LINKING ECOLOGICAL FUNCTION AND ECOSYSTEM SERVICE VALUES OF ESTUARINE

HABITAT TYPES ASSOCIATED WITH A BARRIER ISLAND SYSTEM

A Dissertation

by

JEFFREY MICHAEL FRANCIS

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ABSTRACT

Linking ecological function and ecosystem service values of estuarine

habitats associated with a barrier island system

(December, 2012)

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Ecosystem services are benefits humans receive as a function of natural processes. Many current studies seek to express these benefits as an economic value per unit of habitat type without quantifying the ecological functions that allow for the provision of ecosystem services. This study is designed to model each habitat type in an effort to explicitly link the major estuarine habitat types of Mustang Island (oyster reefs, seagrass meadows, and intertidal salt marsh) to their contribution to Nitrogen cycling services. First, a dynamic biomass model of each foundational species was created using Simile, a declarative modeling framework. Second, a monthly snapshot of Nitrogen captured in living biomass was used to quantify the contribution of each species to the Nitrogen cycling services. Finally, the amount of Nitrogen captured in living biomass was valued using a replacement cost approach. An effort was also made to link the provision of recreational fishing services provided by each aforementioned habitat type by partitioning travel costs and license sales weighted by the density of fish found in each habitat type.

It was found that oyster reefs of Mustang Island contribute $173,000 yr^{-1}, seagrass meadows contribute $12,054,095 yr^{-1}, and intertidal salt marshes contribute $5,242,755 yr^{-1} in potential Nitrogen cycling services. The total value of recreational fishing services within the study site was calculated to be $83.8 million dollars yr^{-1}. A portion of the total value was then attributed to each habitat type: Marsh edge: $2 million; Seagrass meadows: $81 million; and Oyster: $81,000 thousand.

These efforts have been made to translate ecological function into economic benefit to improve communication among a wide variety of stakeholders that are more likely to understand economic value. Further refinement of both the models and the economic data necessary to support them, will have the potential to improve the applicability and results of these tools. These results, and the modeling framework through which they are calculated, provide a platform to evaluate management relevant scenarios in a simple, flexible manner that may be adjusted and transferred to other study sites given appropriate local data.

Acknowledgements:

It is only with the support of family, friends, and colleagues that I was able to accomplish this achievement. First, I would like to thank my advisors, Joe Fox and David Yoskowitz. Joe has long been a mentor to me in the pursuit of higher education. He helped guide me through my Master's work and was the impetus that brought me to the Harte Research Institute as a doctoral student. His constructive criticism has helped me develop the skills required to persevere through two post-graduate degrees. Dave has also been instrumental in helping me develop the skills necessary to be successful in this academic pursuit. He helped guide me as I began a new chapter in my study of Marine Biology by carefully adding the long-needed social component necessary to more effectively communicate the love I have for the marine environment to those around me. Both of you have been greatly influential and appreciated throughout this process, both on a professional and personal level. I appreciate your willingness to share your knowledge, question my thoughts, and challenge me to delve more deeply into my studies and this process. Beyond your professional guidance, I feel that I have gained not only mentors, but friends. I would be remiss if I did not also thank the rest of my advisory committee: Drs. Jim Gibeaut and Greg Stunz. Each of you has been generous with your limited time, but also with your depth of knowledge. I appreciate your ability to help steer me through the pitfalls of a doctoral degree and helping add the tools I have needed to be successful. I have also received plenty of guidance from Drs. Paul Montagna, Jennifer Pollack, David McKee, Donald Thomson, Larry McKinney, Rich McLaughlin, Thom DeWitt, and Robert Vega for which I am very grateful.

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I have long been an odd bird. Not many people spend their life in the desert with their heart in the sea. I could not have made it through without each of you. Thank you.

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Chapter I:

Introduction and Background

Introduction:

Historically, highly productive estuarine habitats were seen as a source of food, transportation, and recreation for those that live near them. But, they have also been seen as inputs into economic production of goods and services with little attention paid to the impact that these exploitive and extractive processes may have on the health of the ecosystems that comprise these systems. This study seeks to make connections between the functions of specific estuarine habitats and some of the benefits humans receive as a result; both as a method of improved communication about our estuarine resources and how best to use them effectively while still maintaining their functional integrity.

Ecosystem services are defined as the contributions from the environment that benefit human well-being (Costanza et al., 1997; MEA, 2005; NRC, 2005). Identifying these benefits has often involved merging both economic and natural sciences in an effort to communicate their value. In the past, this has been accomplished by quantifying natural inputs harvested for human benefit such as fisheries, timber production, and mining. Traditionally, the health of associated ecosystems has been externalized from the economic valuation process and seen simply as an input needed to provide economic goods and services (Polasky and Segerson, 2009). Costanza et al. (1997) promoted the importance of non-market services such as climate regulation, disturbance regulation, nutrient cycling, and aesthetic value. The ultimate goal of

merging ecological knowledge and economic practice is to communicate the value of

ecosystem services and begin to provide a basis for the policy making process that

ultimately determines how society will confront growing environmental challenges

(Chee, 2004; EPA, 2008; MEA, 2005; Polasky and Segerson, 2009).

Much of the work involved in valuing ecosystem services is highly

interdisciplinary and often involves biologists, ecologists, economists, sociologists and

many others from both the natural and social sciences. Farber et al. (2006) delineated

each service into four distinct categories: supportive, regulating, provisioning, and

cultural. Each category is then further defined to clarify specific ecosystem functions

and the goods or services they provide. By providing a common language, the

collaboration among disciplines is streamlined and potentially more effective (NRC,

2005).

Economic valuation of natural resources is not a recent invention; however,

current protocols have largely derived from the necessities presented by the Exxon

Valdez disaster in 1989. Litigation between Exxon, Inc., and the people of Alaska

focused not only on the damage directly caused by the oil spill, but also included losses

associated with disruption of ecosystem processes and resultant economic loss (Carson

et al., 2003). This was unique in that, at that time, few precedents had been established

allowing for valuation of an ecosystem by its' passive attributes (Carson et al., 2003;

Duffield, 1997). In other words, this case was the first to address economic values with

ecosystem services beyond the traditional view of marketable goods. The State of

Alaska ultimately awarded area commercial fishermen $269 million based upon the

decreased market value of the degraded fishery. The Alaskan native corporations

settled out of court for $20 million citing degradation and cultural loss associated with

the spill (Duffield, 1997). This launched an ensuing debate among economists as to the

appropriate value of these resources (Duffield, 1997; Carson et al., 2003).

     The work of Costanza et al. (1997) laid the foundation for current ecosystem

service studies, in that it evaluated many of the goods and services not represented by

traditional economic markets and was designed to exceed that of a legal system focused

exclusively on determination and measurement of liability. It was implied that these

values could be used in the policy making process as a decision and management tool

providing non-market resources equal footing with economic endeavors (Chee, 2004).

The global scale of the study, in which 17 ecosystem services were derived from 16

global ecosystems, was also considered too cumbersome as a functional management

tool (Farber et al., 2002, 2006; NRC, 2005). Over time, very little credence has been

given to the actual values produced by this study, an average of $33 trillion/year. One

criticism is that the amount determined by Costanza et al. (1997) substantially exceeded

the global gross domestic product (GDP) at the time ($18 trillion/year). Despite these

criticisms, the Costanza et al. (1997) paper ultimately spawned the current study of

ecosystem services.

     The vast majority of current ecosystem service studies focus on the economic

benefit and valuation of terrestrial ecosystems (e.g., forests and agricultural production)

(Guo et al., 2001; Kenyon and Nevin, 2001; Torras, 2000; Xue and Tisdell, 2001).

Guo et al. (2001) estimated the economic benefits derived from the forest ecosystem in

Xingshang County, China, to be $6.53 billion, in terms of forest products and tourism.

This value increased ten-fold with the additional consideration of forest related

ecosystem services such as soil and water conservation as well as gas regulation. These

estimates have strong significance for China, whose forest ecosystems comprise 51% of

its surface area. In contrast, little direct economic benefit is provided to inhabitants in

these areas. Guo et al. (2001) used prevention of silt accretion as a basis for evaluation,

comparing control of erosion by the forest ecosystem to cost of labor to remove the silt

from a reservoir thereby maintaining water storage capacity.   According to this study,

the volume of silt accretion prevented by the intact forest ecosystem was 16.6 million

m^3 and the price of removal was calculated to be $0.57 m^{-3} equaling an economic

benefit of $9.46 million in indirect economic savings.

The imbalance of ecosystem service studies that exists between those focused

on terrestrial versus marine habitats can be attributed to different factors. Apart from

commercial and recreational fisheries, very few markets exist that incorporate services

provided by marine habitats. Often, quantifiable and direct relationships between

ecological function and ecosystem services have not been established. Additionally,

studies concerning ecosystem services of marine habitats are generally more difficult to

undertake than those of their terrestrial counterparts (NRC, 2005; Polasky and

Segerson, 2009). The proposed study will serve as one of the first to be conducted on a

barrier island system (Mustang Island) and its varied habitats.

Mustang Island is a high profile barrier island that separates the Corpus Christi

Bay system from the Gulf of Mexico. It is located south of San Jose Island and north of

North Padre Island (Fig. 1.1) (Simms et al., 2006). The Corpus Christi Bay system is large

(e.g., 43,288 ha at mean low water) and includes Redfish, Nueces, Corpus Christi, and

Oso Bays (NOAA, 1998). The surrounding marine and intertidal habitats - oyster flat,

seagrass meadows, intertidal salt marsh, unvegetated bottom, and near shore Gulf

waters - of Mustang Island support an estimated 600 species of saltwater fish and

invertebrates (TPWD, 2009). The northernmost point of the island is bound by dredged

Aransas Pass and Corpus Christi Ship Channels. The southern portion of the island

contains several storm overflow channels as well as Packery Channel (also dredged),

which allows small craft access to the Gulf of Mexico (White et al., 2006). Mustang

Island is a unique area for the study of ecosystem services in that it contains highly

productive habitats within close proximity to human and economic activities such as

commercial shipping, residential housing, and recreational fishing while still remaining

largely undeveloped. The estuarine ecosystems of this area also face three fundamental

changes that may affect the biogeographical make-up of the island: relative sea level

rise, decreased freshwater inflow, and an ongoing shift in salt marshes once dominated

by *Spartina* grasses to those dominated by Black Mangroves (*Avecinia nigricans*). These

changes, coupled with human proximity, provide an opportunity to evaluate the

relationship between the ecosystem services each habitat provides and the value placed

on them by the surrounding human populations before further change and

development proceed.
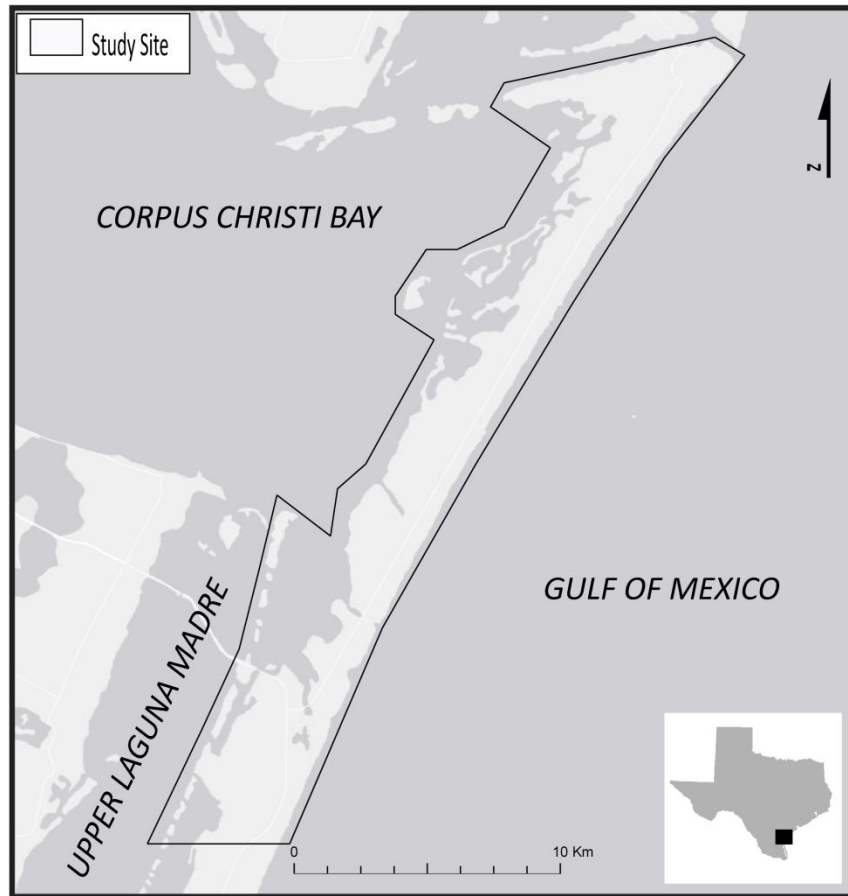
Figure 1.1: Map of Study site (Mustang Island, Nueces County, TX, USA: 27.4 N 97.8 W)

Habitats Types:

Oyster reefs within the Mustang Island system cover approximately 35 ha of bottom surface area, a relatively small area. Currently, this system does not support a commercial oyster fishery; however, the ecological function of oysters is of such importance that it cannot be overlooked (White et al., 2006). Structural complexity of the oyster flats in this area creates refuge habitat for a large variety of organisms (Coen and Luckenbach, 2000; Posey et al., 1999), estimated by Texas Parks and Wildlife Department (TPWD, 2007) to comprise approximately 300 species. Oysters are also

well-known for their ability to filter large amounts of water. Hence, when present, they play a unique role in cycling nutrients, maintaining water quality and even influencing phytoplankton dynamics (Coen and Luckenbach, 2000; Dame et al., 1984; Zeug et al., 2007).

Humans receive several benefits from the ecological functions performed by oysters, namely waste and water regulation, food, raw materials, and recreation (Santopietro, 1998; Henderson and O'Neil, 2003; Hicks, 2004). Although no economic valuation of this habitat appears to exist for the study area, Yoskowitz et al. (*in review*) estimates that the economic value of oyster reefs and the service of nitrogen removal in the Mission-Aransas Bay system (adjacent to the current study site) via assimilation into shell and tissue is $1,196,759 yr^{-1}. The inclusion of other services provided by the oyster reefs would obviously increase their benefits.

Seagrass meadows in the Mustang Island system are prominent, covering approximately 4,058 ha, which represents a significant increase over the last 50 years (White et al., 2006). This increase in coverage has been attributed to relative sea level rise and flooding of low-lying tidal flats. It was estimated that at total gain of 548 ha was seen from the 1950's to 2002-04 (White et al., 2006). These areas are highly productive and play a unique role as essential habitat for estuarine-dependent species. Much of their productivity is available to herbivores in the form of shoots, roots, and rhizomes. This high level of productivity also allows them to metabolically fix large amounts of carbon and, simultaneously, provide dissolved oxygen to benthic communities. Seagrass meadows can effectively filter and fix nutrients from the water

column (Briones, 2004; Harborne et al., 2006; Onuf and Inguld, 2007; Reese et al., 2008; James et al., 2009). Commercially important species such as red drum (*Sciaenops ocellatus*), brown shrimp (*Farfantepenaues aztecus*), and the southern flounder (*Paralichthys lethostigma*) depend on seagrass at some point in their lifecycle (Reese et al., 2008; James et al., 2009).

The ecosystem services provided by seagrasses include habitat and niche availability, nutrient cycling, soil retention, water and waste regulation, recreation, and education. Several studies have attempted to value the ecosystem services that seagrasses provide. McArthur and Boland (2006) estimate that seagrass habitat of the Southern Australian coast provide an economic impact of AUS$ 114 million. This estimate is derived from the secondary production value (harvests) of those fishery products dependent upon this habitat. Using a Geographic Information System that helped define degraded areas, it was determined that a 16% loss of seagrass habitat in this area could represent an economic loss of AUS$ 235,000/year. Samonte-Tan et al. (2007) estimated that seagrass habitat in the Bohol Marine Triangle of the Philippine islands provided $106,000 derived from tourism and resources. The study site covered an area of 112,000 ha while the seagrass habitat covered approximately 2,555 ha.

Intertidal salt marsh covers approximately 925 ha of the Mustang Island system. In light of a relative sea level rise of 1.7cm/year since the 1950's, this habitat has remained relatively stable with respect to coverage area, increasing in coverage only 6% (White et al., 2006). This habitat is unique in that it is not fully submerged throughout the tidal cycle, but performs many of the same ecological functions as the surrounding

submerged seagrass meadows. Intertidal salt marsh is subjected to wide variation in environmental conditions ranging from high salinity and desiccation to complete inundation by freshwater. Studies have shown that this habitat is essential for many of the commercial and recreational fisheries throughout the various estuary systems of Texas. Many species of fish as well as decapod crabs depend on this habitat as a nursery ground and food supply during various life stages (Kimball and Able, 2007; Zueg et al., 2007; Kunza and Penning, 2008; Nanez-James et al., 2009). Nitrogen uptake and waste regulation is also an important ecological function performed by this habitat (Hopkinson and Giblin, 2008).

