat, and management actions, I review the literature on salmon behavior, resiliency, and life history in the face of habitat change.

Salmon population models assume a relationship between the population growth rate and population density. Density-independence suggests that the growth rate of a population is independent of the population’s density. Density-dependence suggests the opposite, i.e. the growth rate of a population is dependent on the density of the population. Determining whether a salmon population exhibits density-dependence or density-independence is important because interactions between salmon and their habitat differ based on the stock-recruitment relationship. By using a modified Leslie matrix model to determine the impact of habitat-specific survival rates on ocean-type Chinook salmon populations in the Puget Sound under three separate density-dependent scenarios (juvenile density-independence with a spawning habitat capacity, density-dependent mortality in various juvenile rearing habitats, and density-dependent migration between rearing habitats), Green and Beechie (2004) find that salmon habitat usage differs in each scenario. However, they also conclude that survival rates do not vary among density-dependent scenarios. Similarly, Huntsman et al. (2017) examines the relationship between density-dependence, density-independence, and habitat selection dynamics**.** Their analysis indicates that habitat selection by spawning Chinook salmon in the Chena River, Alaska is best explained by density-dependent processes.

Salmon habitat selection changes based on various density-dependent and density-independent processes. Furthermore, salmon also respond to changes in the distribution of habitat types, i.e. habitat mosaics, caused by natural or anthropogenic disturbances. Using data on strontium isotopic variation extracted from the otolith of 1,377 adult salmon in Alaska, US, Brennan et al. (2019) determine how salmon respond to changing habitat mosaics. They find that changes in habitat mosaics have relatively little impact on the interannual variability in the production of salmon across the Nushagak River watershed. Their results emphasize the importance of coordinating restoration and management efforts across large spatial scales, as focus on small-scale projects may not result in improvement on par with the resiliency across entire intact watersheds. Additionally, they suggest habitat mosaic changes influence the production of ecosystem services. “[W]e show that shifting habitat mosaics play out at large intermediate scales in addition to the well-documented cases on small spatial scales for providing resiliency to ecosystem services" (pp. 3).

Similar to Brennan et al. (2019), Hall et al. (2018) suggest a positive relationship between salmon population resiliency and habitat complexity. Specifically, "watersheds with greater complexity offer greater population resilience with more consistent rates of productivity in the face of environmental variation" (pp.18). Hall et al.'s study focuses on ten river watersheds, including the Nisqually River, that flow into the Puget Sound by exploring the relationship between habitat quantity, habitat complexity, peak river flows, adult spawner densities, and the production of juvenile Chinook salmon. They find a strong positive correlation between habitat complexity and juvenile salmon productivity, suggesting habitat complexity provides a buffer for subyearlings against spawner densities and peak flows. Although ample evidence suggests habitat complexity improves juvenile salmon abundance, juvenile salmon in several different regions are documented utilizing a fraction of suitable area available to them. For example, juvenile Chinook, coho, and sockeye salmon are seen using three-fourths of suitable habitat in summer months in the lower Taku River, BC/US (Murphy et al. 1989). The most influential factors contributing to habitat usage includes water velocity, water turbidity, competition, habitat stability, and distance to spawning habitat.

Extensive research has been done to estimate the impact of habitat quality, habitat complexity, and habitat mosaics on salmon productivity. However, the studies mentioned above do not incorporate the behavioral responses of fisherman to changing salmon productivity attributed to habitat disturbances. Furthermore, many studies have been conducted to estimate the economic value of nonmarket ecosystem goods and services provided by wetlands, but few focus on habitat-fisheries interactions and even fewer specifically explore the relationship between salmon habitat and commercial, recreational, and tribal salmon stocks.

3.3 Method

3.3.1 Approach

A bioeconomic model is an integrated modelling framework that incorporates dynamic ecological population models and fisher behavior functions to estimate optimal values of fish stock, fish harvest, and fishing effort. The optimal values are used to estimate economic value, making bioeconomic models a revealed preference non-market valuation technique. Value in bioeconomic models is revealed, as opposed to stated, through the inclusion of coho salmon market data, including salmon price and fishing costs.

Due to the complexity of salmon population dynamics and the difficulty in estimating salmon productivity and habitat quality indicators, few bioeconomic models specify a relationship between commercial fish stocks and habitat changes (Foley et al. 2011). In fact, in a literature review by Foley et al. (2011), 15 bioeconomic papers consider habitat interactions, while even a smaller number of papers (one) modelled salmon fisheries and salmon habitat. Habitat is often included in bioeconomic models via the stock function, harvest function, or profit function (Foley et al. 2011).

In Knowler et al (2003), they develop a bioeconomic model to calculate the value of coho salmon fisheries habitat in the Strait of Georgia. The habitat value is calculated by comparing the net social benefit of a coho salmon fishery under pristine habitat conditions and the net social benefit under degraded habitat conditions. The difference in the estimates constitute the social gain or loss associated with a change in habitat quality. The population dynamics of coho are described by a delayed recruitment model (Clark 1976, as cited in Knowler et al., 2003). Coho recruitment to the exploitable stock was modeled as a modified Beverton-Holt stock-recruitment function (Hilborn and Walters 1992, as cited in Knowler et al., 2003). Value of habitat capacity and habitat quality parameters were determined in a multi-stage procedure using data from 16 streams. They estimate the value of salmon fisheries habitat in the Strait of Georgia to range from $0.93 to $4.95 per ha of watershed, or $1322 to $7010 per km of spawning stream. Morton et al. (2017) employs the same bioeconomic model to estimate the value of food production, recreational fishing, and nutrient cycling ecosystem services supplied by salmon populations in the Columbia River under different scenarios. They conclude that a re-prioritization of hydropower production would result in a loss of net economic benefits of \$2.2 million yr^{-1} from commercial fishing, nearly \$1 million yr^{-1} from recreational fishing, \$393 thousand yr^{-1} from tribal subsistence fishing, and \$200 yr^{-1} from nutrient cycling compared to the status quo.

I employ the bioeconomic model created by Knowler et al. (2003) to value salmon habitat in the Nisqually River Delta by adjusting the model parameters and using data exclusive to the Nisqually River Delta. There is a difference in the parametrization procedure used in Knowler et al. and this paper. Knowler et al. calibrate two B-H SR parameters (b and Q) using three years of data and then simulate 10 years of coho abundance based on the B-H SR relationship defined by the parametrization. They then find new values for b and Q corresponding to the empirical relationship between their habitat concerns index (HCI) and average rates of change in abundance. I calibrate parameters a and b (Knowler et al. uses literature values for a) over 13 years of coho salmon data and assume habitat quality (Q) enters the B-H SR relationship by influencing the productivity of salmon (a). Moreover, Knowler et al.'s model operates in a commercial troll fishery while the model is this paper is exclusive to treaty commercial fishing.

Using the bioeconomic model, I estimate economic value of salmon habitat in the Nisqually River Delta in a series of six interrelated steps. First, I estimate the parameters of a B-H SR model. Second, I estimate the equilibrium salmon stock, harvest, and fishing effort using the B-H SR model defined in step one. Third, I estimate the baseline economic value of treaty commercially caught coho salmon in the Nisqually River Delta. Fifth, using the parametrization from step one, I estimate a new economic value for the Nisqually river Delta under habitat quality change scenarios. Lastly, I difference the economic value found in step four and step six to find the economic value gain/loss attributed to habitat quality change.

3.3.2 Study Area

This study takes place in the Puget Sound, with specific emphasis on the Nisqually River Delta (Figure 12). The size and shape of the Puget Sound is a result of glacier movement nearly 15,000 years ago (Rice et al., 2015). Its boundary is 2,143 km long, which amounts to a surface area of 2,632 km^2 (Rice et al., 2015). The Puget Sound is roughly 83 percent freshwater with a total volume of 168 km^3, nearly 100 km^3 more than the Chesapeake Bay (Rice et al., 2015). Estuaries in the Puget Sound consist of river deltas, embayments, beaches, and rocky coasts. Although 22 river delta restoration projects were completed from 2006 to 2014, the total length of river deltas has declined by 47 percent, the size of embayments has reduced by 67 percent, the length of beaches has fallen by 8-12 percent, and the length of rocky coasts has diminished by 9.5 percent since the 19^{th} century. (ESRP, 2015; Simenstad et al., 2011).

The focus of this study is the Nisqually River, which starts atop Mount Rainer and feeds 78 miles into the Southern end of the Puget Sound at the Refuge. To improve natural wetland habitat and combat the 301 km^2 loss of wetlands in the Sound since the mid-19^{th} century, the Nisqually River Delta Restoration project (1996, 2006, 2009) restored 308 ha of tidal wetlands (Simenstad et al., 2011). However, two main wetland lands types (emergent marshes and riparian forest) both shrank from 1957 to 2015 (Ballanti et al. 2017). Specific data on coho salmon populations and salmon habitat in the Nisqually River Delta will be discussed in Section 3.5.

3.4 Model

In this section, I describe the bioeconomic model developed by Knowler et al. (2003). I use the same notation for equations (13) – (18) as used in Knolwer et al. (2003). The bioeconomic model is a discrete-time constrained dynamic optimization problem where mangers choose the optimal number of commercially harvested coho salmon to maximize social welfare. Social welfare is defined as:

Equation 13: Social Welfare

$$W(X_t, h_t) = B(X_t) - C(X_t, h_t)$$

where X_t is salmon recruitment at time t, h_t is commercial coho salmon harvest at time t. $B(h_t)$ defines the benefit function and $C(X_t, h_t)$ defines the cost function.

Social welfare is calculated as total producer surplus gained from the production and sale of coho salmon. Social benefits arise solely from the selling of treaty commercial coho salmon, meaning:

Equation 14: Social Benefits

$$B(Xh_t) = ph_t$$

where p is the whole price of fish. The cost function can be derived from a simple relationship between harvest, fishing effort, and fish catchability.

Equation 15: Social Cost Function

$$h(X_t, h_t) = X_t(1 - e^{-qE_t})$$

where e is the natural logarithm, q is the catchability coefficient, and E_t is fishing effort at time t. Solving equation 15 for E_t gives a social cost function equal to:

Equation 16: Social Cost Function Rearranged

$$C(X_t, h_t) = cE_t = \frac{c}{q}[\ln(X_t) - \ln(X_t - h_t)]$$

The initial transition equation describing the population dynamics of coho salmon is written as:

Equation 17: Initial Transition

$$X_t = R(X_{t-3} - h_{t-3})$$

where $X_{t-3} - h_{t-3}$ is equivalent to spawner escapement. The three-year time lag is consistent with the life cycle of coho salmon (figure 13), where recruitment in year t is a function of recruitment in year t-3. Specifically, the recruitment of coho salmon is modeled a Beverton-Holt stock-recruitment function. The B-H and Ricker SR functions are commonly used for salmon population modelling.

Equation 18: Beverton-Holt Stock-Recruitment

$$R(X_{t-3} - h_{t-3}) = \frac{aQ(X_{t-3} - h_{t-3})}{1 + \frac{a}{b}(X_{t-3} - h_{t-3})}$$

where a is a productivity parameter that is interpreted as the number of recruits per spawaner, b is a capacity parameter that is interpreted as peak recruitment, and Q is a habitat quality index that ranges from zero (completely degraded or no habitat) to one (pristine or perfect habitat).