The ecosystem services provided by intertidal salt marshes include waste and water regulation, habitat for commercially important species, net primary productivity, disturbance regulation, and recreation (Bergstrom, 1990; Bell, 1997; Kazmierczak, 2001). Recognized as an important resource for the Gulf of Mexico, Bergstrom (1990) estimated the economic impact of coastal salt marsh to be $6,471/acre/year along the west coast of Florida alone. This rate was determined using 1984 prices and only considers the contribution of coastal salt marsh to recreational fishing. Bell (1997) estimated the economic value of coastal salt marsh in Louisiana as a function of recreational use to be $145 million yr^{-1} statewide. In a separate study, Kazmierczak (2001) evaluated the same habitat along the Louisiana coast, and determined the improvement of water quality, to have a value of $323/acre/year.

Placing an economic value on each ecosystem service is not always applicable or possible. Often when an economic value does not adequately address the importance

of something, it is referred to as intrinsic value (Lockwood, 1997). This concept

recognizes values not tied to economic utility, rather inherent value attributed to a

habitat, ecosystem, or species. An example of this is efforts in the U.S. to protect

endangered species such as the bald eagle. Although it is likely that only cultural and

aesthetic ecosystem services are actually provided by this animal, substantial effort has

been undertaken to protect it. This concern derives from its symbolic relevance to the

U.S. and other anthropomorphic and sentimental reasons. Similarly, the giant panda of

China has also been protected despite its minor utility to humans. These cases illustrate

that value cannot be completely communicated using only monetary metrics

(Lockwood, 1997; Attfield, 1998; Justus et al., 2008). It is difficult to communicate these

values since they are not based on a common "currency", but an effort is made in this

research to communicate value of each habitat of Mustang Island monetarily, while it is

still important to recognize the nonmonetary importance of this barrier island.

Each habitat type contributes to the nitrogen cycle in some way. Three main

processes govern the movement of nitrogen in an estuary: input, flux, removal

(Seitzinger, 1988; Vitousek et al., 1997; Tobias, 2009). Each of these processes is highly

variable by temperature, light regimes, sediment chemistry, water column chemistry,

density of foundational species, as well as a host of other stochastic variables.

Ultimately, these define the level of ecosystem service provision provided by a given

habitat (Cloern, 2007; Eyre and Maher, 2010; Barbier et al., 2011). The data

requirements for defining these three processes at a meaningful level are demanding.

Therefore, this project seeks to simplify the contributions of each habitat to the

nitrogen cycle by capturing nitrogen bound in the living biomass in an effort to explain as a "snapshot" the provision of ecosystem services.

Research Questions and Objectives:

The Research questions this study seeks to address are the following:

1. Is it possible to create a model that captures the ecological function of the major estuarine habitats (oyster reef, seagrass meadow, intertidal salt marsh) and explicitly link that function to the provision of nitrogen cycling services?

2. Can this "snapshot" of nitrogen cycling services be transferred to similar systems given the appropriate spatial and temporal data?

3. Can this "snapshot" of nitrogen cycling services be used to ask management-relevant questions with alternate outcomes that can be evaluated in side-by-side comparisons?

4. Can a model also be constructed that explicitly links the provision of recreational fishing services based on the productive capacity of each major estuarine habitat?

The objectives of this study are:

1. Model the ecological function and contribution to nitrogen cycling for oyster flats, seagrass meadows, and intertidal salt marshes of the Mustang Island system;

2. Explicitly link ecological function to ecosystem service provision to create a means of valuation of those services;

3. Use replacement cost methodology as a means of valuing nitrogen cycling services provided by these habitats; and

4. Define the contribution of the major marine habitats of this study in terms of recreational fishing services and their value as a function of travel cost and license sales.

Approach:

Conceptual framework and example:

The present study follows a general conceptual framework similar to other ecosystem service studies (Fig. 1.2) with a particular focus on improving the relationship between ecological metrics and economic data. To properly assess the value of an ecosystem service, it is necessary to link a specific habitat to an ecological function capable of quantification. Once a direct connection between habitat and function has been made, it is coupled with socioeconomic data to provide an estimate of value represented by habitat (NRC, 2005; Turner et al., 2008).

This process involves four basic steps that are replicated for each habitat. First, an aerial delineation of the habitat was determined, either by direct mapping or using the best available data. The oyster complex within the study site required direct mapping, while the areal extent of both the seagrass and intertidal salt marsh are well-defined (Cowardin et al., 1979; White et al., 2006; Gibeaut et al., 2010). Second, a population model based on biomass of the foundational species for each habitat is created in a declarative modeling platform Simile. This preliminary step quantitatively defines the ecological function of each habitat based on total biomass of the

foundational species within the defined area. Third, functional linkages between

biomass and nitrogen cycling contributions are made to quantitatively express the level

of service provision of each habitat. Finally, locally relevant economic data was used to

place a value on each habitat expressed as the value of service provided per unit area.

This framework creates a direct connection among habitat, ecological function, service

provision, and value.
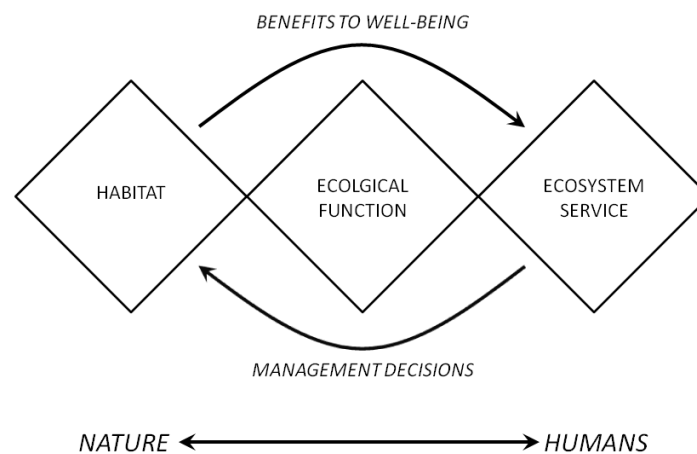
Figure 1.2: Conceptual framework for ecosystem service studies

The development of each step from ecological function to economic value

involves fitting current biological data into a framework that can be economically

evaluated. Recreational fishing is somewhat different in that the valuation must include

the density and abundance of fish within the study site coupled with the areal extent of

each habitat. A biomass based model of the pertinent recreational fisheries is not

necessary. This will provide a picture of the abundance of fish associated with each habitat and which will ultimately be considered equivalent to the productivity of a given habitat within the Mustang Barrier Island system. The productivity of each habitat will then be linked to studies that have evaluated the travel cost expenditures for related habitats along the Texas coast and elsewhere (Barbier et al, 1997 and Johnson et al, 2002, Southwick Associates, 2006).

Standardized methods do not currently exist for valuing ecosystem services. Most studies are based on some type of economic indicator that requires information collected from a subset of the population of interest (Farber et al., 2006). Therefore, the overarching goal of this project is to create logical, relevant connections among information regarding ecological function, ecosystem use, and the value of the goods and services provided. This process is highly individual with respect to the habitat that is being valued. Intertidal salt marsh is a good example. Previous studies have calculated the recreational value of the salt marsh habitat in Louisiana to be approximately $32.40/ha/year (Barbier et al., 1997), yet the high amount of freshwater inflow, different make up of recreational fisheries, and differences in the local economy surrounding the area may have a significant impact on the actual value as it is calculated. These differences require that a localized approach be used to help adjust values as a function of these differences. In the case of recreation in the intertidal salt marsh, it is important that bird habitat and the associated tourism industry be included when valuing these systems along the Texas coast. Bowker and Stoll (1988) measured the benefits of the existence of the Whooping Crane as a willingness-to-pay per

respondent to be from $21 to $70. The addition of this type of local information along

the Texas coast will most likely increase the recreation value associated with the

intertidal salt marsh. This is just one example of the differences necessary to be

understood in order to make a realistic valuation.

Economic methods:

Acceptable methods of economic evaluation used in ecosystem services studies

were described by Farber et al. (2006). The first method employs a <u>revealed-preference</u>

<u>approach</u>. This approach includes travel costs (i.e., costs incurred to travel to a

particular location), market methods (e.g., price of timber harvested), hedonic methods

(i.e., willingness to pay as a function of a related market), and production approaches

(e.g., increased shrimp harvested near increased wetland areas). The second category

of valuation techniques is the <u>stated-preference approach</u>. Stated-preferences are

measured by contingent valuation (i.e., subjects are surveyed in terms of their

willingness to pay for a service such as clean air) and conjoint analysis (i.e., people are

asked to rank specific actions and outcomes in an effort to express their preference).

Finally, <u>cost-based approaches</u> look at replacement cost (e.g., the cost of cleaning water

without the help of natural processes) and avoidance cost (e.g., medical costs avoided

by clean water being provided naturally). The specific method used to value ecosystem

services of each habitat will vary depending on the types of economic data available as

well as familiarity with the ecological service provided.

Overall, the goal of this study is to create a replicable process that clearly

defines the ecological and economic data required to establish local valuations of

ecosystem services among the various coastal communities bordering the Gulf of

Mexico. To accomplish this, it was necessary to create an ecological model of each

habitat that captures the contribution each has to the nitrogen cycle. Once the

ecological function of each habitat is defined, this information will be used to determine

the economic value of nitrogen cycling services provided by each habitat. Strengthening

this relationship allows for improved information for management-relevant decisions as

well as the creation of a useful representation of the valuation process for subsequent

transfer to other systems. Ultimately, the goal will be to create a process for creating

metrics and values that can be used by natural resource managers when making

decisions affecting the ability of each habitat to provide ecosystem services.

Dissertation organization:

This dissertation is organized into six chapters. Chapter I presents an

introduction, research questions and objectives, as well as the conceptual approach.

Chapters II, III, and IV will describe each habitat and the associated model used to

calculate nitrogen cycling contributions. Chapter V will describe the recreational fishing

values attributed to each habitat. Chapters II, III, IV, and V are presented as individual,

stand-alone pieces of work that may be submitted for publication. Therefore, each

contains individual introductions and conclusions. Chapter VI is a brief summary and

conclusions drawn from the creation of the entire dissertation. References for all

chapters can be found at the end of Chapter VI.

Chapter II:

Modeling nutrient regulation services and biomass of seagrasses

in response to surface irradiance.

Abstract:

Seagrass ecosystems are highly productive and provide a multitude of ecosystem services including habitat for commercially important fisheries, protection from soil erosion, and nutrient regulation. Using surface irradiance as the main input, a predictive ecosystem service model was created to value the nutrient regulation service provided by seagrass. The baseline model was initiated under the assumption that 50% of the surface irradiance would reach the seagrass canopy. Under these conditions the model calculated a mean biomass of 125 gdwt m^{-2}. Coupling nitrogen acquisition and biomass calculations with the cost to construct a wastewater treatment facility designed to remove nitrogenous waste, a mean nitrogen acquisition value of \$1,004,507 mo^{-1} was calculated. Sensitivity analyses showed that model output is driven by changes in surface irradiance and nitrogen acquisition rate. In general, biomass and nitrogen acquisition values varied directly with surface irradiance and nitrogen acquisition rate inputs. Ultimately, the model was able to translate ecological function into broadly understood economic terms in an effort to inform the management process across a wide range of stakeholders.

Introduction:

Seagrass ecosystems are recognized as being both highly productive and important in the provision of ecosystem services and human well-being (Short and Wyllie-Echeverria, 1996; Costanza, 1997; Burkholder et al, 2007; Duarte, 2000) while simultaneously being degraded, mostly by human activity (Duarte, 2000; Fourqurean et al, 2003; Heck et al., 2003; Erftemeijer et al., 2006; Greening and Janicki, 2006; Orth et al., 2006; Russell et al., 2011). Seagrass communities contribute to both recreational and commercial fisheries in the form of a predation refuge for many economically important species (Beck et al., 2001; Sheridan and Minello, 2003; Stunz, 2010) protect coastlines from soil erosion (Duarte, 2000; Duarte, 2002; Newell and Koch, 2004), decrease suspended particles in the surrounding water (Newell and Koch, 2004; Erftemeijer and Lewis, 2006), and play an important role in nutrient regulation both in the water column and the sediment (Duarte, 1990; Hemminga et al., 1991; Herbert, 1999; Hemminga and Duarte, 2000; Welsh, 2000; Romero et al., 2006). Defining the contribution to human well-being by seagrass communities in quantitative terms, may help to focus management and policy aimed at preserving seagrass habitat and the services it provides.

Light is the driving force for photosynthesis and changes in light regimes have strong implications on the density, areal extent, and health of seagrass communities (Dunton, 1994; Kaldy and Dunton, 2000; Burd and Dunton, 2001). In Texas, the loss of approximately 140km^2 of seagrass in the lower Laguna Madre has been attributed to decreased light at depth from increased turbidity resulting from maintenance dredging

of navigation channels (Quammen and Onuf, 1993; Onuf, 1994). Large inputs of

nutrients are also able to decrease light at depth by allowing phytoplankton and

epiphytes, often able to take up nutrients more quickly than seagrasses, to bloom and

diminish the light available to the seagrass canopy (Valiella et al., 1992; Dennison et al.,

1993; Tomasko et al., 1996; Cloern, 2001; Cardoso et al., 2004; Paling et al., 2009).

Regardless of the root cause, light availability is important in maintaining the health of

seagrass communities.

The role of seagrass as part of the nitrogen cycle within the estuary is important

in that a large amount of nitrogen is able to be taken up by the plants, yet much of it is

recycled within each individual or lost to the sediment through senescence with little

denitrification taking place (Dunton, 1996; Lee and Dunton, 1999; Kaldy and Dunton,

2000; Burd and Dunton, 2001). Seagrasses are able to acquire nitrogen through both

shoots and roots/rhizomes making both water column and sediment nitrogenous

species available for uptake. This also creates a microhabitat resulting in bacterial

nitrogen processing within the rhizosphere (Herbert, 1999; Welsh, 2000). Therefore, it

can be difficult to delineate the overall nutrient dynamics of the seagrass ecosystem

(Lee and Dunton, 1999; Evrard et al., 2005; Lee et al., 2007).

The goal of this project is to build a model capable of predicting shifts in

biomass, quantify the value of the nitrogen acquired by the seagrass complex, and

communicate the importance of seagrasses in broadly-understood, economic terms.

The model seeks to quantify the ecosystem service value associated with nitrogen

uptake as a method of communicating the relative importance of the non-market

nitrogen acquisition value of seagrasses within the current study site. It is also built in a manner that would allow others to populate the model with local data in order to gain insight into the management of their own sites of interest.

Methods:

Study site:

Mustang Island is a high profile barrier island (Fig. 2.1) that separates the Corpus Christi Bay system from the Gulf of Mexico (Simms et al., 2006). The Corpus Christi Bay system is large (497 km^2 at mean low water) and includes Redfish, Nueces, Corpus Christi, and Oso Bays . The average depth of the Corpus Christi Bay system is 3.0m with very low freshwater inflow (34m^3/s on average), and a 32-year mean (±S.E.) salinity of 30.1 (±0.05) with a range from 11.9 to 59.0 (NOAA, 1990).

The seagrass complex within the study site covers approximately 4,057 ha (Gibeaut et al., 2010) mainly bordered by Mustang Island and North Padre Islands and the Gulf Intracoastal Waterway (GICW) (Fig. 2.1). Average shoot density and Root:shoot ratios were collected monthly from March 2007 to June of 2008 (Gutierrez, 2007) and used to delineate the biophysical makeup of the seagrass complex within the study site (Table 2.1). All units were transformed to a standard temporal and spatial scale by average value per month and per square meter in an effort to capture seasonal variability and facilitate scaling up to the entire study site.
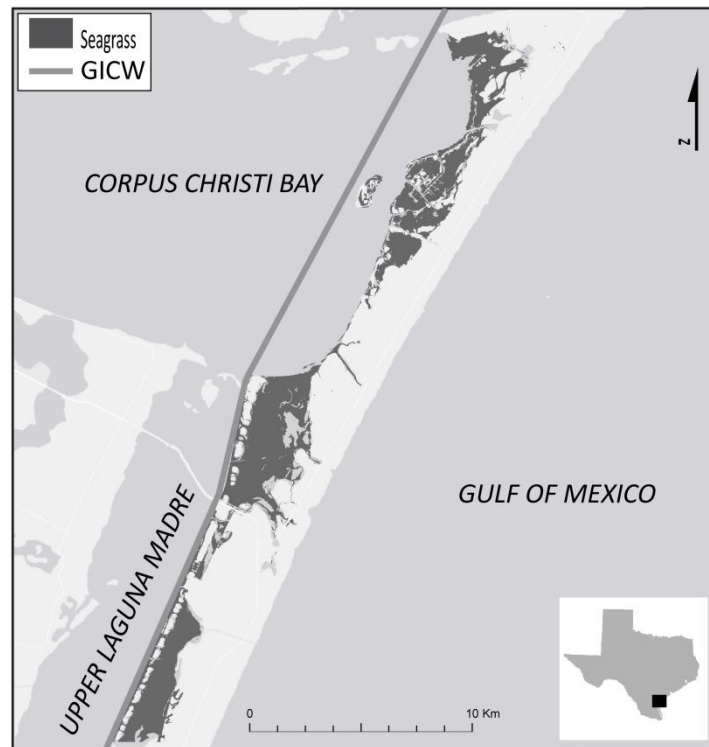
Figure 2.1: Map of the Mustang Island study site, Nueces Co., Texas, USA: 27.4N 97.8W.