The equilibrium stock (X*) and harvest (h*) are found by solving the following constrained dynamic optimization problem:

Equation 19: Constrained Dynamic Optimization Problem

$$max\ \sum_{t=0}^{\infty} \delta^t (ph_t - \frac{c}{q}[\ln (X_t) - \ln (X_t - h_t)]) \text{ s.t}$$

$$X_t = \frac{aQ(X_{t-3} - h_{t-3})}{1 + \frac{a}{b}(X_{t-3} - h_{t-3})}$$

where δ is the discount term. The problem is set up using a Lagrangian.

Equation 20: Lagrangian

$$L = \sum_{t=0}^{\infty} \delta^t (ph_t - \frac{c}{q}[\ln(X_t) - \ln(X_t - h_t)])$$

$$- \lambda_t \left(X_t - \frac{aQ(X_{t-3} - h_{t-3})}{1 + \frac{a}{b}(X_{t-3} - h_{t-3})} \right)$$

Equation 21: First Order Conditions 1

$$\frac{\partial L}{\partial X_t} = -\delta^t \frac{c}{q}\left(\frac{1}{X_t} - \frac{1}{X_t - h_t}\right) - \lambda_t +$$

$$\lambda_{t-3} \left[\frac{a\left(1 + \frac{a}{b}(X_{t-3} - h_{t-3}) - \frac{a}{b}\left(a(X_{t-3} - h_{t-3})\right)\right)}{[1 + \frac{a}{b}(X_{t-3} - h_{t-3})]^2} \right] = 0$$

Equation 22: First Order Conditions 2

$$\frac{\partial L}{\partial h_t} = \delta^t p - \delta^t \frac{c}{q}\left(\frac{1}{X_t - h_t}\right) -$$

$$\lambda_{t-3} \left[\frac{a\left(1 + \frac{a}{b}(X_{t-3} - h_{t-3}) - \frac{a}{b}\left(a(X_{t-3} - h_{t-3})\right)\right)}{[1 + \frac{a}{b}(X_{t-3} - h_{t-3})]^2} \right] = 0$$

Equation 23: First Order Conditions 3

$$\frac{\partial L}{\partial \lambda_t} = X_t - \frac{aQ(X_{t-3} - h_{t-3})}{1 + \frac{a}{b}(X_{t-3} - h_{t-3})} = 0$$

The optimization procedure results in two equilibrium conditions. Equation (24) is the ecological, or biological, equilibrium. It states that the stock of coho salmon will remain constant when the current stock equals new recruitment. Equation (25) is the economic equilibrium condition. According to Knowler et al., equation (25) “ensures that harvesting occurs so that fish left at sea provide a rate of return just equal to that of financial assets (r)” (pp. 265).

Equation 24: Biological Equilibrium

$$X - \frac{aQ(X - h)}{1 + \frac{a}{b}(X - h)} = 0$$

Equation 25: Economic Equilibrium

$$\frac{aQ\left(1 - \frac{c}{pqx}\right)}{\left[1 + \frac{a}{b}(X - h)\right]^2 \left[1 - \frac{c}{pq(X - h)}\right]} = (1 + r)^3$$

Solving the system of equations yields an economic optimum for X* and h*. The economic value of treaty commercial coho salmon fishing is calculated by estimating net social welfare.

Equation 26: Optimal Social Welfare

$$W(X^*, h^*,) = [B(X^*) - C(X^*, h^*)]$$

3.5 Data

3.5.1 Salmon Data

I use data on coho salmon escapement and treaty commercial coho salmon harvest to estimate historic coho salmon recruitment. Coho salmon escapement data was taken from the Washington Department of Fish & Wildlife website. Total escapement estimates are based on serial spawner counts in index areas in Twenty-fie Mile Creek, Toboton Creek, and Tanwax Creek (WA FWS, 2020). Actual recruitment is not directly observable and must be estimated. As seen in Figure 13, recruits represent the total number of adult salmon returning to coastal waters prior to being exploited by fisherman. Spawners represent the number of salmon that lay eggs. Spawning takes place in the Nisqually watershed up to LeGrande and in McAllister Creek. To estimate recruits, data on harvest is required. Salmon harvest is incorporated using data from the WA FWS (Table 12). Harvest estimates include all treaty commercial catches in the Nisqually River (area 83D) and Fox Island – Nisqually Reach (Area 13). From 2000 to 2018, 62,419 coho salmon were commercially harvest by the Nisqually Indian Tribe, 2,623 by the Puyallup Indian Tribe, and 245 by the Squaxin Indian Tribe. Salmon are caught using a beach seines, tentative tangle nets, experimental pound traps, and gillnets.

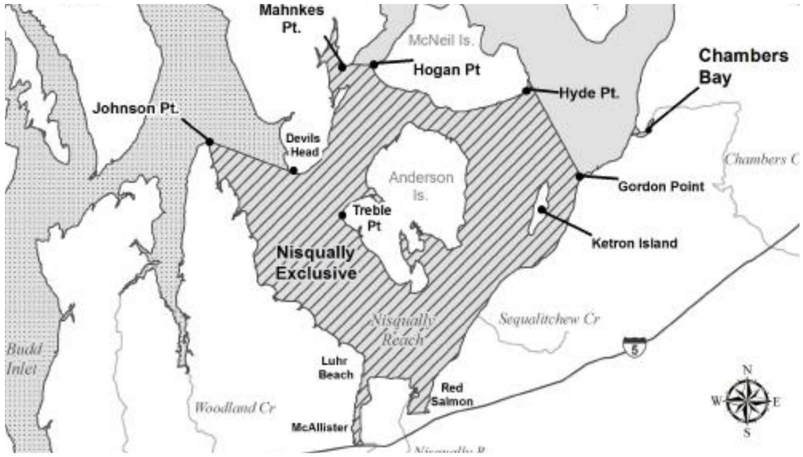

Figure 12: Nisqually Exclusive Harvest Area[5]

Recruits are estimated using this equation: $R_t = S_t + H_t$, where R_t is the number of recruits, S_t is the number of spawners, and H_t is the number of salmon harvested at time t.

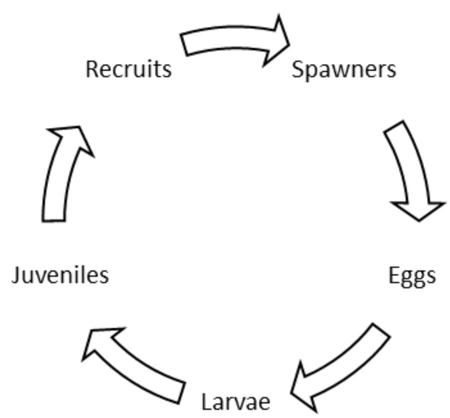

Figure 13: Coho Salmon Life Cycle

[5] Source: Nisqually GIS Program, Nisqually Indian Tribe. (2005). Retrieved from: http://www.nisqually-nsn.gov/files/8714/3864/0912/Map3_CatchAreas_BW_8x11_070715.pdf.

Table 12: Coho Salmon Data

Year	Spawners	Harvest	Recruits
2000	2200	3103	5303
2001	3300	1944	5244
2002	2100	786	2886
2003	4924	2477	7401
2004	6238	7513	13751
2005	1218	2703	3921
2006	1974	5086	7060
2007	3932	4018	7950
2008	808	4271	5079
2009	529	5535	6064
2010	151	3520	3671
2011	220	2630	2850
2012	3891	8390	12281
2013	2136	3499	5635
2014	57	2581	2638
2015	59	881	940
2016	1839	1002	2841

3.5.2 Economic Data

Estimating the value of salmon habitat in the Nisqually River Delta requires information regarding the price of fish sold and the cost of fishing. I extracted coho salmon price data from shenahnamseafood.com. She Nah Nam Seafood is a for profit business managed by Medicine Creek Enterprise Corp. that sells tribally harvested seafood. She Nah Nam Seafood sells various cuts of coho salmon fillets, one of which is called "Native Coho Salmon Portions, PBO, Skin On, Vac Pack (Per Lb. Pricing $15.05ea) (shenahnamseafood.com, 2020). While accounting for the average weight of salmon coho harvested in the Puget Sound (roughly 6.6 pounds), I estimate the price per fish to be $100 (Losee et al. 2019).

Largely due to the fact that treaty commercial fishing information is proprietary, very little information exists on the cost of tribal commercial fishing in the Puget Sound. Furthermore, estimates for catchability of coho salmon using tribal means (gillnet, beach seine, etc.) are also extremely rare. Cox-Rogers and Jantz (1993) examine catchability of sockeye salmon in the Skeena River gillnet test fishery. They report a catchability coefficient for coho salmon of 0.00184. However, because they cannot directly measure escapement in the test fishery of coho salmon, they suggest the estimate is of "questionable validity" (pp.2). Even so, it is likely to be more accurate than catchability coefficient estimates from commercial troll fisheries in the Puget Sound or substance fishing in other parts of the world.

To estimate the cost per fishing day, I use data compiled by Ness (1976) on the annual cost of herring gillnet fishing in Alaska. Ness interviewed skiff operators to determine investment and operating costs of herring gillnetting. In his survey, gillnet vessels supported skiffs and typically two fishermen. He concludes that the total annual cost of herring gillnet fishing is $10,624 (2020 USD). Ness includes operating costs such as groceries, fuel and crew share, and other miscellaneous costs. Following Knolwer et al., I do not include sunk costs (e.g. depreciation and interest) in the annual cost.

The use of gillnets, beach seines, experimental pound traps and gillnets is permitted by the Nisqually Annual Fishing Regulations (2015). In area 83D, gillnetting is permitted three days a week for six weeks a year, beach seining is permitted three daylight days a week for two weeks a year, tentative tangle netting is permitted one daylight day a week for two weeks a year, and the use of experimental pound traps is permitted seven days a week for eight weeks a year. Treaty coho fishing is also permitted in McAllister Creek three to four days a week for eight weeks a year. Assuming tribal fishermen fish every day in September, October, and December for coho salmon, the total daily cost of fishing is $116.75.

3.5.3 Habitat Change Scenarios

The habitat change scenarios in this paper are hypothetical. I do not directly tie changes in the habitat quality index (Q) to historical or future exogenous factors that influence ecosystem change. The habitat quality index ranges from 0 (completely degraded or no habitat) to 1 (pristine or perfect habitat). Nisqually exclusive fishing area is assumed to be in pristine condition (Q=1). The economic value of treaty commercial fishing is estimated under pristine conditions and with degraded habitat (Q=0.75) and poor habitat (Q=0.50). Via the B-H SR relationship, habitat quality augments salmon productivity (a). For example, a quality index of 0.5 lowers salmon productivity by half. In other words, habitat quality deterioration influences the salmon stock-recruitment relationship by lowering the number of recruits per spawner.

3.6 Results

I use the R package called nlsLoop to estimate the parameters of the B-H SR relationship (Padfield, 2015). The fitting procedure results in parameter estimates of 7.12 and 6941 for a and b, respectively, under pristine conditions (Table 13). In pristine conditions, the steady state stock (X*) is equal to 4,972 and the steady state harvest (h*) is equal to 2,511. With habitat degradation corresponding to a Q of 0.75, the steady state stock (X*) is equal to 4,555 and the steady state harvest (h*) is equal to 2,073. Similarly, with habitat degradation corresponding to a Q of 0.5, the steady state stock (X*) is equal to 3,815 and the steady state harvest (h*) is equal to 1,436.