Table 2.1: Model parameters, values, and citations.

Parameter	Variable	Value	Units	Source
Areal coverage	AC	4057	Ha	Gibeaut et al., 2010
Growth	μm	9.26×10^{-4}	None	Calculated, This study
Respiration	R	2.52×10^{-4}	None	Calculated, This study
Light-saturated Photosynthesis	P_{max}	317.625	$\mu mol\ 0_2\ gdwt^{-1}\ hr^{-1}$	Dunton, 1996
Whole-plant respiration	R_{avg}	86.375	$\mu mol\ 0_2\ gdwt^{-1}\ hr^{-1}$	Dunton, 1996
Average carbon-based biomass	C_{BM}	86.62	$gC\ m^{-2}$	Dunton 1996; Duarte, 1990
Light saturation	I_k	319	$\mu mol\ m^{-1}\ s^{-1}$	Dunton, 1996
Surface Irradiance	$I(0)$	Input	$\mu mol\ m^{-1}\ s^{-1}$	Dunton and Tomasko, 1994
Light at depth	I_z	Input	$\mu mol\ m^{-1}\ s^{-1}$	Calculated, This study
Average depth	Z	1	m	Constant, This study
Light attenuation coefficient	K	Input	None	Calculated, This study
Average shoot density	SD	2200	$shoots\ m^{-2}$	Gutierrez, 2007
Average shoot biomass	ASB	18.7	$gdwt\ shoot^{-1}$	Dunton, 1996
Root:Shoot	RSR	2.0	None	Dunton, 1996
Nitrogen Acquisition Rate	NA	Input	$gN\ m^{-2}\ mo^{-1}$	Kowalski et al., 2009
Replacement Cost	RC	4.90	$\$\ kgN^{-1}$	Smith, 2008

Model development:

The model simulation of nitrogen acquisition by the seagrass complex was created using Simile v.5.7 (Simulistics, 2011). The modeling environment is based on a "declarative approach" in that each visual component of the model is a functional unit rather than a line of code needed to fulfill further model functions. This allows for the description and implementation of the model to be equivalent as well as remain flexible enough for model components to "stand alone" (Simulistics, 2011). Simile has been used in several large modeling efforts including the Forested Land-Oriented Resource Envisioning System (FLORES), Modelling Mediterranean Ecosystem Dynamics (ModMED), and Multiscale Integrated Earth Systems Model (MIMES) modeling frameworks (Simulistics, 2011). Each of these modeling frameworks is multidisciplinary and collaborative in nature. Therefore, Simile offers flexibility in development of stand-alone components that can then be integrated at will (Haggith et al., 2003; Van Bers et al., 2007). Although much of these efforts have been focused on terrestrial systems, the approach is also applicable to aquatic systems.

Development of the model is structured around a common species within the site, *Halodule wrightii*, and four main components: Light reaching the canopy, change in seagrass shoot density, whole plant biomass, and nitrogen acquisition. The conceptual structure of the model uses light as a driving force to predict seagrass shoot density which is then converted into seagrass biomass. The biomass is then coupled with the amount of nitrogen acquired by the seagrass complex and valued using a replacement

cost methodology based on the cost needed to construct a secondary wastewater treatment facility designed to remove inorganic nitrogen (Smith, 2008).

Change in shoot density (dSD/dt) is the base calculation within the model in that it describes the number of shoots per square meter which is then converted to biomass. Change in shoot density is empirically derived as (Newell and Koch, 2004);

$$dSD/dt = \mu_m[1-e^{(-Iz/Ik)}]SD - rSD \qquad (1)$$

where μ_m represents the growth rate coefficient, I_z is the amount of irradiance reaching the seagrass canopy, I_k is the saturation coefficient, and r is the rate of respiration coefficient.

Light reaching the canopy of the seagrass complex is calculated based on the amount of light reaching the surface of the water diminished by the amount of light that is attenuated due to physical properties of the water column including scattering, absorption, reflection, and water depth, among others. The amount of light reaching the canopy of the seagrass complex was calculated using Beer's Law;

$$I_z = I(0)\, e^{(-kz)} \qquad (2)$$

where I_z (μmol m^{-2} s^{-1}) is the amount of light reaching the canopy, $I(0)$ (μmol m^{-2} s^{-1}) is the surface irradiance, k is the light attenuation coefficient, z (m) is the depth of the water.

Both μm and r represent the metabolic gains and losses from the seagrass complex. These coefficients were calculated based on photosynthetic activity and carbon-based biomass following the form (Newell and Koch, 2004);

$$\mu m = (P_{max}/1.1) / (C_{BM}) \hspace{2cm} (3)$$

where μm is the maximum growth rate coefficient, P_{max} is the light-saturated rate of photosynthesis then divided by the photosynthetic coefficient 1.1 to derive carbon fixation, and C_{BM} which is the average carbon-based biomass.

Respiration rate was also derived by (Newell and Koch, 2004);

$$r = (R_{avg}/1.1) / (C_{BM}) \hspace{2cm} (4)$$

where r is the maximum respiration rate coefficient, R_{avg} divided by the photosynthetic coefficient 1.1 to derive carbon loss, and C_{BM} which is the average carbon-based biomass.

Above-ground biomass (BM_S) of the seagrass complex within the study site was calculated on a monthly basis using the shoot density and average shoot biomass calculated as,

$$BM_s = (SD)(ASB) \hspace{2cm} (5)$$

where BM_S (gdwt m^{-2}) is the above-ground biomass, SD (shoots m^{-2}) is the shoot density, and ASB (gdwt shoot^{-1}).

Below-ground biomass (BMR) is calculated using the root:shoot ratio representative of the study site calculated as,

$$BMR = (BMS)(R{:}S) \qquad (6)$$

where BMR (gdwt m-2) is the below-ground biomass, BMS (gdwt m-2) is the above-ground biomass, and R:S is the root-to-shoot ratio.

Overall monthly biomass of the study site is then scaled using areal coverage. seagrass complex within the study site calculated as;

$$BM = (BM_S + BM_R)AC \qquad (7)$$

where BM (kgdwt) is the total seagrass biomass in the study site, BM_S(kgdwt m^{-2}) is the above-ground biomass, BM_R (kgdwt m^{-2}) is the below-ground biomass, and AC(m^2) is the areal coverage of seagrass complex within the study site.

Nitrogen acquisition value (NAV) was calculated as a function of biomass and reported monthly nitrogen acquisition (Kowalski et al., 2009) based on the replacement cost price of $4.90 kg N^{-1} (Smith, 2008) following the general form;

$$NAV = (BM)(NA)(RC) \qquad (8)$$

where NAV ($) is the value of nitrogen acquired by the seagrass complex, BM (kgdwt) is the total dry weight biomass of the seagrass complex, NA (kg N kgdwt^{-1}) is the monthly nitrogen acquisition rate, and RC ($ kg N^{-1}) is the replacement cost ($4.90 kg N^{-1}) to engineer a facility designed to remove nitrogen from wastewater (Smith, 2008).

Sensitivity analysis:

Sensitivity of the model output, BM and NAV, of the seagrass complex was tested by systematically varying the level of select model inputs. Two separate sensitivity analyses were conducted based on the range of input parameters. First, I_z was systematically varied such that 75%, 50%, and 25% of *I(0)* reaches the canopy of the seagrass complex to delineate the effect on both BM and NAV. Second, monthly NA were systematically varied by ±25% and ±50% of the baseline input for a total of four model output variations on NAV alone due to NA having no affect on the calculation of BM within the model.

Results:

Baseline model simulation:

The baseline model was constructed using a light attenuation coefficient (k = 0.693) such that 50% of the surface irradiance reached the top of the canopy at one meter of depth. Under these assumptions, mean BM (± S.E.) was calculated to be 125.06 gdwt m^{-2} (± 6.67) while mean NAV (± S.E.) was calculated at $1,004,507.90 mo^{-1} (± $43,492.24) (Fig. 2.2). The highest BM and NAV levels were predicted from May to

August during the growing season resulting in an annual NAV of $12,054,095 yr^{-1}. This

value represents the sum of all monthly average NAV from January to December.

A)   B)
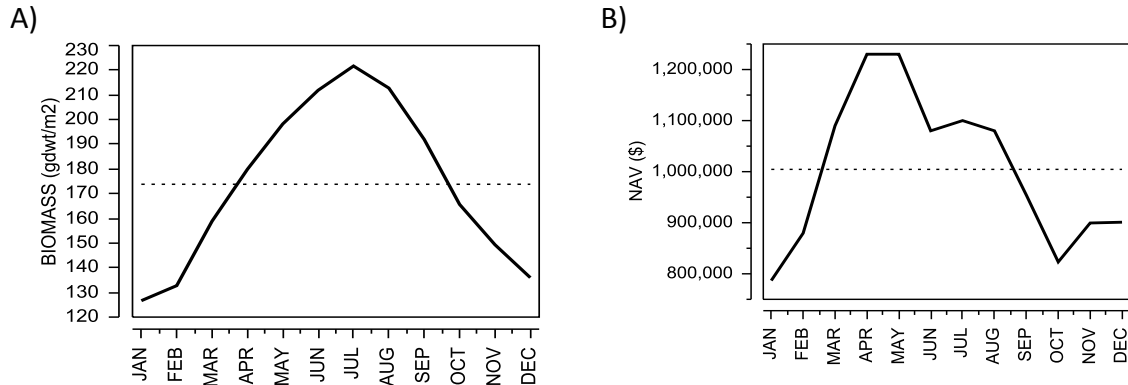
Figure 2.2: Baseline output for A) biomass (---=125.06 gdwt m^{-2} and B) NAV by month
(---=$1,004,507.90 mo^{-1})

Sensitivity analysis (Light):

Both BM and NAV varied significantly as a function of I_z. Both BM and NAV vary

directly with I_z. The baseline results, assuming 50% *I(0)*, were compared to the alternate

intensities, both 75% and 25% *I(0)* to define the upper and lower bounds of the model.

Under the most intense light conditions (75% *I(0)*), BM was calculated to reach a

maximum of 257 gdwt m^{-2}, while under the weakest light conditions (25% *I(0)*), BM was

calculated to reach a minimum of 93 gdwt m^{-2}. Under the same light conditions, NAV

was calculated to reach a maximum of $1,490,000 mo^{-1} and a minimum of $508,754 mo^{-}

^{1}, respectively. A one way ANOVA of the means of both BM and NAV for all intensities

of *I(0)* show that each are statistically different from each other and therefore, light is a

driving force within the model (Fig. 2.3).

Sensitivity analysis (Nitrogen acquisition rate):

NAV varied significantly as a function of NA (Fig 2.4). NAV varies directly with

the NA. By varying the NA as a percentage of the baseline, the results elucidate the

upper and lower bounds of the model output. Under the highest rate of NA (150% of

A)　　　　　　　　　　　　　　　　　　B)

% I(0)	k	Mean BM (gdwt m^{-2})	% Change BM	Mean NAV ($)	% Change NAV
75%	0.287	^{A}208.06	19.6%	^{A}1,206,354	20.1%
50%	0.693	^{B}173.90	0%	^{B}1,004,508	0%
25%	1.386	^{C}125.05	-28.1%	^{C}723,587	27.9%
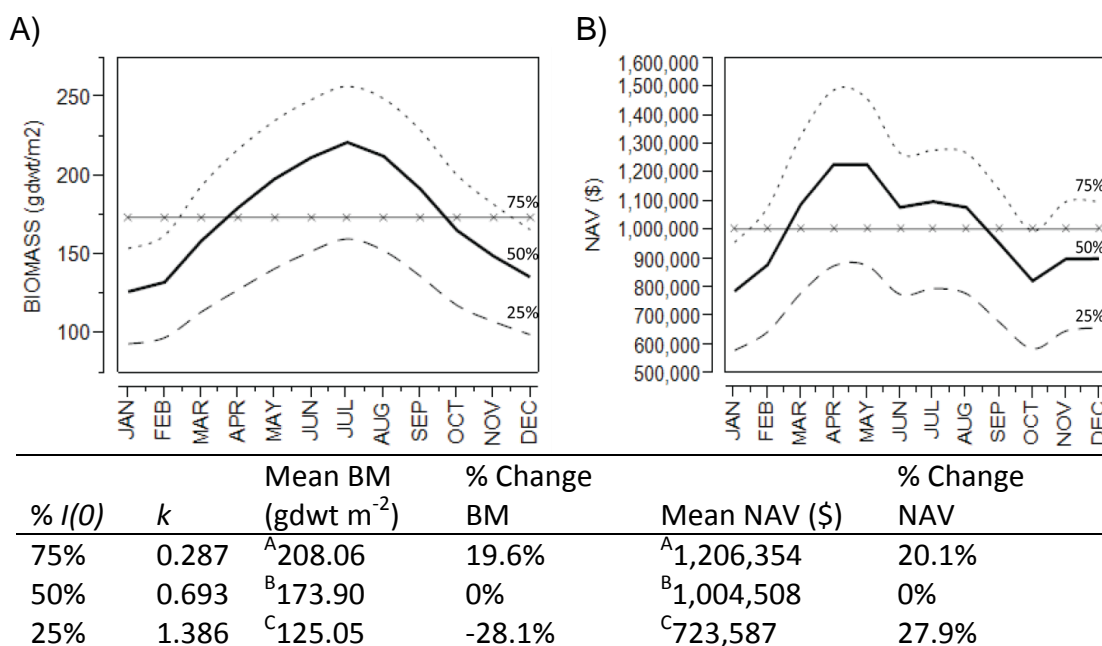
Figure 2.3: Changes in A) BM and B) NAV as a function of I(0). *Note superscripts denote significant differences among means (p<0.05).*

the baseline) the maximum NAV was calculated as $2,550,000 mo^{-1} while at the lowest

rate of NA (50% of the baseline) the minimum NAV was calculated to be $524,507 mo^{-1}.

A t-test shows that that the mean values for NA are statistically different from one

another except between the 75% and 50% levels where no difference was detected (Fig.

2.4).
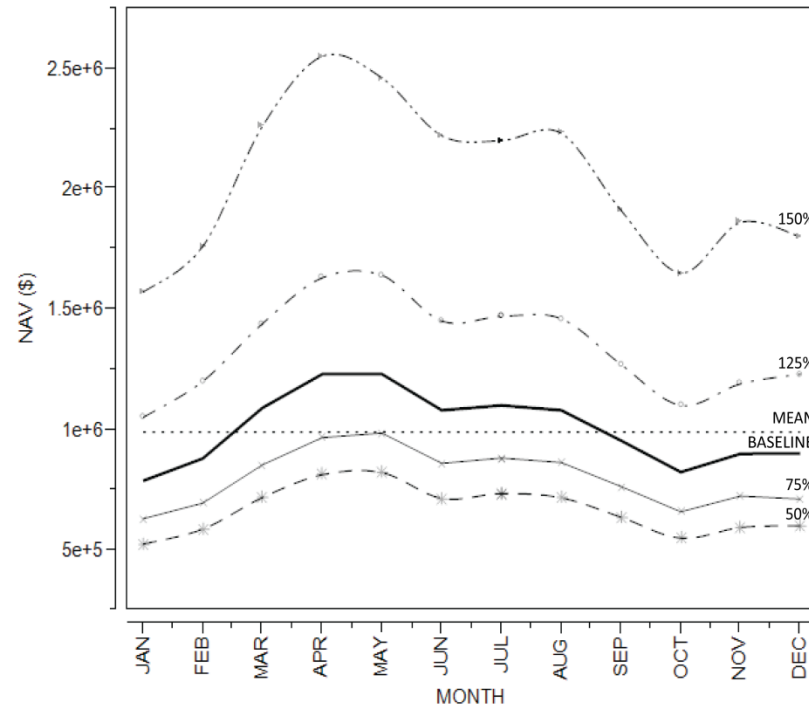
% of Baseline NA	Mean NAV ($)	% Change NAV
150%	[A]2,039,167	130%
125%	[B]1,344,167	33.8%
Baseline	[C]1,004,508	0%
75%	[D]799,198	20.4%
50%	[D]667,239	33.6%

Figure 2.4: Changes in NAV as a function of NA. *Note superscripts denote significant differences among means (p<0.05).*

Discussion:

Baseline model output:

The baseline model output follows an expected pattern of increased BM during the late spring then beginning to decline in early fall (May-September). This follows a similar pattern to the driving force of the model, *I(0)*. Although this trend was expected, it should be mentioned that temperature was not included in the model framework. Surface irradiance and temperature are correlated on the basis of monthly averages throughout the year (Dunton, 1994; Lee and Dunton, 1999; Burd and Dunton, 2001; Lee

et al., 2007). The lack of water temperature within the model has important

implications in that two important model components, respiration and production, are

most likely driven by the seasonal variation in water temperature. Surface irradiance,

rather than temperature, was chosen based on the need for light during photosynthetic

processes (Lee and Dunton, 1999; Lee et al., 2005). Therefore, in future iterations of the

model, surface irradiance and water temperature could be more closely examined in

order to more explicitly delineate the effects each has on the biomass of the seagrass

complex within this and other study sites.