Table 13: Bioeconomic Model Results

Pristine Habitat (Q=1)		Degraded Habitat (Q=0.75)		Poor Habitat (Q=0.5)	
Parameter	Value	Parameter	Value	Parameter	Value
a	7.12	a	5.34	a	3.56
b	6941	b	6941	b	6941
Variable	Value	Variable	Value	Variable	Value
X*	4972	X*	4555	X*	3815
h*	2511	h*	2073	h*	1436

In pristine habitat conditions, the annual economic value of treaty commercial coho salmon fishing in the Nisqually River Delta is $251,050, which equates to a net present value of $5,021,000 (Table 14). By dividing the annual economic value by the number of fish harvested, I can calculate the economic value per coho salmon. The annual economic value per coho salmon harvested is $100 in pristine habitat, $81.42 in degraded habitat, and $79.14 in poor habitat. The Nisqually River watershed is roughly 186,225 ha, which corresponds to an economic value of $26.96 per ha of watershed under pristine conditions (Puget Sound Institute, 2020). The Refuge is 1832.1 ha and resides in the Nisqually River watershed, constituting an economic contribution of $49,393 to the production of commercially available coho salmon.

Table 14: Economic Value of Treaty Commercial Coho Salmon Fishing in The Nisqually River Delta

Pristine Habitat (Q=1)		Degraded Habitat (Q=0.75)		Poor Habitat (Q=0.5)	
Variable	Value	Variable	Value	Variable	Value
Social Benefits	$ 251,050	Social Benefits	$ 207,331	Social Benefits	$ 143,551
Social Costs	$ 44,612	Social Costs	$ 38,527	Social Costs	$ 29,952
Annual Economic Value	$ 251,050	Annual Economic Value	$ 168,804	Annual Economic Value	$ 113,599
Perpeturity (5% discount rate)	$ 5,021,000	Perpetuity (5% discount rate)	$ 3,376,071	Perpetuity (5% discount rate)	$ 2,271,989

3.7 Discussion

The economic value of salmon habitat in the Nisqually River is equal to $26.96 per ha of watershed, with the Refuge contributing $49,393. Because the Refuge's estuarine habitat is vital to the survival of juvenile salmonids and likely contributes a larger share to the productivity of the salmon stock, the estimate of $49,393 should be regarded as a lower bound. These results suggest that a decline in habitat quality that leads to 50 percent reduction in salmon productivity may result in an economic loss of up to $2,749,011. Knowler et al. (2003) found similar results in the in the Strait of Georgia coho fishery. Specifically, habitat degradation in the South Thompson River reduced the economic value of the commercial troll fishery by $1.878 million, which corresponds to a loss of $2.63 per ha of watershed and $3,731 per km of coho stream. Furthermore, in Knowler et al. the optimal exploitation rate fell from 53.3 percent to 19.5 percent, whereas it fell from 50.5 percent to 37.6 percent in this study when the habitat quality index fell by 50 percent.

As the predecessor to this paper, Knowler et al. makes a couple of comments on the weakness of the model that I will reiterate here. First, this model assumes the coho stock in the Nisqually River Delta is managed independently of other fisheries. If it is not, then habitat degradation may cause fishing area closures to protect other salmon species, e.g. Chinook and pink salmon, which would cause the value estimates here to be less reliable. Second, this analysis does not consider recreational or tribal subsistence fishing that occurs in the delta. Undoubtedly, the Nisqually Indian Tribe relies on non-commercial subsistence fishing to meet their calorie needs. Also, recreational salmon fishing is popular in the delta. Although I do not value the contribution of the Nisqually River Delta in the production of salmon caught by subsistence and recreational fishers, it is undeniably economically valuable as rearing habitat for salmonids.

It is necessary to consider the institutional structure (property rights regime) that governs the fishery when using bioeconomic models. In this case, I assume optimal management of coho salmon in the Nisqually River Delta, which implies maximizing economic returns, as opposed to an open access fishery, where economic rents dissipate to zero (Knowler et al., 2003). Closing Nisqually exclusive fishing areas to commercial and recreational fishers when salmon runs are low due to degraded habitat is an economical decision to avoid financial losses. However, habitat change will endogenously affect the price of coho salmon. The economic value may be understated under degraded and poor habitat conditions as the price may rise as the supply of coho salmon decreases. The Nisqually River has been closed to sport fishing and treaty commercial fishing multiple times in the last decade. Both recreational and commercial fishing closures are intended to increase the number of salmon returning to spawn. In some cases, closures are necessary to restore overexploited stocks, while in other cases they aim to restore salmon stocks that have been depleted due to unforeseen natural causes (e.g. drought).

3.8 Conclusion

In order to enhance native estuarine habitats in the Nisqually River Delta, 308 ha of wetlands have been restored in the Refuge. As the largest tidal marsh restoration project in the Pacific Northwest, it may be difficult for resource managers to assess tradeoffs without the knowledge of how specific decisions may impact the value of goods and services provided by the habitat. Understanding how decisions impact the economic value of estuarine habitat may help support the recovery and protection of endangered species and their habitats. To support management decision making and policy design, I employ a production function non-market valuation technique to estimate the economic value of salmon habitat in the Nisqually River Delta. The results provide a clear picture, i.e. salmon habitat in the Refuge is extremely valuable to treaty commercial coho salmon fishing. The Economic value of treaty commercial coho salmon fishing in the Nisqually River Delta is roughly $5 million with the Refuge contributing at least $49,000 to the total value.