Ultimately, the output of the model agreed with several studies from estuaries

along the coast of Texas with respect to seasonal fluctuations in BM. In the East Flats

area, contained within the current study site, Dunton (1994) observed a similar seasonal

pattern in biomass in which highest annual biomass measurements were observed

during the summer growing season. High annual biomass measurement ranged from

150 - 500gdwt m^{-2} (Dunton, 1994), which compares favorably to the current model

output of 225 gdwt m^{-2}. Within the lower Laguna Madre, Burd and Dunton (2001)

reported an annual maximum biomass of 124 - 229 gdwt m^{-2}, which again is similar to

the output of the current model. In a nearby study site, Kowalski et al (2009) reported a

maximum biomass of 272 - 279 gdwt m^{-2}. Therefore, it is believed that the current

model preserves the seasonal trend as well as the magnitude of the predicted BM based

on comparable, independent measurements taken along the Texas coast.

Previous nitrogen uptake studies have reported a general seasonal trend of high

uptake rates during the fall, while the lowest rates are observed in the spring (Dawes

and Guiry, 1992; Perez-Lloens and Niell, 1993; Short et al. 1993; Lee and Dunton, 1999).

The current model output does not follow this trend due to the combination of both BM

and NA being used to calculate NAV. Although the seasonal trend for NAV within the

model is different, annual inorganic nitrogen acquisition by *Thalassia testudinum* was

measured in the East Flats, which is in the northern portion of the study site, and it was

found that the average annual nitrogen acquisition rate was 97 g N m^{-2} yr^{-1} (Lee and

Dunton, 1999). If this measurement were scaled to represent the entire areal extent of

the current study site (4057 ha) using the same replacement cost ($4.90 kg N^{-1}), the

annual NAV would be $19,282,921 yr^{-1} as compared to the current model output NAV of

$12,054,095 yr^{-1}. Therefore, it is likely that these numbers are representative of the

contribution the seagrass complex has to nitrogen regulation within the bay system.

Currently, very few studies exist that delineate the value of ecosystem services

provided by seagrass communities and often focus on services such as habitat

(McArthur and Boland, 2004; Unsworth et al., 2010), food provisioning (Samonte-Tan,

2007), and raw materials (Hargreaves-Allen, 2010). One study found focused on the

contribution of seagrass communities to nutrient regulation and reported a value of

$27,600 ha^{-1} yr^{-1} (Brenner et al., 2010).   If this value was scaled to represent the area of

the current study, it would equate to $111,973,200 yr^{-1} as compared to the model

calculation of $12,054,095 yr^{-1}. It is difficult to compare the vast differences between

these two studies in that the valuation methodology for each is quite different. Brenner

et al. (2010) used a benefit transfer methodology where a value used in one site was

adjusted in a manner allowing its use at another site. This value is also based on a point

transfer. There are some difficulties in this type of valuation in that it is not able to

capture the drivers of value such as physical location, the demographics of those benefit

from the given service, nor attributes of the study that might not be applicable to a new

site (Bergstrom and Taylor, 2006; Brookshire and Chermak, 2007).   Conversely, the

current study is based on a replacement cost approach. In this study, the ecological

function, NA, is calculated and then the price of replacing the same service is based on

engineering a wastewater treatment facility to replace the same function. Considering

NA of the current model is very similar to that reported in other studies (Lee and

Dunton, 1999), it is likely that the NAV calculated by the model accurately represents

the value of nitrogen uptake performed by the seagrass complex within the study site.

Sensitivity analysis:

Surface Irradiance ($I(0)$) is the driving force behind the model calculation,

therefore model output for BM and NAV are directly correlated and highly sensitive to

$I(0)$. $I(0)$ is adjusted by the coefficient k to calculate I_z. Therefore, $I(0)$ has a direct

correlation with both BM and NAV. This model assumes that 50% of $I(0)$ (k=0.693) is the

baseline output. This value is chosen primarily due to the shallow nature of the study

site and the relative abundance of light reaching the canopy, even though *Halodule*

wrightii has a minimum light requirement of 20% *I(0)* (Dunton, 1996). Therefore, this

model was not designed to predict the loss of seagrass BM due to long periods of light

limitation stress. It is important to note that although *I(0)* follows a seasonal,

predictable cycle (with the exception of cloud cover), variations in k can be driven by

more stochastic parameters that affect water clarity such as water depth, wind,

sediment suspension, high nutrient runoff, and algal blooms (Dennison et al., 1993).

The output of the model may be improved by including finer scale data for light

attenuation. Yet, in its current form, the model output was in general agreement with

other studies performed in estuaries along the Texas coast (Dunton, 1994; Dunton,

1996; Lee and Dunton, 1999).

The second portion of the sensitivity analysis shows that NA is also a highly

influential portion of the NAV calculation. NA within the model was based on the whole

plant. Previous studies have shown that different species of nitrogenous compounds

such as nitrate, nitrite, and ammonium are taken up more readily by different portions

of each plant (Dawes and Guiry, 1992; Perez-Lloens and Niell, 1993; Short et al. 1993;

Lee and Dunton, 1999). This model does not explicitly partition above and below

ground biomass. Therefore, it is possible to use more fine-scale data to predict uptake

of a specific nitrogenous compound within the model if the input data is available. In

the current model, this was not done due to the inability to derive valuations for more

fine-scale acquisition of individual nitrogenous compounds.

Management implications:

It is important to understand that although seagrasses are important in nutrient

regulation, they are not responsible for widespread removal of nitrogen from the

estuarine system (Duarte, 1990; Hemminga et al., 1991; Hemminga and Duarte, 2000,

Romero et al., 2006. Much of the nitrogen fixation and denitrification actually takes

place within the rhizosphere by bacteria that reside there (Herbert, 1999; Welsh, 2000).

Shoot acquisition of nitrogen is usually in the form of nitrate and highest during the

summer season while root acquisition of nitrogen is usually in the form of ammonium

and dominant during the winter (Lee and Dunton, 1999). Also, a large amount of

nitrogen is recycled both within individual plants and lost to the sediments that

surround the seagrass complex (Dunton, 1996; Lee and Dunton, 1999; Kaldy and

Dunton, 2000; Burd and Dunton, 2001). Due to the complicated nature of nitrogen

dynamics, the current model is a tool used to help quantify the relative contribution of

the seagrass complex to nutrient regulation within the study site.

Light is the driver of primary production and ultimately the health of seagrasses

is hinged on the appropriate quantity and quality of light reaching the canopy. Light

limitation can be driven by several factors including eutrophication as well as increased

turbidity from dredging activities. Eutrophication often reduces seagrass cover through

increased phytoplankton and epiphyte growth which result in light limitation (Valiella et

al., 1992; Dennison et al., 1993; Tomasko et al., 1996; Cloern, 2001; Cardosa et al., 2004;

Paling et al., 2009). Loss of seagrass cover often results in an irreversible regime shift

where macroalgal species become dominant. Although, it is possible to reduce nitrogen

loading within the system, reversing the regime shift is far more difficult (Lodge et al.,

2006). Similar results have been seen in other forms of light limitation stress such as

those resulting from increased turbidity from maintenance dredging and other activities

(Odum, 1963; Fonseca et al., 1998; Erftemeijer et al., 2006; Orth et al., 2006). The GICW

borders much of the study site (Fig 2.1). This is an economic artery important to the

maritime shipping industry within the state of Texas. Regular dredging is required to

maintain a functional depth of the GICW, but often results in the deposition of poor

quality sediments and increased turbidity for the surrounding seagrass communities (Pulich, 1999). Therefore, the loss of seagrass cover results in the loss of the ecosystem services they provide (Duarte, 2000).

The strength of the model lies in the ability to communicate ecological function in more broadly-understood, economic terms. This is important in light of the threats that are affecting seagrass communities worldwide: nutrient loading which induces eutrophication, light limitation due to changes in water clarity, and changes in community structure. Many of these problems are anthropogenically driven. Helping people to understand the value of healthy, intact seagrass communities is paramount to their conservation (Beck et al., 2001; Deegan et al., 2002; Fourqurean et al, 2003; Heck et al., 2003; Erftemeijer et al., 2006; Greening and Janicki, 2006; Orth et al., 2006; Russel et al., 2011).

Chapter III:

Linking ecological function and ecosystem service provision of the coastal salt marsh

plant, *Spartina alterniflora*, as a function of nitrogen standing stock

Abstract:

Salt marsh habitats are recognized as being highly productive, ecologically important, and economically valuable. The contribution of salt marsh habitats to nitrogen cycling in estuarine and coastal waters has received considerable attention. In this study, a dynamic ecosystem model was assessed to predict the seasonal fluctuation of above- and below-ground biomass of the salt marsh.   This model is then used to calculate the value of the nitrogen standing stock using a replacement cost methodology. Therefore, the goal of this project was to create a framework explaining the economic importance of the salt marsh as it contributes to nitrogen cycling in a manner that is transferrable to multiple study sites given the appropriate local data. The model developed calculated the value of the nitrogen standing stock as $1612 ha^{-1} yr^{-1} which falls within the range of similar studies, yet is limited in its application by multiple factors.

Introduction:

Spartina alterniflora, is the dominant species found in intertidal salt marsh habitats along the Atlantic and Gulf coasts of the U.S., with the largest singular concentration being found along the coast of Louisiana (Darby and Turner, 2008a). The economic importance of coastal salt marshes was formally recognized in the scientific literature as early as 1928 (Viosca, 1928). In subsequent years, considerable effort has

been made to communicate the ecological and economic value of the salt marsh. It is

widely accepted that salt marshes are highly productive ecosystems that play an

important role in nutrient cycling (Teal, 1962; Valiela & Teal, 1979; Buresh et al., 1980;

Turner et al., 2004, Darby & Turner, 2008a). The valuation of salt marsh ecosystems has

mainly focused on the abatement of storm damage as well as recreational use of the

salt marsh for fishing, bird watching, and boating (Costanza et al., 1989; Barbier et al.,

1997; Bell, 1997; Woodward & Wui, 2001; Johnson et al., 2002; Curtis, 2004; Costanza,

2008)

The role of salt marsh ecosystems in nitrogen cycling has been particularly

relevant in light of historical increases of agriculturally derived nitrogen inputs as

fertilizer into estuarine waters (Teal et al., 1979; de Groot, 1992; Rozema et al., 2000).

Three main processes provide pathways for nitrogen movement within the salt marsh

habitat: inputs, losses, and exchanges. Salt marshes are able to remove nitrogen from

coastal systems through denitrification (DeLaune et al., 1983; Hamersley & Howes,

2005; Seitzinger et al., 2006; Sousa et al., 2012), seasonal plant uptake (Valiela & Teal,

1979; Anderson et al., 1997; Moseman, 2007), and long-term sequestration through

burial (Hutchinson et al., 1995; Goodman et al., 2007) in benthic sediments. These

pathways allow for opportunities to mitigate the possible effects of excess nitrogenous

inputs (Tobias, 2009). Salt marshes also represent a unique link in the coastal nitrogen

cycle in that many are considered nitrogen limited with respect to ability to accumulate

above-ground biomass (Morris, 1991; Darby and Turner, 2008a; Darby & Turner, 2008b).

On the other hand, below-ground processes and transfers within the plant are not as

readily understood (Turner et al., 2004; Darby & Turner, 2008a). The difficulty in using

these processes to define the provision of ecosystem services on a broad spatial scale is

that many are highly stochastic and dependent on regional processes that would be

impossible to generalize across the natural range. For this reason, using seasonal

fluctuations in biomass and areal extent as the basis for service provision allows for a

transferrable method of comparison and communication.

Monitoring efforts for biomass and areal extent of the salt marsh have been

used as indicators of salt marsh health. Techniques have been developed to monitor

these parameters on a large scale, namely using multispectral satellite data (Hardisky et

al., 1983; Hardisky et al., 1984; Gross et al., 1991; Jensen et al., 2002; Adam et al., 2010).

One common technique requires calculation of the Normalized Difference Vegetation

Index (NDVI), the ratio of red-light wavelengths to near-infrared wavelengths in the

electromagnetic spectrum. It is based on the ability of the chlorophyll contained in

green plants to absorb the red portion of the spectrum, whereas the rest of the plant is

more likely to reflect the near-infrared portion of the spectrum (Tucker, 1979, Jackson

et al., 1983, Tucker et al., 1991).   NDVI values range from -1.0 to 1.0 where the more

positive portion of the index is interpreted as having more viable green plant material

and the more negative portion more barren surfaces such as rock, ice, and bare earth.

Quantifying biomass can also be accomplished through the collection of field data. Field

collection is labor- and time-intensive as well as difficult to interpret over large spatial

scales. The NDVI method allows for monitoring of large spatial scales and the

availability of large amounts of data are useful in this type of modeling platform

(Eidenshink, 2006).

Salt marsh habitats have been recognized as providing at least 22 ecosystem

services having at least one form of economic valuation attached to them. An extensive

literature search identified 186 studies that placed an economic valuation on the salt

marsh as a function of the ecosystem services provided by this habitat type throughout

the world. Many of these studies focus on only two ecosystem services, disturbance

regulation and recreation, which comprise nearly one-fourth of that total (GecoServ,

2011). Of the 186 total studies identified, only five placed an economic valuation on the

nutrient cycling services. The range of economic values appears to vary widely, from

$5.96 to $8,767 ha^{-1} yr^{-1}, both indexed for 2008 (de Groot, 1992; Bystrom, 2000; Badola

and Hussain, 2003; Curtis, 2004; Zhao et al., 2004; Ragkos et al., 2006). Although overall

range of values is great, for those studies using replacement cost methodologies, the

range shrinks to $1,011 to $8,766 ha^{-1} yr^{-1} (de Groot, 1992; Bystrom, 2000; Badola and

Hussain, 2003).

In light of these values, the goal of this project was to create a methodology

linking the contribution of salt marsh habitats to nitrogen cycling and communicate this

important ecosystem service in broadly-understood, economic terms.   The model seeks

first to quantify above- and below-ground *Spartina* sp. biomass using multispectral

satellite data in an effort to quantify the amount of nitrogen found in the standing stock

biomass (NSS). Using this snapshot of NSS found in the living biomass, a replacement

cost approach is used to create an economic valuation of the contribution of the salt

marsh to nitrogen cycling processes. The model is built in a manner that allows similar types of analyses to be applied to various study sites given the availability of NDVI data in the United States.

Methods:

Study site:

Mustang Island is a high profile barrier island that separates the Corpus Christi Bay system from the Gulf of Mexico and lies within Nueces County, TX (Simms et al., 2006). The Corpus Christi Bay system is large (497 km^2 at mean low water) and includes minor bays Redfish, Nueces, and Oso. The average depth of the Corpus Christi Bay system is 3.0 m with very low freshwater inflow (34m^3 s-^{1} on average), and a 32-year mean (±S.E.) salinity of 30.1 (±0.05) with a range from 11.9 to 59.0 (NOAA, 1990). The salt marsh complex within the study site covers approximately 925 ha bordered by Mustang and North Padre Islands as well as Corpus Christi Bay and the Gulf Intracoastal Waterway (GICW) (Fig. 3.1). Nueces County has a population of approximately 340,000 people and is growing at a rate of approximately 1% annually. Approximately 140,000 housing units are found within the county with a median household income of $43,280. The median household income for the state of Texas is $49,646 with annual population growth at 2.1% (US Census, 2010). This imbalance leaves room for growth of the coastal areas of Nueces County, including the Mustang Island study site.