Untangling the endogenous effects of ecological change on salmon populations has proven to be difficult; however, recent studies have devised unique methods to determine the impact of exogenous factors that influence ecosystem change on salmon productivity and salmon habitat capacity (Brennan et al. 2019; Hall et al. 2018). Future valuation research aimed to estimate the impact of climate change on the economic value of treaty commercial salmon fishing in the Nisqually River Delta needs to incorporate novel techniques that quantify the relationship between salmon productivity, peak recruitment, and estuarine habitat mosaics. Although a multitude of factors affect the size of wetland land types over time, it’s imperative to the future economic viability of treaty commercial coho fishing to know the effect of sea level rise on the size of specific wetland land types, such as emergent wetland, scrub-shrub wetland, forested wetland, and aquatic vegetation beds.

APPENDIX 1

Table 17 is extracted from V&J (2020) and shows the results of MRM2. Table 15 and 16 are also extracted from V&J (2020) and provide descriptions of each variable in MRM2.

Table 15: Meta-Analysis Variable Descriptions and Summary Statistics

Table 2 Meta-analysis variables and summary statistics

Variable	Definition	Habitat metadata Mean (SD)	Habitat and area metadata Mean (SD)
ln_wtp	Natural log of willingness to pay (WTP) per household, adjusted to 2016 US dollars. Range in habitat metadata: −2.81 to 2.94	2.74 (2.94)	2.57 (2.81)
ln_absolute_change	Natural log of percentage point habitat commodity change, if measured in absolute terms—change on external 0–100 scale—(zero otherwise). Range in habitat metadata: 0 to 4.61	0.87 (1.43)	0.77 (1.37)
ln_relative_change	Natural log of percentage point habitat commodity change, if measured in relative terms—change as a proportion of baseline—(zero otherwise). Range in habitat metadata: −5.56 to −.70	1.04 (2.38)	1.03 (2.27)
ln_affected_area	Natural log of the size of the resource (or marsh) area affected by change. Unit: acres. Range in habitat metadata: 1.59–16.65	10.37 (4.30)	9.73 (4.50)
ln_income	Natural log of median household income of the US places sampled by the stated preference study (e.g., states, counties, etc.), based on the historical U.S Census data. Where the sample covers multiple US places for which an aggregate median income is not provided by the Census (e.g., multiple counties), population-weighted averages over these places are used. Unit: US dollars. Range in habitat metadata: 9.99–11.15	10.67 (0.24)	10.70 (0.26)
ln_sampled_area	Natural log of the area in which respondents for each study were sampled (the sampled market area). Unit: acres. Range in habitat metadata: 9.39 to 21.56.	16.69 (3.31)	16.10 (3.58)
peer_review	Binary variable indicating that the study is from a peer reviewed source (1=peer reviewed source). Range: 0 or 1	0.83 (0.37)	0.84 (0.36)
annual_wtp	Binary variable indicating that WTP payment would be paid annually, and zero otherwise (1 = annual payment). Range: 0 or 1	0.68 (0.47)	0.72 (0.45)
yearindex	Variable indicating the year in which the survey was conducted (converted to an index by subtracting 1985). Range in habitat metadata: 2–28	18.24 (10.19)	18.71 (9.95)
dichotomous	Binary variable indicating that the type of elicitation method is dichotomous. (1= dichotomous elicitation method; 0= other elicitation types). Range: 0 or 1	0.32 (0.46)	0.28 (0.45)
habitat_fish	Binary variable indicating that the survey scenario addressed fish habitat or services of these habitats (1= fish habitat; 0 = habitat for multiple species, shellfish, bird, wildlife, or endangered species). Range: 0 or 1	0.17 (0.38)	0.15 (0.36)
habitat_multiple	Binary variable indicating that the survey scenario addressed combined fish, shellfish and wildlife habitats or services of these habitats (1= multiple species habitat; 0 = habitat for fish, shellfish, bird, wildlife, or endangered species). Range: 0 or 1	0.57 (0.50)	0.50 (0.50)

Table 16: Meta-Analysis Variable Descriptions and Summary Statistics Continued

Table 2 (continued)

Variable	Definition	Habitat metadata Mean (SD)	Habitat and area metadata Mean (SD)
salt_other_habitat	Binary variable indicating that the type of marsh described in the survey is a combination of salt marsh and other habitat (1 = combined salt marsh and other habitat; 0 = all other coastal marsh types, see Table 1). Range: 0 or 1	0.38 (0.49)	0.33 (0.47)
riparian_marsh	Binary variable indicating that the type of marsh used in the survey is riparian or forested coastal marsh (1 = riparian or forested coastal marsh; 0 = all other coastal marsh types, see Table 1). Range: 0 or 1	0.10 (0.31)	0.12 (0.32)
change_harvest	Binary variable indicating that the valued commodity is a change in harvest or harvest potential (1 = change in harvest; 0 = all other types of change, see main text). Range: 0 or 1	0.22 (0.41)	0.19 (0.39)
change_population	Binary variable indicating that the valued commodity is a change in marsh species population size (1 = change in population size; 0 = all other types of change, see main text). Range: 0 or 1	0.19 (0.40)	0.17 (0.38)
change_survival	Binary variable indicating that the valued commodity is a change in population or survival for threatened or endangered species (1 = change in survival probability; 0 = all other types of change, see main text). Range: 0 or 1	0.13 (0.33)	0.11 (0.32)
change_size	Binary variable identifying observations for which the valued commodity is defined as a raw change in wetland area or size rather than a change in a specified wetland service or commodity. (1 = change in area; 0 = change in specific habitat or habitat service, see main text). Range: 0 or 1	–	0.12 (0.32)

Table 17: MRM2 Results

	Unrestricted model: habitat
ln_absolute_change	−0.189
	(0.117)
ln_relative_change	0.231**
	(0.108)
ln_sampled_area	−0.158*
	(0.083)
ln_income	4.385***
	(1.109)
ln_affected_area	0.138**
	(0.0564)
change_harvest	−1.354***
	(0.339)
change_population	−1.244***
	(0.276)
change_survival	−0.701*
	(0.343)
riparian_marsh	−1.146***
	(0.396)
annual_wtp	−2.284***
	(0.580)
habitat_fish	−0.280
	(0.263)
habitat_multiple	−1.502***
	(0.282)
dichotomous	−0.318*
	(0.177)
peer_review	0.716
	(0.607)
yearindex	−0.124***
	(0.0292)
salt_other_habitat	−0.0733
	(0.280)
intercept	−39.87***
	(11.33)
N	133
R-sq	0.851
RMSE	0.656

$^*p<0.10$; $^{**}p<0.05$; $^{***}p<0.01$

APPENDIX 2

Proximity Analysis

To determine if the size of the study area has an impact on the estimated benefits of the restoration project, I compare the results of three separate regressions that restrict the dataset to properties that are 5, 10, and 20 miles from the Refuge.

Table 18: Proximity Regression Results

	Dependent Varible:			
	log(PRICE18)			
	(<5)	(<10)	(<20)	(All)
DDIKE	1.0E-05***	5.8E-06***	2.5E-06***	2.07E-06***
	(5.5E-07)	(2.6E-07)	(7.7E-08)	(5.53E-08)
DDIKE*PDIKE09	-3.5E-06***	-1.4E-06***	-7.5E-07***	-4.18E-07***
	(5.0E-07)	(1.4E-07)	(4.9E-08)	(3.3E-08)
FORECLOSURE	0.017***	-4.7E-04	-7.2E-03***	-5.46E-03***
	(0.0031)	(0.001)	(2.9E-04)	(2.47E-04)
Structural	Yes	Yes	Yes	Yes
Neighborhood	Yes	Yes	Yes	Yes
Environmental	Yes	Yes	Yes	Yes
Submarket FE	Yes	Yes	Yes	Yes
Time FE	Yes	Yes	Yes	Yes
Constant	11.92***	11.18***	12.23***	12.08***
	(0.34)	(0.17)	(0.0067)	(0.06)
Observations	9,003	24,715	62,763	78,928
R^2	0.78	0.76	0.78	0.79
Note:		*p<0.1; **p<0.05; ***p<0.01		

The direction of the coefficient on DDIKE and DDIKE*PDIKE09 are robust in the proximity of property sales to the Refuge. Properties within five miles of the Refuge experience the largest disservices before and after the removal of the dike. Also, FORECLOSURE is statistically significant and positive. This result suggests that properties near foreclosure sales are being sold at a premium, possibly due to positive expectations about future property prices. The coefficient on DDIKE with the inclusion of property sales within ten miles of the Refuge is smaller in magnitude than the coefficient on DDIKE with the inclusion of sales within five miles. The difference implies that homes within five to ten miles of the Refuge experience less disservices than homes within five miles. Additionally, the difference in magnitude of the coefficient on DDIKE*PDIKE09 implies that dike removal improved housing values of homes from five to ten miles from the Refuge more than housing values of homes that are less than five miles from the Refuge.

Table 19: Proximity Regression MIP

< 5 miles

Coefficent	Mean Sale Price	MIP	Post MIP	Change
1.0E-05	333,307	3.33	3.22	-4%
-3.5E-07		-0.12		

< 10 miles

Coefficent	Mean Sale Price	MIP	Post MIP	Change
5.8E-06	325,477	1.89	1.43	-24%
-1.4E-06		-0.46		

< 20 miles

Coefficent	Mean Sale Price	MIP	Post MIP	Change
2.5E-06	327,115	0.82	0.57	-30%
-7.5E-07		-0.25		

All Data

Coefficent	Mean Sale Price	MIP	Post MIP	Change
2.1E-06	335,444	0.69	0.55	-20%
-4.2E-07		-0.14		

As shown in Table 18, the relationship between log price and distance to the dike is nonlinear. Houses within twenty miles of the Refuge experienced the largest percentage change in value attributed to the dike removal. Table 19 displays the MIP calculations for all proximity regressions. The pre- and post-MIP falls in distance to the Refuge. In other words, the disservices provided by the Refuge to nearby homeowners are largest for homes in a relatively close proximity to the Refuge; however, homes that are in a close proximity to the Refuge also benefit the most from the 2009 Brown Farm dike removal.

Regression Results Table

Table 20: All Regression Results 1

	Dependent Varible:			
	log(PRICE18)			
	(1)	(2)	(3)	(4)
YELMTHURSTON	0.076**	-0.054	-0.079	-8.83E-04
	(0.036)	(0.036)	(0.02)	(0.019)
PUYALLUP	-0.12***	-0.29***	0.046***	-0.28***
	(0.0071)	(0.0088)	(0.0041)	(0.0066)
CARBONADO	-0.29***	-0.66***	-0.12***	-0.33***
	(0.05)	(0.05)	(0.028)	0.027)
UNIVERSITYPLACE	0.12***	0.04***	0.18***	-0.13***
	(0.012)	(0.012)	(0.0066)	(0.0075)
SUMNER	0.056***	-0.22***	0.079***	-0.31***
	(0.0084)	(0.012)	(0.0048)	(0.0082)
DIERINGER	0.32***	0.03*	0.22***	-0.30***
	(0.013)	(0.016)	(0.0071)	(0.01)
ORTING	-0.11***	-0.37***	-0.099***	-0.33***
	(0.01)	(0.013)	(0.0059)	(0.0084)
CLOVERPARK	-0.18***	-0.23***	-0.015**	-0.22***
	(0.01)	(0.011)	(0.006)	(0.0065)
PENINSULA	0.28***	0.12***	0.22***	-0.28***
	(0.0086)	(0.0099)	(0.0049)	(0.0076)
FRANKLINPIERCE	-0.27***	-0.4***	-0.086***	-0.35***
	(0.01)	(0.011)	(0.0059)	(0.007)
BETHEL	-0.16***	-0.31***	-0.1***	-0.28***
	(0.0074)	(0.0087)	(0.0042)	(0.0059)
EATONVILLE	-0.21***	-0.49***	-0.15***	-0.082***
	(0.016)	(0.018)	(0.0091)	(0.011)
AUBURN	0.064***	-0.23***	0.16***	-0.31***
	(0.012)	(0.015)	(0.0067)	(0.01)
WHITERIVER	0.01	-0.34***	0.063***	-0.32***
	(0.011)	(0.015)	(0.0059)	(0.0097)
FIFE	0.014	-0.19***	0.07***	-0.37***
	(0.012)	(0.014)	(0.0069)	(0.0091)
CAPITAL	-0.0029	-0.079***	0.062***	0.16***
	(0.0087)	(0.0089)	(0.0049)	(0.0078)
NORTHTHURSTON	-0.14***	-0.13***	-0.022***	0.075***
	(0.0088)	(0.0087)	(0.005)	(0.008)