Model development:

The model simulation of NSS by the salt marsh complex was created using Simile v.5.7 (Simulistics, 2011). The modeling environment is based on a "declarative

approach" in that each visual component of the model is a functional unit rather than a line of code needed to fulfill further model functions. This allows for the description and implementation of the model to be equivalent as well as remain flexible enough for model components to "stand alone" (Simulistics, 2011). Simile[TM] has been used in several large modeling efforts including the Forested Land-Oriented Resource Envisioning System (FLORES), Modelling Mediterranean Ecosystem Dynamics (ModMED), and Multiscale Integrated Earth Systems Model (MIMES) modeling frameworks (Simulistics, 2011). Each of these modeling frameworks is multidisciplinary and collaborative in nature. Therefore, Simile[TM] offers flexibility in development of stand-alone components that can then be integrated at will (Haggith et al., 2003; Van Bers et al., 2007). Although much of these efforts have been focused on terrestrial systems, the approach is also adaptable to aquatic systems.

Development of the model is structured around the predominant plant species within the site, *Spartina alterniflora*, and four main components: (1) biomass calculated as a function of NDVI and published correlations between living above and below-ground biomass, (2) the average mass (g) of nitrogen found in a gram of dry weight for both above- and below-ground biomass, (3) areal extent of the salt marsh complex within the study site, and (4) the replacement cost price associated with building a water treatment facility to remove nitrogenous compounds from wastewater in order to improve water quality. All units were transformed to a standard temporal and spatial scale by average values per month and per square meter in an effort to capture seasonal variability and facilitate scaling up to the entire study site (Table 3.1).
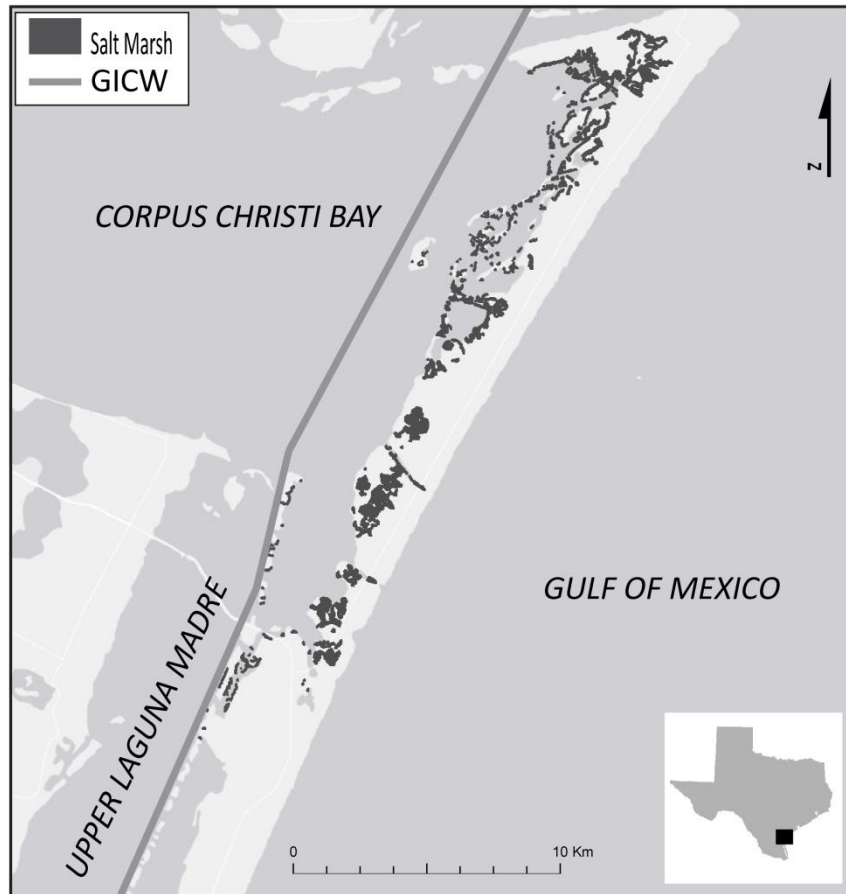
Figure 3.1: Map of the Mustang Island study site. Nueces Co., Texas, USA: 27.4 N 97.8W. *Note: GICW refers to the Gulf Intercoastal Waterway.*

Table 3.1: Model inputs, values, and citations.

Parameter	Abbreviation	Value	Units	Source
Normalized Difference Vegetation Index	NDVI	Input	None	Eidenshink, 2006
Areal Extent	AE	924.5	Ha	Gibeaut et al., 2010
Replacement Cost	RC	4.90	$ kgN^{-1}	Smith, 2008

Above-ground biomass (ABM) is the base calculation within the model that allows for the eventual valuation of NSS. Hardisky et al. (1984) developed this relationship using spectral radiance data collected along transects of a mostly short-form, monotypic *Spartina alterniflora* salt marsh in Delaware using a hand-held

radiometer configured to emulate bands 3,4, and 5 of the Landsat 4 thematic mapper.

These data were used to calculate NDVI and correlated with field samples of biomass

using regression techniques (r^2=0.64) in an effort to create a relationship capable of

remotely sensing salt marsh biomass. Therefore, ABM is calculated using the NDVI

following the general form (Hardisky et al., 1984):

$$dABM/dt \ = \ exp[(NDVI\text{-}0.149) \, / \, (0.096)] \qquad (1)$$

where ABM is the average live above-ground biomass per month (gdwt m^{-2} mo^{-1}) and

NDVI is the average monthly index between 1989 and 2006 for path 41 row 26 of the 1-

km Advanced Very High Resolution Radiometer (AVHRR) satellite data (Eidenshink,

2006). This data is smoothed and averaged to account for cloudiness and noise within

the data set to derive a monthly NDVI value (Swets et al., 1999). This was then centered

using historical peak biomass of 371.5 gdwt m^{-2} (Turner and Gosselink, 1975). This data

is also correlated with the National Landcover Dataset in order to classify each 1km^{-2}

pixel (USGS, 2000). Therefore, this model is based only on NDVI values identified for

herbaceous wetlands.

Once ABM has been calculated, it is used as the basis for calculating

belowground biomass (BBM) following the form (Gross et al., 1991):

$$dBBM/dt = exp(0.713 * ln(ABM) + 2.235) \qquad (2)$$

where BBM is the average monthly below-ground biomass (gdwt m^{-2} mo^{-1}) and ABM is

the average monthly above-ground biomass (gdwt m^{-2} mo^{-1}).

In order to calculate the NSS both ABM and BBM must be converted to nitrogen

units. This is accomplished by multiplying the average grams of nitrogen found in each

gram of ABM and BBM following the form (Darby and Turner, 2008a):

$$NSSa = ABM * 0.014 \tag{3}$$

$$and$$

$$NSSb = BBM * 0.0182 \tag{4}$$

where NSSa is the nitrogen standing stock in above-ground biomass (gN $gdwt^{-1}$) and

NSSb is the nitrogen standing stock in below-ground biomass (gN $gdwt^{-1}$).

NSS value (NSSV) was calculated as a function of NSS, areal extent (AE) of the salt

marsh, and the replacement cost (RC) price following the general form;

$$NSSV_a = (NSS_a)(AE)(RC) \tag{5}$$

$$and$$

$$NSSV_b = (NSS_b)(AE)(RC) \tag{6}$$

where $NSSV_a$ ($) is the value of NSS in the ABM of the, $NSSV_b$ ($) is the value of NSS in

the BBM, AE (m^{-2})is the areal extent of the salt marsh complex, and RC ($ gN^{-1}) is the

replacement cost price, $4.90 kgN^{-1},is based on a study done by the Pennsylvania

Department of Environmental Protection where it was determined that this was the

price to engineer and build a one million gallon per day wastewater treatment facility

designed to reduce Total Nitrogen from 18mg l^{-1} to 8mg l^{-1}. This price was calculated in

an effort to inform a developing nutrient trading program (Smith, 2008).

Model performance and comparison:

The output of the model was compared to published data of NSS in a Louisiana

salt marsh (Darby and Turner, 2008a) in order to assess the validity of the model with

respect to NSSV. This was done by converting published NSS for both above-ground and

below-ground portions of the salt marsh complex by calculating NSSV in the same

manner described in equations 5 and 6 above. Once common units were derived, each

data set was compared using one-way ANOVA and post-hoc Tukey-HSD test for

comparison of means using JMP version 9.0 statistical software (SAS Institute Inc.: 100

SAS Campus Dr., Cary, NC, 27513, USA).

Results:

Baseline model output:

The baseline model was constructed using NDVI data from path 41 row 26 which

encompasses the entire study site to calculate ABM (mean: 225.2 gdwt m^{-2}; range: 176.3

- 371.5 gdwt m^{-2}) and BBM (mean: 441.5 gdwt m^{-2} range: 373.4 - 635.4 gdwt m^{-2}). This

translated into a $NSSV_a$ (mean: $142,921 mo-1; range: $111,834 - $235,623 mo^{-1}) and a

$NSSV_b$ (mean: $364,041 mo^{-1}; range: $307,945 - $523,880 mo^{-1}) (Figs. 3.2 a-d).

Model performance and comparison:

Darby and Turner (2008a) measured the NSS of a monotypic stand of *Spartina*

sp. in Louisiana by drying grinding and measuring total nutrient content using a nutrient

analyzer. Using the data collected in their study to calculate NSSV and using the

methodology above, a comparison shows that the model was able to accurately capture

the seasonal variability of the aboveground NSS while not mimicking the same trend in

the belowground NSS (Figs. 3.3 a- c). The model output for $NSSV_a$ was statistically

different from the measured data set, while the $NSSV_b$ and $NSSV_{a+b}$ were not statistically

different from one another (Table 3.2).
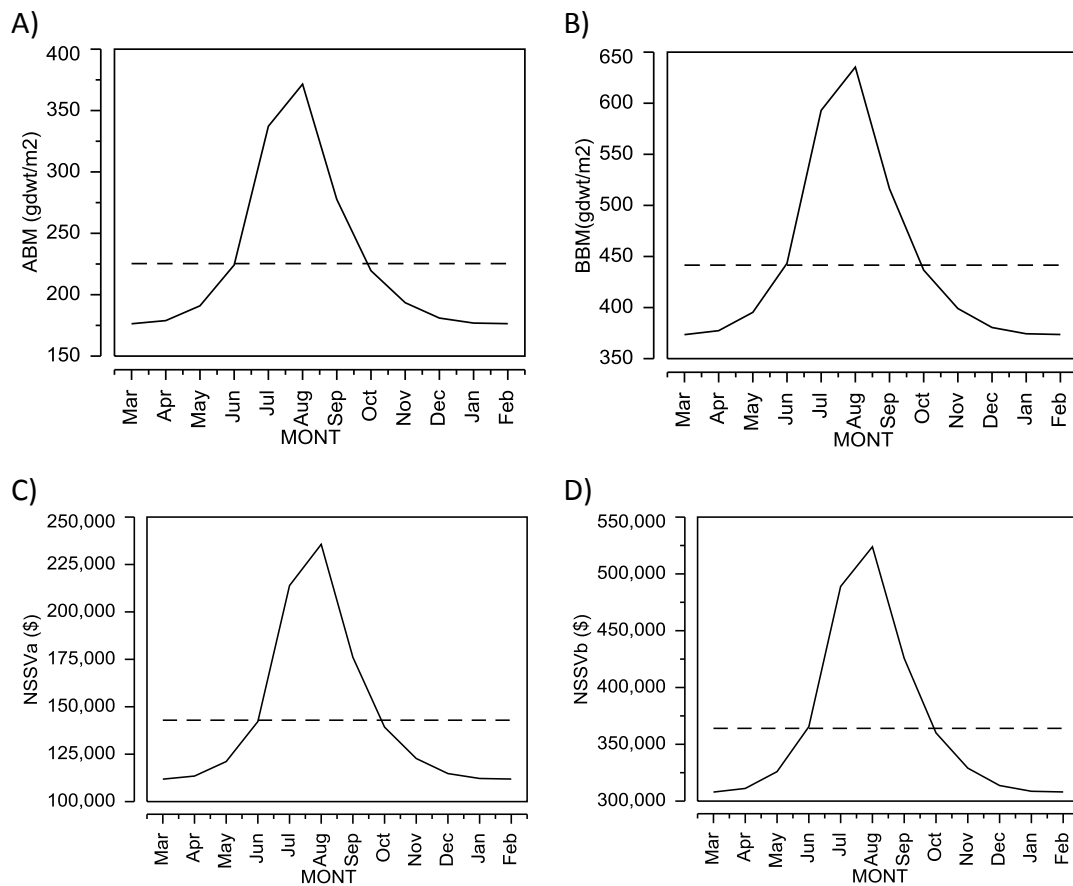
Figure 3.2: Baseline model output for A) above (---=225.2 gdwt m^{-2}) and B) below-ground (---=441.5 gdwt m^{-2}) biomass and C) above- (---=$142,921 mo^{-1}) and D) below-ground (---=$364,041 mo^{-1}) NSSV.

A)
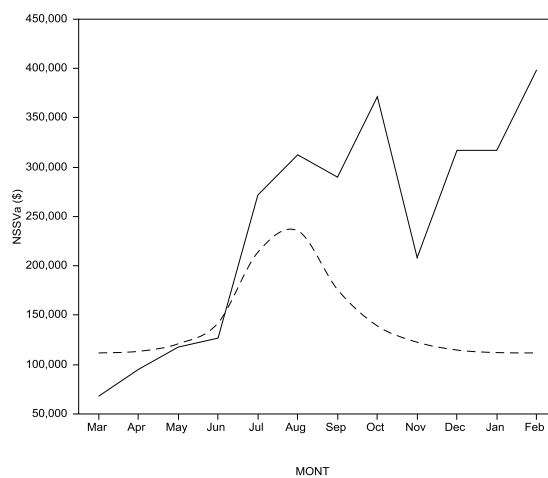
B)
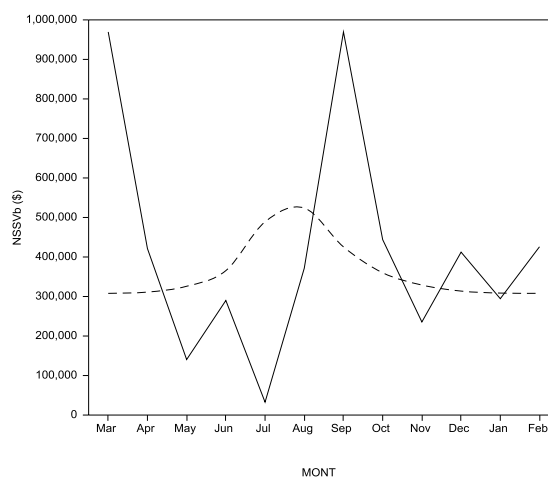
C)
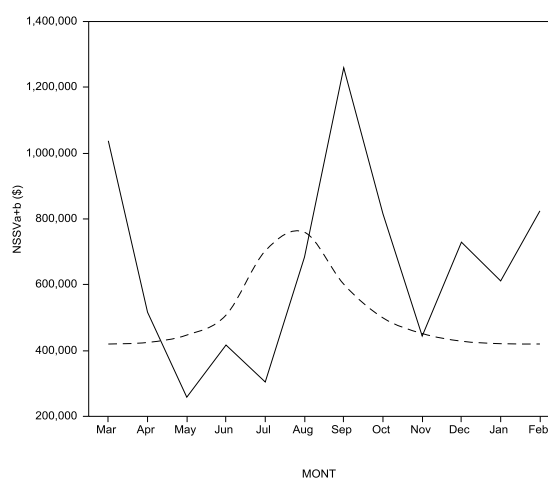
Figure 3.3: Model (dashed line) comparison to Darby and Turner (solid line) (2008a) for A) NSSV$_a$, B) NSSV$_b$, and C) NSSV$_{a+b}$.

Table 3.2: Mean values (± S.E.) of model output as compared to field measurements.
Note: Superscripts denote significant differences between means (p<0.05)

| Parameter | Mean Value ($ mo-1) | | S.E. of comparison |
	Model	Darby and Turner, 2008	
NSSV$_a$	[A]$142,921	[B]$241,24	±24,818
NSSV$_b$	[C]$364,041	[C]$417,245	±60,429
NSSV$_{a+b}$	[D]$506,961	[D]$658,486	±65,376

Discussion:

Baseline model output:

The baseline model output sought to capture the fluctuations in biomass using

measurements of NDVI and correlations between above- and below-ground biomass

(Hardisky et al., 1983; Hardisky et al., 1984; Gross et al., 1991). With respect to both

above- and below-ground biomass, the model replicated the general increase of

biomass in the spring with peak living biomass being observed in late summer to early

fall published in the scientific literature (Turner and Gosselink, 1975; Morris and Haskin,

1990; Mitch and Gosselink, 1993; Darby and Turner, 2008a; Darby and Turner, 2008b).

But, the model is unable to replicate the variations in NSSV. This may be explained by

several factors that determine how the model is structured and the data that supports

it.