Table 21: All Regression Results 2

OLYMPIA	-0.061***	-0.089***	0.078***	0.2***
	(0.0084)	(0.0084)	(0.0048)	(0.0078)
RIVERRIDGE	-0.08***	-0.028***	0.019***	0.096***
	(0.0079)	(0.008)	(0.0045)	(0.0082)
BLACKHILLS	-0.091***	-0.17***	0.011**	0.17***
	(0.0091)	(0.0093)	(0.0051)	(0.0079)
TIMBERLINE	-0.17***	-0.18***	-0.045***	0.085***
	(0.0082)	(0.0081)	(0.0046)	(0.0078)
YELM	-0.25***	-0.36***	-0.16***	-0.073***
	(0.0083)	(0.0088)	(0.0047)	(0.0077)
TUMWATER	-0.028***	-0.11***	-0.025***	0.14***
	(0.0096)	(0.0098)	(0.0054)	(0.008)
TENINO	-0.26***	-0.38***	-0.18***	0.089***
	(0.012)	(0.013)	(0.0071)	(0.009)
ROCHESTERHIGH	-0.23***	-0.49***	-0.17***	0.18***
	(0.011)	(0.014)	(0.0065)	(0.0091)
YR2005	0.13***	0.057***	0.16***	0.1***
	(0.0046)	(0.0065)	(0.0026)	(0.0048)
YR2006	0.23***	0.16***	0.25***	0.18***
	(0.0047)	(0.0065)	(0.0027)	(0.0071)
YR2007	0.24***	0.18***	0.26***	0.19***
	(0.005)	(0.0068)	(0.0028)	(0.0072)
YR2008	0.14***	0.076***	0.17***	0.11***
	(0.0053)	(0.007)	(0.003)	(0.0068)
YR2009	0.036***	0.033***	0.071***	0.053***
	(0.0057)	(0.0057)	(0.0032)	(0.0052)
YR2010	-0.022***	-0.023***	0.0053	-0.014***
	(0.0059)	(0.0059)	(0.0033)	(0.0037)
YR2011	-0.095***	-0.098***	-0.11***	-0.11**
	(0.0062)	(0.0061)	(0.0035)	(0.0034)
YR2012	-0.12***	-0.12***	-0.15***	-0.13***
	(0.0058)	(0.0058)	(0.0033)	(0.0034)
YR2013	-0.074***	-0.079***	-0.1***	-0.08***
	(0.0053)	(0.0052)	(0.003)	(0.003)
YR2014	-0.053***	-0.058***	-0.072***	-0.065***
	(0.0053)	(0.0053)	(0.003)	(0.0034)
DDIKE		3.18E-06***		2.07E-06***
		(9.07E-08)		(5.53E-08)
DDIKE*PDIKE09		-8.4E-07***		-4.18E-07***
		(6.07E-08)		(3.3E-08)

Table 22: All Regression Results 3

DSEATTLE				-4.34E-06***
				(4.87E-08)
INCOME18				5.6E-06***
				(9.6E-07)
ACRES			0.072***	0.075***
			(9.09E-04)	(8.78E-04)
$ACRES^2$			-0.0019***	-0.002***
			(5.31E-05)	(5.01E-05)
SQFT			5.18E-04***	5.12E-04***
			(4.57E-06)	(4.28E-06)
$SQFT^2$			-3.4E-08***	-3.49E-08***
			(9.07E-10)	(8.5E-10)
AGE			-1.14E-05***	-4.17E-04***
			(3.47E-05)	(3.38E-05)
BEDROOM			-0.03***	-0.025**
			(9.38E-04)	(8.81E-04)
BATHROOM			0.1***	0.096***
			(0.0015)	(0.0014)
STORIES			-0.14***	-0.14***
			(0.0016)	(0.0015)
DWATER				-8.72E-06***
				(3.81E-07)
DROAD				1.31E-05***
				(3.37E-07)
DTRAIL				1.12E-07
				(7.87E-08)
DPARK				3.37E-06***
				(2.24E-07)
THURSTON				0.023**
				(0.011)
FORECLOSURE				-0.0055***
				(2.47E-04)
Constant	12.65***	12.6***	11.78***	12.08***
	(0.0076)	(0.0084)	(0.0067)	(0.06)
Observations	78,928	78,929	78,930	78,931
R^2	0.23	0.24	0.76	0.79
Note:			*p<0.1; **p<0.05; ***p<0.01	

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BIOGRAPHY

Anthony Good graduated from Pleasant High School, Marion, Ohio, in 2011. He received his Bachelor of Science from Wright State University in 2015 and a Master of Arts from George Mason University in 2017.