Model performance and comparison:

Biomass is calculated in the model using NDVI determined from AVHRR images,

which can be somewhat problematic. The scale at which the AVHRR images are taken, 1

km^{-2}, is much greater than the areal extent of any singular patch of *Spartina* found in the

current study site. This could lead to significant "bleeding" of other land cover types

captured within a singular pixel. This data has been correlated with the National Land

Cover Dataset at a regional level in an effort to control for this problem (USGS, 2000).

The current model employs only the NDVI values classified as "herbaceous wetlands".

The data set was also averaged on a monthly basis across the 15-year span in an effort

to represent an "average" year. Although this data is less than ideal, it was chosen due

to the uniform nature in which it is collected. Standardized methodology allows for

access to data processed in the same manner for study sites across the United States

(Eidenshink, 2006).

Beyond scale, another possible explanation for the inability of the model to

replicate fluctuations in NSSV between model output and field data is the use of remote

sensing data. Remote sensing biomass, especially of wetalnds, can be confounded by

soil saturation, canopy geometry, among other factors (Hardisky et al, 1984; Klemas,

2001; Adam et al., 2010). In the current model, there is no method for controlling for

these factors. Field studies in contrast have the ability to measure salt marsh

characteristics directly where these factors are easily controlled or unimportant.

Comparing model output to field collected data is done in an effort to gauge the utility

of the model. Although the model is able to replicate the general biomass trend found

in the literature (Turner and Gosselink, 1975; Morris and Haskin, 1990; Mitch and

Gosselink, 1993; Darby and Turner, 2008a; Darby and Turner, 2008b), there are

considerable differences among the two types of data that are not able to be captured.

It is also difficult to compare the model output to the field data based on

differences between the physical make-up of the salt marsh in the current study site

and those that would be typical along the Louisiana coast. In comparison, *Spartina*

marshes in the current study site are much smaller in areal extent and are not as continuous as those found in Louisiana (White et al., 2002; Darby and Turner, 2008a; Darby and Turner, 2008b; Comeaux et al., 2012). Lack of freshwater inflow to the current study site is also a difference, as salinities in the current study site range from 11.9 to 59.0 while the Louisiana study site range from 7 to 20 (NOAA, 1990; Darby and Turner, 2008a). Tidal range, erosion, nutrient inputs, and relative sea-level rise all contribute to difference between salt marshes found in the study site and those found in Louisiana.   Despite a multitude of differences, comparing data from such different study sites was done on the basis that the NSS data published in Darby and Turner (2008a) is the only know time series of its kind in the published literature.

Discrepancies in the model output and field measurements may also be explained by stochastic environmental variables that drive nitrogen retention and translocation within each individual plant. The model does imitate the high level of NSSV represented in the living biomass during the growing season; it does not capture those processes taking place during the winter and into the beginning of spring. During this time, the shift in biomass is from above- to below-ground, may explain these discrepancies (Darby and Turner, 2008a). Internal exchanges within the plant are a significant portion of nitrogen needed for maintenance of off-season biomass, which studies show to be approximately 1 to 33 gN m^{-2} yr^{-1} (Tobias, 2009) from above- to belowground tissues and vice versa during the growing season. These exchanges represent a significant and important source of nitrogen needed by salt marsh plants (White & Reddy, 2009).

The current model expresses NSSV as a product of biomass and a direct ratio to average mass of nitrogen per gram of dry weight regardless of molecular form. This is calculated in an effort to capture a singular economic metric that could be used to compare nitrogen cycling processes across spatial and temporal scales. Perhaps the best measurement of the ability of salt marshes to remove nitrogen is the level of denitrification resulting in the loss of nitrogen gas to the atmosphere. Studies show that denitrification rates range from 0 to 60 gN m^{-2} (Tobias, 2009), but comparison among studies is difficult based on discrepancies in methodologies (Seitzinger, 2006). Burial of nitrogenous compounds in the sediments surrounding salt marshes is perhaps another metric that would describe the ability to remove excess nitrogen from a given system. Studies show that burial rates can range from 1 to 23 gN m^{-2} yr^{-1} (Tobias, 2009). The rate of burial is again stochastic and subject to relative sea level rise and other drivers making it difficult to create a common metric for comparison (Hutchinson et al., 1995; Goodman et al., 2007).

Given the lack of explanatory capability in the model, it is able to predict the seasonal fluctuations in both above- and below-ground biomass and calculate NSSV that is both in-line with other studies and not statistically different from field measurements on a whole-plant basis (Darby and Turner, 2008a). Therefore, the model is able to capture a portion of the ecological function of the salt marsh plant, *Spartina alterniflora*, in regards to nitrogen cycling and effectively communicates that function in economic terms (Barbier, 2000; Mitsch & Gosselink, 2000; Turner et al., 2000).

The NSSV output ($1,612 ha^{-1} yr^{-1}: converted for comparison) from the model falls within the range of economic values published in the scientific literature (de Groot, 1992; Bystrom, 2000; Badola and Hussain, 2003; Curtis, 2004; Zhao et al., 2004; Ragkos et al., 2006) (Table 3.3). Several of these studies use a replacement cost approach also to determine economic valuation. Unfortunately, none directly link the ecological function of the salt marsh at the spatial scale found in the current study (de Groot, 1992; Bystrom, 2000; Badola and Hussain, 2003). Therefore, the current study represents a considerable improvement in the ability to calculate similar values for salt marshes found in other regions. It should be noted that this RC price may not be appropriate for all study sites, but choosing an appropriate value and substituting it into the model should be left to the discretion of those with local knowledge of the study site in question. Transferring this RC price to the current study site is an area that is likely to introduce error in that it may not adequately represent the construction, engineering, or operation costs associated with a similar facility built locally. The platform of the model allows for this number to be easily substituted with an appropriate figure given more locally representative information.

Management Implications:

The ultimate goal of connecting the ecological function of the salt marsh with ecosystem service provision was to create a methodology that could be easily transferred to multiple study sites given sufficient local data. The current model was chosen because it fits these criteria. This study has shown that the model provides a tool with the ability to provide insight into the ecological function that drives the nitrogen cycling services of

Table 3.3: Published values of nutrient cycling in saltwater wetlands. *Modified from GecoServ (2011).*

Author	Method	Value	Units
Badola & Hussain (2003)	Replacement Cost	$1,011	US$ 2008 ha^{-1}
Bystrom (2000)	Replacement Cost	$8,767	US$ 2008 ha^{-1} yr^{-1}
De Groot (1992)	Replacement Cost/Benefit Transfer	$3,836	US$ 2008 ha^{-1} yr^{-1}
Ragkos et al. (2006)	Willingness-to-Pay	$60	US$ 2008 person^{-1}
Zhao et al. (2004)	Benefit Transfer	$5,708	US$ 2008 ha^{-1} yr^{-1}
Curtis (2004)	Delphi Panel/Mulit-model Criteria Analysis	$6	US$ 2008 ha^{-1} yr^{-1}

the salt marsh plant, *Spartina*. It also highlights the lack of predictive capablility in nitrogen-uptake processes that could refine the model and ultimately provide more finely scaled information (Anderson et al., 1997; Darby and Turner, 2008a; Tobias, 2009).

In the current management framework, integration of ecological and economic information is typically neither explicit nor mandated. The efforts of this project and others like it are an attempt to create these linkages to help explain the effects of management decisions on both ecosystem and human health (Farber, 2006). The ecosystem service concept provides a means to communicate value in both monetary and nonmonetary terms. Including the metrics of economic value and human well-being can help to forge meaningful dialog and understanding of management decisions that affect the provision of ecosystem services (Farber, 2006; Brauman et al., 2007). This type of understanding facilitates the setting of environmental goals in restoration, conservation, and development that may result in land/water-use changes. Once goals

are set, this framework then allows for measurement of the success or shortcomings of

management decisions (MEA, 2005; Farber, 2006; Brauman et al., 2007). Having the

capability to understand and ultimately predict the possible effects of management

decisions provides, a path forward to more effective use of our shared resources.

Chapter IV:

The value of nitrogen uptake by estuarine oyster reefs:

a scalable, transferable, and dynamic ecosystem services model.

Abstract:

Eastern oysters, *Crassostrea virginica*, provide multiple ecosystem services ranging from food provisioning to water quality improvements. The purpose of this study was to construct a widely transferable oyster model populated with existing data and used for communicating changes in potential nitrogen removal services as a function of alternate management scenarios. Results of the model calculations were similar to previous studies that indirectly measured nitrogen removal rate of a known quantity from the water column. The annual potential nitrogen removal service value of a small sample site (~35ha) near Mustang Island, Texas, U.S.A. approximated $173,000. A dredge scenario was created as a management test case in which it was assumed that a six-month dredging project near the reef would double the total suspended solids load for the duration. Total suspended solids affect the oyster filtration rate of particulate matter from the water and therefore will affect the nitrogen uptake ability of the oyster reef complex. Results indicated that considerable levels of nitrogen removal services could be preserved if the dredging were to take place from October-March rather than the alternate six-month period, April-September.

Introduction:

Eastern oysters, *Crassostrea virginica*, provide many important ecosystem services, those goods and services that benefit human well-being (Costanza et al., 1997;

MEA, 2005). Oyster reefs provide nutrient cycling (Dame et al., 1984; Dame et al., 1989), food, raw materials, and biogenic habitat for commercially- and recreationally-important species (Peterson et al., 2003; Hicks, 2004; Grabowski and Peterson, 2007; Stunz et al., 2010), attenuate the impacts of storm-driven water movements, and stabilize sediments for submerged aquatic vegetation (Meyer et al, 1997; Newell and Koch, 2004). The commercial harvest of oysters along the Texas coast exceeded 5.2 million pounds at a worth of approximately $18.9 million in 2010 (NMFS, 2012). The extent of nitrogen recycling is demonstrated by removal potential of oysters in the Mission-Aransas Bay complex, approximating 12 million pounds and equating to a replacement-cost value of $21 million (Yoskowitz et al. *in review*). Therefore, the value of the intact reef will likely increase when including the benefits received from other ecosystem services provided by oysters.

The role of oyster reefs in nutrient cycling is of increasing importance considering growth of human population along the coastal margins of the U.S. and other countries. As human population increases, there is a concomitant increase in nutrient loading of surrounding nearshore waters. Ultimately, higher nutrients levels could result in eutrophication and, at higher magnitude, create "dead-zones" (as describe by Newell et al., 2005). Oysters remove organic nitrogen from the water column, which is partially assimilated into soft-body tissue or released into the water column as inorganic ammonium. Ammonium is rapidly taken up by phytoplankton and submerged aquatic vegetation. The standing biomass of the oyster reef represents a nitrogen sink for the estuarine environment (Dame, 1984).

Assessments of ecosystem services are undertaken to communicate, in common terms (U.S. dollars), the value of those ecological functions that are performed of a particular habitat and the ability of that habitat to improve human well-being (MEA, 2005; NRC, 2005; Farber et al., 2006). The goal of this research was to develop a nitrogen uptake model of an oyster complex capable of: 1) being readily transferred to other systems, 2) being populated with readily-available and attainable data, and 3) able to provide insight into various management scenarios and their impact on the ecological functions of a specific oyster complex, and finally 4) express changes in ecological function in economic terms using replacement cost methodology.

Methods:

Oyster reef characterization:

Mustang Island is a high profile barrier island (Fig. 4.1) that separates the Corpus Christi Bay system from the Gulf of Mexico (Simms et al., 2006). The Corpus Christi Bay system is large (497 km^2 at mean low water) and includes Redfish, Nueces, Corpus Christi, and Oso Bays (NOAA, 1990). The average depth of the Corpus Christi Bay system is 3.0m with very low inflow (34 m^{-3}s^{-1} on average), and a 32-year mean (±S.E.) salinity of 30.1 (±0.05) with a range from 11.9 - 59 (USEPA, 1999).

The oyster reef complex within the study site covers ~35 ha and is bordered to the North by the Piper Channel (Fig. 4.1). Samples taken from the reef over a three-month period (April 2011 - July 2011) were coupled with long-term data (January 1986 - December 2008) gathered by the Coastal Fisheries Division of Texas Parks and Wildlife
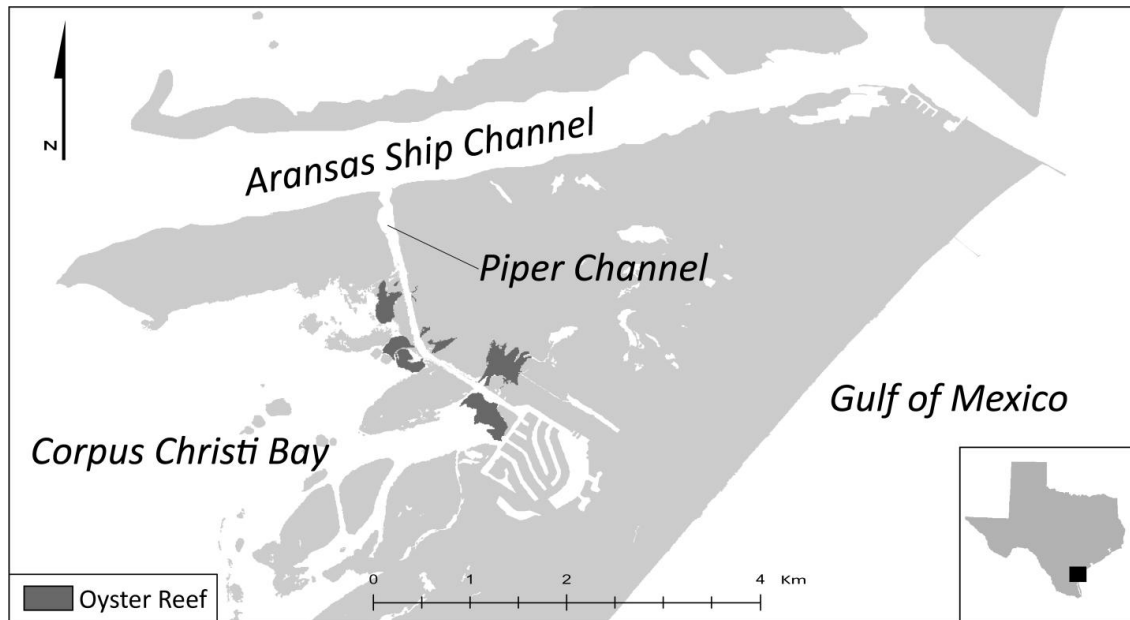
Figure 4.1: Map of the Mustang Island study site.
Nueces Co., Texas, USA: 27.4 N 97.8 W.

Department to characterize standing-stock biomass, areal coverage, density, live:dead

ratio, and average oyster size of the complex (Table 4.1). The areal coverage of the

oyster reef complex was delineated using a differentially corrected hand-held GPS

receiver (Trimble GeoXT). Density and live:dead ratio of adult oysters (>25mm shell

length) were determined by analyzing $0.25m^2$ quadrats randomly sampled along

randomly selected transects throughout the reef. All shells above dark-colored "anoxic"

sediments were removed from each quadrat. Shells were counted only if both portions

of the shell were present. It was also noted whether the shell contained a living oyster

or not. These data were used to calculate the ratio of live to dead oysters on a per unit

area basis.

Table 4.1: Physical Parameters of the oyster reef study site.

Parameter	Value
Areal extent	34.45(Ha)
Live:Dead Ratio	0.43(#live:#dead)
Density of Live oyster	11.38(# m^{-2})
Estimated No. in Area	3,918,692(Total #)
Biomass	1,621,077(gDWT)

Live oysters collected from each quadrat were examined for shell length and biomass parameters (wet weight and dry weight). These data were used to create allometric relationships (Table 4.2) for both adult oysters and spat and expressed as a power function of the general form (Dame, 1972; Dame, 1976):

$$L=aW^b \qquad (1)$$

where *L* is shell length (mm) and *W* is an expression of either wet or dry meat weight (g) (Figs 4.2 & 4.3). Both parameters *a* and *b* were derived through nonlinear regression fitting of field data using JMP version 9.0 statistical software (SAS Institute Inc.: 100 SAS Campus Dr., Cary, NC, 27513, USA).

Model Development:

Simulation of potential nitrogen removal by estuarine oyster reefs was conducted using Simile v.5.7 modeling software (Simulistics Ltd.: 2B Penland Park, Loanhead, Midlothian, United Kingdom, EH20 9PA) in which the modeling environment was based on a "declarative approach:" each visual component of the model was a

functional unit rather than a line of code needed to fulfill further model functions. This

allowed for description and implementation of the model to be equivalent as well as

Table 4.2: Allometric relationship between shell length (mm) of *C. virginica* and soft-bodied mass where WWT and DWT are wet-weight and dry-weight (g), respectively.

a	Size Class	Site	Source
$L=10^{1.582}W_{WWT}^{0.330}$	Adult (>25mm)	Mission-Aransas Estuary, TX, USA	Pollack et al. (2011)
$L=10^{1.846}W_{DWT}^{0.258}$	Adult (>25mm)		
$L=10^{1.67}W_{WWT}^{0.37}$	Spat (<25mm)	Mustang Island, Texas, USA	*This study*
$L=10^{1.93}W_{DWT}^{0.38}$	Adult (>25mm)		

 remain flexible enough for model components to "stand alone" (Simulistics Ltd., 2012).

Development of the model was structured around four main components:

hydrodynamic inputs (temperature, salinity, total suspended solids, and chlorophyll *a*

concentration), standing-stock biomass of the oyster reef complex, filtration potential of

the oyster reef complex, nitrogen uptake level and replacement cost. All environmental

parameters were obtained from the Mission-Aransas National Estuarine Research

Reserve accessed through the National Estuarine Research Reserve System Centralized

Data Management Office (NOAA, 2011). This data collection site, Mission-Aransas

National Estuarine Research Reserve Station #5 (27° 50' 17" N, 97° 3' 1" W), was the

closest in proximity to the study site (<1km). All data were averaged over a monthly

interval from January, 2007 - December, 2008, to populate the model. Standing stock

biomass was first derived as a function of mass per unit area (g m^{-2}) and converted to

energetic units (kcal m^{-2}) using a conversion factor of 5.07kcal g^{-1} (Dame, 1972; Dame, 1976). Standing stock biomass followed the general form:

$$BM=B-D+P-R \qquad (2)$$

where *BM* is total biomass (kcal), *B* is number of spat per oyster (kcal of spat individual^{-1} adult oyster), *D* is loss of oysters either through predation, disease, or death (% dead month^{-1}), *P* is oyster growth (kcal), and *R* is respiration (kcal). Spat per oyster (B) and loss (D) were determined using data collected from the 32-year oyster sampling program from Texas Parks and Wildlife Department. Monthly averages were calculated from Texas Parks and Wildlife Department data for the Mission-Aransas Bay system and then applied to the model as a function of biomass gain or loss per individual.

Respiration was empirically derived (Dame, 1972; Dame, 1976) and represents all metabolic processes included in oxygen consumption and waste excretion:

$$R=(242.38+43.82T)W^{0.75} \qquad (3)$$

where *R* is respiration (kcal), *T* is temperature (°C), and *W* is dry weight (g). The production term in Eq. 2 represents the addition of biomass as a function of growth and reproduction and was derived from the general equation of production for non-insect invertebrates (Humphreys, 1979):

$$P=1.033R+0.3757 \qquad (4)$$

where P is equal to production (kcal) and R is respiration (kcal).

The aforementioned model was run in monthly time-steps and described the fluctuation in biomass as a function of season and change in water quality parameters. Uptake of nitrogen for this specific oyster reef complex system was not typically measured-- therefore it was necessary to convert Chlorophyll a concentrations into nitrogen units, more specifically, particulate organic nitrogen (PON). It was assumed that phytoplankton were the sole food source (i.e., Chlorophyll a concentration was a proxy for phytoplankton biomass). Using the Redfield (1963) ratio, Chlorophyll a was converted to carbon concentration at a rate of 1:50. This was then converted to nitrogen at a rate of 106:16:1 (carbon:nitrogen:phosphorous) based on molarity.

Filtration rate of the oyster reef complex was empirically derived (Powell et al., 1995) using the general form:

$$FR=(L^{0.96}T^{0.95})/2.95 \qquad (5)$$

where FR is equal to the filtration rate (ml ind^{-1} min^{-1}), L is the shell length of the oyster (mm), and T is the temperature (°C). This equation for filtration rate was modified as a function of salinity (Powell et al., 1995):

$$at\ S\geq7.5‰ \qquad FR_S=FR$$

$$at\ 3.5‰<S<7.5‰ \quad FR_S=FR(S-3.5)/4.0 \quad (6)$$

$$at\ S≤3.5‰ \quad\quad FR_S=0$$

where S is salinity (‰) and FR_S is the salinity adjusted filtration rate (ml ind^{-1} min^{-1}).

Filtration rate was also modified by total suspended solids (Powell et al., 1995):

$$FR_{TSS}=FR_S[1-0.01(log_{10}TSS+3.38/0.0418)] \quad\quad (7)$$

where FR_{TSS} is the filtration rate adjusted for total suspended solids (ml ind^{-1} min^{-1}), FR_S

is the salinity adjusted filtration rate (ml ind^{-1} min^{-1}), and TSS is the measurement of

total suspended solids (mg L^{-1}).

Nitrogen uptake is a function of both clearance rate and assimilation efficiency.

Assimilation efficiency is assumed to be 75% of the PON that passes through an

individual oyster based on clearance rate which is calculated from Powell et al. (1995)

as:

$$CR=f*FR_{TSS} \quad\quad (8)$$

where CR is the clearance rate (mg ind^{-1} min^{-1}), f is the measured PON (mg ml^{-1}) in the

water column, and FR_{TSS} is the filtration rate (ml ind^{-1} min^{-1}) adjusted for TSS. This

resulted in an estimated rate of PON removal by each individual oyster. The product of

the estimated total number of individual oysters in the oyster reef complex and the

replacement cost price of \$4.90 kg^{-1} of PON removed per oyster was used to calculate

the monthly average value of nitrogen removal. The above price represents the

replacement cost needed to construct a secondary treatment facility designed to

remove organic nitrogen from the water column (Smith, 2008). Total annual nitrogen

removal is calculated by summing the amount of PON removed from the system at each

monthly time-step.

Sensitivity analysis:

Sensitivity of the model output, value of nitrogen removed by the oyster reef

complex, was tested by systematically varying the level of each hydrodynamic input.

The main hydrodynamic inputs include TSS, Chlorophyll *a* concentration, and

temperature. These parameters were chosen for model construction due to the large

amount of data available. Each parameter was varied by the baseline input multiplied

by ±0.25 and ±0.50 for a total of four model output variations for each parameter.

Salinity was not added to the analysis due to the relatively high values measured in this

area. The depression of filtration was only measured in the model below 7.5‰ (Powell

et al., 1995); therefore, it was not sensitive enough to detect salinity fluctuations

captured as part of this analysis.

Management Scenario - Dredging:

Construction of the Gulf Intracoastal Waterway (GIWW) began in the early

1900's and continued for several decades after (Alperin, 1983). The final connection

between Redfish Bay and Corpus Christi Bay, to include the previously constructed

Aransas Ship Channel was completed in 1959 (Odum, 1963). Construction of the GIWW

has been an economic driver for much of the Texas and Louisiana Coastal regions; however, dredging is still required to keep the channel open. Dredging activity along the GIWW has been shown to increase the amount of TSS in the water column and even adversely affect the productivity of surrounding habitats (Odum, 1963). It was assumed that dredging along the Piper Channel (Fig.4.1), which borders the current study site, would have similar effects along the habitats that border it. Therefore, the current oyster model was used to evaluate the assumed affects of maintenance dredging near the oyster reefs of northern Mustang Island.

A dredging scenario was developed to demonstrate the ability of the model to determine change in nitrogen removal benefits. It was assumed that dredging near the oyster reef would double the natural baseline TSS for a period of six months. Two different six-month model scenarios were developed, April-September and October-March. It was assumed that TSS would double as a result of resuspended and newly deposited sediment (Mallin et al, 2003; Je et al., 2007), whereas remaining hydrodynamic input parameters would be unaffected.

Results:

Oyster Reef Characterization:

The oyster reef within the study site was delineated within the model based on the collected data from Table 4.1 . The allometric relationships that predict the length (mm) of sampled oysters as a function of dry weight (Fig 4.2) for adults and wet weight (Fig 4.3) for spat are shown graphically.
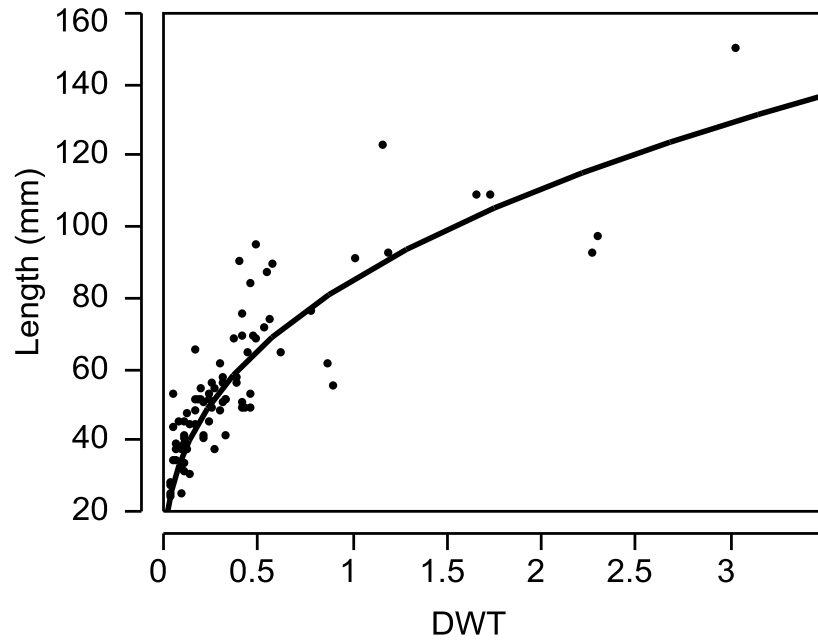
Figure 4.2: Scatter plot of length (mm) and dry weight (g) of adult oysters (n=85) sampled from the study site. The line represents the predicted length (mm) calculated as $L=10^{1.93}W_{DWT}^{0.38}$.
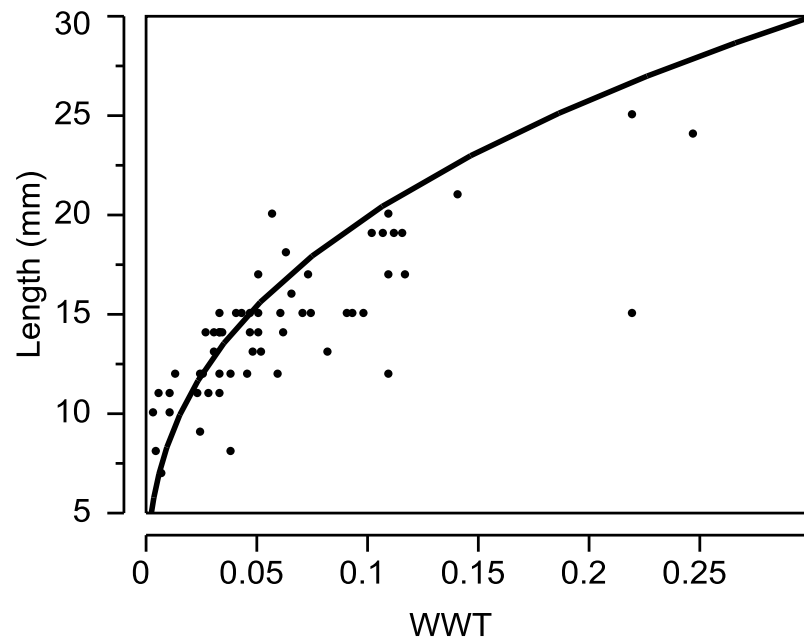
Figure 4.3: Scatter plot of length (mm) and wet weight (g) of oyster spat (n=62) sampled from the study site. The line represents the predicted length (mm) calculated as $L=10^{1.67}W_{WWT}^{0.37}$.

Baseline Model Simulation:

The base model simulation calculated a mean value of nitrogen removal of $0.33 min^{-1} ±0.01 (S.E.) over a 120 month period for the entire area of reef. Over a four-year period, values ranged from $0.16 min^{-1} to $0.80 min^{-1} (Fig 4.4). This was based on monthly mean values over an entire "average" year. Scaling up, the largest monthly value of nitrogen removal was calculated at $34,452 month^{-1}, whereas the smallest was $6,843 month^{-1}. The calculated annual value based on the aggregated monthly means was $173,370 year^{-1}.
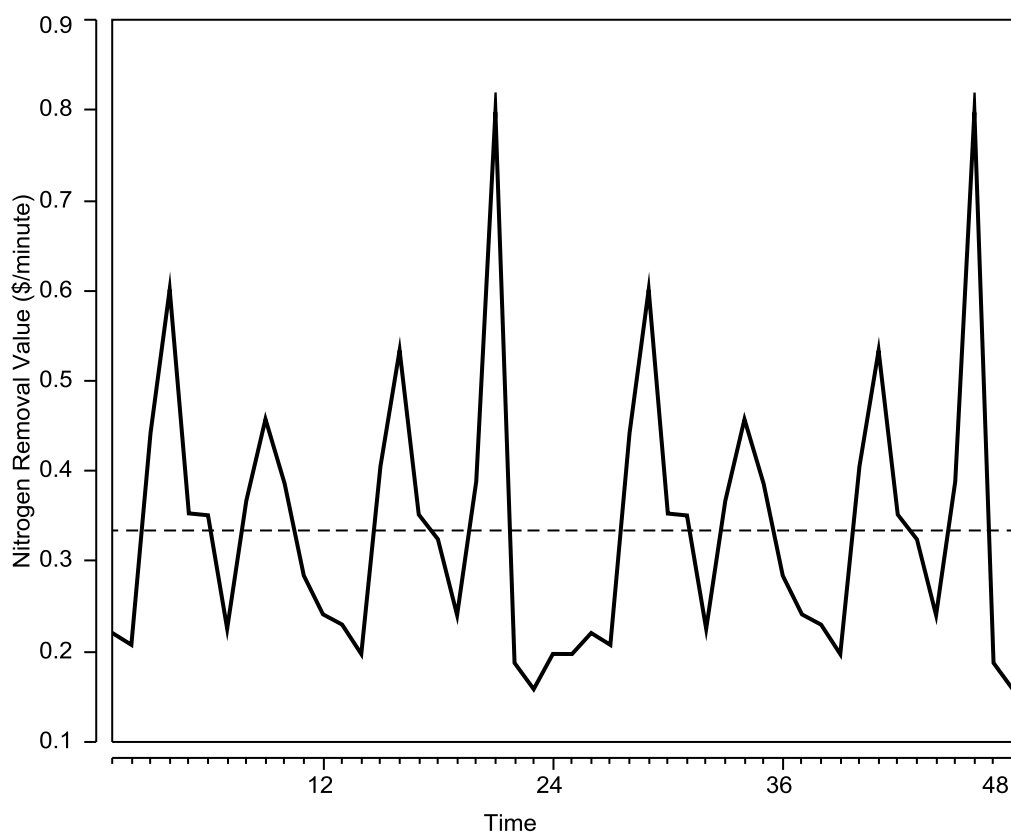
Figure 4.4: Calculated nitrogen removal value in the oyster reef study site (---‾=$0.33). Mustang Island, Nueces Co., Texas.

Sensitivity analysis:

Variation of model inputs by ±25% and ±50% of the baseline showed that the value of nitrogen removed from the system was most sensitive to changes in temperature and Chlorophyll *a* concentration (Table 4.3). It is likely that temperature will also have an effect on Chlorophyll a concentration, but this link is not expressed within the model framework.

Table 4.3: Model sensitivity analysis showing the manipulated level of each input and the resulting change in mean value of the nitrogen removal service.

Variable	Test Value (% of Baseline)	Nitrogen removal rate ($ min^{-1})	% Change	Mean annual value ($US2008)	% Change
Baseline	100	$0.3334	0.0	$173,370.87	0.0
Chlorophyll *a* (μg L^{-1})					
	150	$0.5001	50.0	$262,894.45	50.0
	125	$0.4168	25.0	$219,076.11	25.0
	75	$0.2500	-25.0	$131,443.99	-25.0
	50	$0.1667	-50.0	$87,635.88	-50.0
Temperature (°C)					
	150	$0.4901	47.0	$257,614.92	47.0
	125	$0.4121	23.6	$216,648.38	23.6
	75	$0.2537	-23.9	$133,353.99	-23.9
	50	$0.1726	-48.2	$90,721.27	-48.2
Total Suspended Solids (NTU)					
	150	$0.3078	-7.7	$161,802.33	-7.7
	125	$0.3193	-4.2	$167,858.01	-4.2
	75	$0.3516	5.5	$184,809.97	5.5
	50	$0.3772	13.1	$198,265.50	13.1

Management Scenario - Dredging:

The initial baseline output of the model calculated a yearly benefit of $173,370 for an "average" year. During the April-September scenario, nitrogen removal benefits were diminished by $12,481, whereas during October-February benefits were reduced to only $8,708 (Table 4.4).

Table 4.4: Dredge Scenario and the resulting calculated potential nitrogen removal value for both scenarios: May-October and October-March using the oyster reef study site as the subject.

	Baseline	Dredge April - September	Dredge October - March
Replacement Cost Value	$173,330	$160,889	$164,662
Change in Value	$0	$12.481	$8,708
Percent Change	0%	-7.2%	-5.0%

Discussion:

The oyster nitrogen removal benefits model presented in this paper were designed to estimate the relationship between the ecological function of water filtration by estuarine oyster reefs and the ecosystem service of nutrient cycling. The potential amount of nitrogen removed from the system was valued using replacement cost methodologies. Ultimately, the model was designed to be transferrable among systems with minimal inputs of data with respect to oyster reef characterization and readily-collected water quality parameters. Results of the sensitivity analysis elucidated the forcing factors in the model allowing better understanding of the components of the natural system that affect the ability of the oyster reef to remove nitrogen. Once the

model was populated, a platform was then available to ask relevant management

questions about the oyster reef system being modeled.

Sensitivity analysis:

Hydrodynamic inputs (e.g., water temperature, Chlorophyll *a* concentration, and

TSS) were the primary inputs into the model. Each input is independently modified in

order to understand the ability of each to drive the output of the model. The relative

behavior of the model, subject to individual parameter modification, allows the user to

develop an intuitive understanding of what the model is capable of doing as well as its

limitations. It is also important to note that the independent nature of the

hydrodynamic inputs allows the user to create simulations that represent extreme

conditions such as storms or other anomalous events. For example, an increase in

Chlorophyll *a* concentration does not trigger an automatic increase in TSS within the

model. Therefore, it is necessary to use sound professional judgment when

manipulating the model inputs to define a management scenario that is reasonable.

Water temperature had the largest impact on the final valuation output. Much

of this was due to the effect of water temperature on ecological function of the oyster

reef (e.g., filtration rate, respiration, production) (Dame, 1972; Humphreys, 1979). Few

management scenarios chronically increase the temperature of the water column. An

exception could be the warm water outfall from power generating plants. Chlorophyll *a*

concentration has a direct 1:1 relationship with nitrogen removal services due to its

function in the model as a proxy measurement for phytoplankton abundance and

ultimately PON (Fritz et al., 1984; Powell et al., 1995; Newell et al., 2005; Coen et al.,

2007). Because temperature directly affects metabolic rate of the oyster, it was used to calculate respiration and ultimately production (Dame, 1972; Dame, 1976; Humphreys, 1979). As Chlorophyll *a* concentration served as a proxy for PON in the system, it therefore controls nitrogen removal by its relative availability to the oysters in the system (Powell et al., 1995). Therefore, a decrease in Chlorophyll *a* directly affects the ability of the oyster reef to remove it from the water column. This fact points out a limitation to the model. The input of Chlorophyll *a* into the model is not restricted and therefore as the model is constructed, a limitless amount of Chlorophyll *a* is able to be removed from the system by the oyster reef. There is not an explicitly defined connection among the hydrodynamic inputs that explain the relative behavior or interconnected nature of each. Future iterations of the model could be improved by coupling data in a manner that will define the interconnection of the hydrodynamic inputs. The addition of a population model for phytoplankton may also increase the ability of the model to predict nitrogen removal based on phytoplankton biomass.

Total Suspended Solids in the system also affect the ability of the oyster reef to successfully remove nitrogen from the water column (Gerritsen et al., 1994; Mann, 2000; Nelson et al., 2004). In the model, the composition of the TSS is not clearly defined. Phytoplankton, the assumed sole feed source, would be considered part of the TSS which has an inhibitory affect within the model (Powell et al., 1995). Organic solids such as phytoplankton are necessary to feed the oysters, but inorganic solids such as sediments will have an inhibitory effect on filtration (i.e., filtration is a mechanical process and within the model framework, it does not necessarily differentiate among

particles while in reality, the oyster is able to exclude/include specific particle types). It

is obvious that increasing phytoplankton concentrations around the oyster reef would

be beneficial, but further definition of the make-up of the TSS would help to further

define the limits of the model.

Transferability:

Although one of the goals in building the nitrogen removal model was to create

something that is widely transferrable, it is necessary to temper its output and

subsequent valuation relative to alternate scenarios. The goal of ecosystem service

valuation, in the context of this paper, was to provide a vehicle of communication in

units—U.S. Dollars—that are common among many different types of stakeholders,

both familiar and unfamiliar with the technical aspects of nutrient cycling and ecological

functioning of oyster reefs (Costanza et al., 1997; Farber et al., 2006). The model output

itself is typically most useful when a baseline is created from valid historical data and

then various management scenarios are applied as a means of direct comparison to the

resulting output. This valuation is considered to be a proxy measurement of the

ecological function of the oyster reef with respect to nitrogen cycling and water

filtration. Nitrogen uptake data by oyster reefs is not commonly available, therefore

calibration of the model is difficult. Dame and Libes (1993) reported the Chlorophyll *a*

uptake of an intertidal oyster reef in the North Inlet-Winyah Bay National Estuarine

Research Reserve, South Carolina, USA (NIWBNERR) during several summer months to

average 7.7×10^{-5}g Chl *a* oyster^{-1} hour^{-1}. In that case, hydrodynamic data—Chlorophyll *a*,

water temperature, and salinity—were obtained from the NIWBNERR at the same

location where the study was undertaken (NOAA, 2011). These data were used to

populate the model along with published oyster reef characteristics. The resulting

potential uptake of nitrogen by oysters was estimated to be 6.95×10^{-5} g Chl a oyster^{-1}

hour^{-1} (Dames and Libes, 1993). The nature of these numbers do not allow for

statistically rigorous comparison, but the fact that both are within the same degree of

magnitude and less than one integer in difference leads us to believe that they are

comparable in the context of this model and its output.

Management Scenario – Dredging:

Dredging projects are often valued in the tens of millions of dollars and are

crucial to the economic stability of much of the Gulf Coast (Alperin, 1983; USACE, 2011).

Therefore, it is important to note that this management scenario is solely based on the

time of dredging. The seasonal fluctuation of filtration rate, based on the components

of the model, implies that setting the dredging schedule with ecosystem services in

mind may decrease the loss of potential nitrogen uptake services. The modeled

decrease in benefits may not result in the cancellation of a dredging project, nor is this

the goal. However, it could help to schedule the project in a manner that optimizes the

benefits received from the oyster reef. Beyond dredging, the model was able to address

questions appropriate to other management scenarios. The two main categories of

inputs into the model used to evaluate management scenarios are the physical

characteristics of the oyster reef and the hydrodynamic patterns surrounding reefs.

Management scenarios that have an effect on oyster density, areal extent, average

oyster size, live:dead ratio, water temperature, salinity, TSS, and Chlorophyll a

concentration can all be individually evaluated in the model framework. Each of these parameters represents a variable in the construction of the model that can be manipulated. The manipulation of each of these parameters will have an effect, positive or negative, on the final valuation output of the model.

In conclusion, this paper has estimated a value associated with nitrogen removal that is reasonably, but not completely accurate. The estimation of value should be tempered by the knowledge that (1) oyster reefs provide several services above and beyond nutrient cycling, (2) a model designed to be highly transferrable cannot be highly precise, and (3) replacement cost is a term that implies a service can be replicated. Eutrophication of a system cannot be easily reversed. It is not as simple as building a waste-water treatment plant to remove nitrogen from a system that was once removed by an oyster reef. Habitat degradation and/or human activity that results in a loss of oysters will likely create a situation where replacement of the habitat will be much more costly than the value of an individual service alone. Overall, the value of the oyster reef is most likely much higher than the calculated values mentioned above based solely on the fact that oysters provide many other ecosystem services such as habitat, food and raw materials (Peterson et al., 2003; Hicks, 2004; Grabowski and Peterson, 2007; Stunz et al., 2010). It is with this knowledge that the output of this model is to be used as a communication tool to allow meaningful discussion and insight into management scenarios that might affect the ability of an oyster reef to provide those valuable nitrogen removal services that directly benefit human well-being.

Chapter V:

Relative production value of major estuarine habitats of recreational

fishing services along Mustang Island, Texas, USA

Abstract:

Recreational fishing is an important ecosystem service and economic driver in many coastal communities. The major estuarine habitat types investigated as part of this study, marsh edge, seagrass, and oyster reefs, have the ability to provide recreational fishing opportunities in the form of habitat and refuge for game fish and other species. Combining socioeconomic data with habitat use and areal coverage showed that the annual economic value of recreational fishing within the Mustang Island study site was $83,664,532: marsh edge, seagrass and oyster reef at $2,032,201, $81,550,847, and $81,484, respectively. These values were further differentiated by angler preference to delineate the relative annual value of each recreationally important species: red drum – *Sciaenops ocellatus* ($33,465,813), spotted seatrout – *Cynoscion nebulosus* ($16,732,906), and southern flounder - *Paralichthys lethostigma* ($6,693,163). These are considered conservative estimates of the recreational fishing services provided by these habitats in that they were based solely on travel expenditures and license sales. The goal of this study was to provide a framework for derivation of habitat-based values capable of being implemented in a variety of coastal communities. Further understanding of how each habitat is used by the angling public will help refine this model in order to communicate the relative value of each habitat in the provision of recreational fishing services.

Introduction:

The value of recreational fishing as an ecosystem service is determined by two distinct yet associated components: the human preferences that control how money is spent for recreational fishing experiences and the biological ability of a given habitat type to support and provide stocks of targeted species of fish necessary for this activity. In Texas, it is estimated that 6 million resident and non-residents participated in at least one form of wildlife recreation (e.g., birding, hunting, fishing, etc. (Southwick, 2006). These activities produce an estimated $8.91 billion in annual retail sales ($981 million attributed to saltwater fishing) and support approximately 139,404 jobs throughout the state (Southwick, 2006). These activities are unique in that they represent an export of economic activity from urban centers to smaller, rural economies that depend upon the influx of tourist dollars (Southwick, 2006). It is likely that the true value of these resources is even more based on time spent fishing, travel costs, license sales and other expenses not include in the figures above.

Apart from expenditure of money to engage in fishing, the angling public has specific attitudes pertinent to the recreational fishing resource and its management. Most anglers along the Texas coast support saltwater fisheries management tools such as minimum size limits, stock enhancement, and daily bag limits (Ditton and Hunt, 1996). These anglers are also heavily invested in this activity as they are not willing to substitute fishing for other activities (Ditton and Sutton, 2004). It is these types of attitudes and preferences that determine how money is allocated with respect to recreational fishing and its management.

The production of fisheries resources is dependent upon the habitats that support them. Habitat-fisheries linkages and the way in which the National Resource Damage Assessment (NRDA) views them emphasizes the importance of habitat type to the production of fisheries resources (Strange et al, 2004; Parsons and Kang, 2010). The NRDA process has only recently favored habitat replacement resulting from fisheries damages versus the direct one-to-one replacement of individual fish (Bell, 1997; Flores and Thatcher, 2002). To an extent, this recognizes multiple ecosystem services provided by "healthy" fisheries as an extension of the habitats incapable of being replaced by solely substituting one portion of the pair (Strange et al, 2004; Parsons and Kang, 2010). This position is also supported by fisheries data that helps to explain the relative value of various habitat types to the production of fisheries resouces(Boesch and Turner, 1984, Rooker et al., 1998; Stunz et al., 2002; Stunz et al., 2010).

Certainly, difficulties can arise in terms of understanding the relationship between the preferences of the angling public to the ability of the major estuarine habitats to provide recreational fishing opportunities. Several studies have sought to place an economic value on recreational fishing, yet few explicitly address the aforementioned relationship. Bell (1997) incorporated a production model approach in which the marginal value of a single hectare of salt marsh along the Gulf coast was valued based on fish caught there , and showed a singular hectare of salt marsh had a recreational fishing value of $19,300 yr^{-1}.

The objective of this study is to identify and explore methods to delineate recreational fishing value in a manner that not only captures the preferences of the

angling public, but also the productive capacity of major estuarine habitat types. It is

also seeks to develop a framework for this type of analysis at local scales. This will be

accomplished by combining existing socioeconomic data for recreational fisheries,

habitat use data for major estuarine habitats, angler preference for recreationally

important fish species, and fine-scale areal coverage data to define the relative

productive value of each chosen estuarine habitat type for the provision of recreational

fishing services.

Methods:

Study site:

Mustang Island is a high profile barrier island that separates the Corpus Christi

Bay system from the Gulf of Mexico (Simms et al., 2006). The Corpus Christi Bay system

is large (497 km^2 at mean low water) and includes Redfish, Nueces, Corpus Christi, and

Oso Bays (NOAA, 1990). The average depth of the Corpus Christi Bay system is 3.0m

with very low inflow (34 m^{-3} s^{-1} on average), and a 32-year mean (±S.E.) salinity of 30.1

(±0.05) with a range from 11.9 to 59.0 (USEPA, 1999). The three major estuarine

habitats (Figs. 5.1a-c) cover approximately 4,198 hectares: marsh edge (ME), seagrass

(SG), and oyster reef (OYS). The contribution of these major habitat types is based on

economic data, creel surveys, relative abundance of fish per habitat, and areal coverage

of each habitat within the study site (Table 5.1).

A)

B)

C)

Figure 5.1: Map of study site (Mustang Island, Nueces County, TX, USA: 27.4 N 97.8 W) and habitats: A) Marsh Edge , B) Seagrass, and C) Oyster
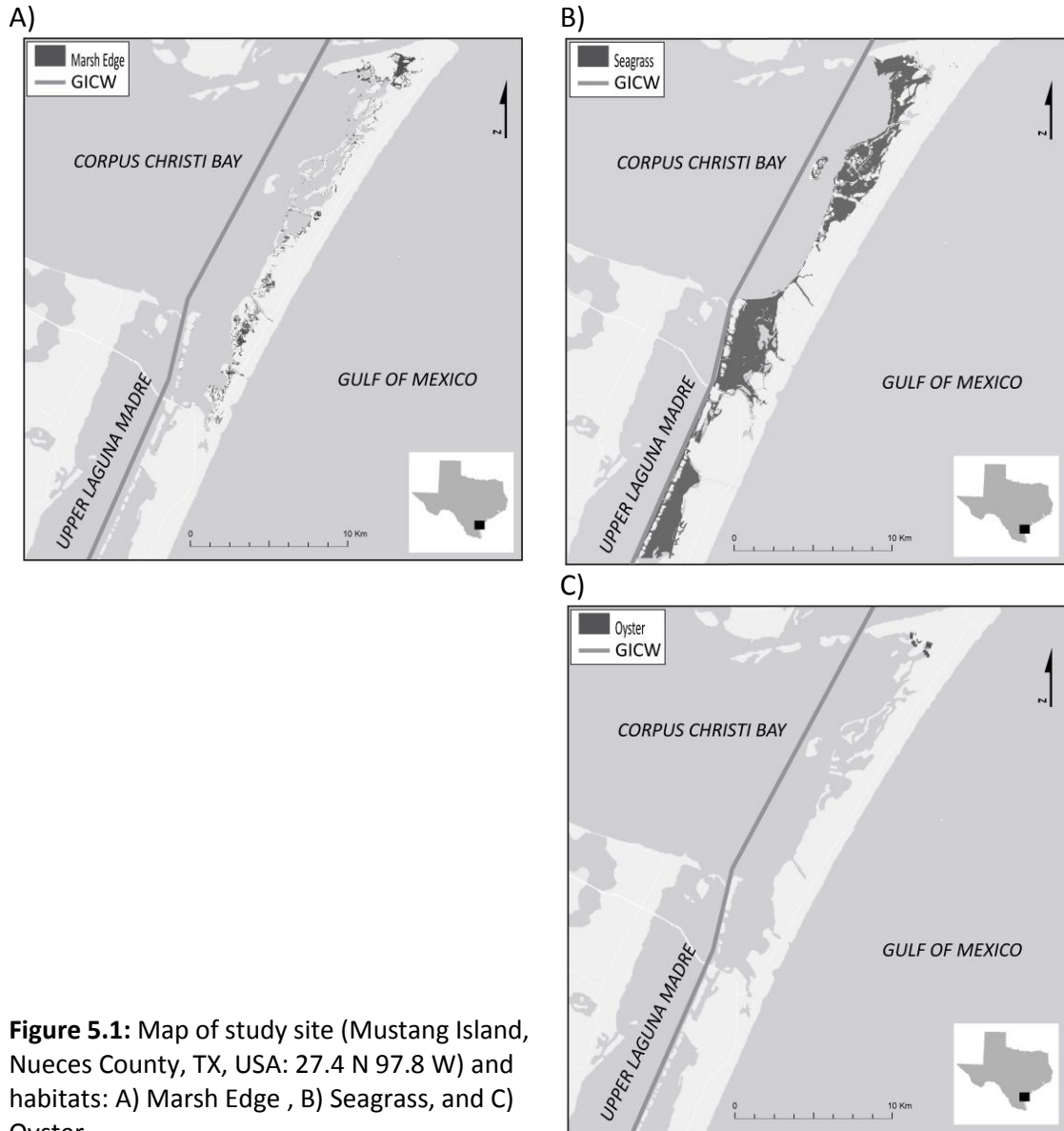
According to the U.S. Census Bureau, the 2010 population of Nueces County was 340,223 with a density of 406 people per square mile. The median household annual income was $43,280. These figures are generally below the median income for both the state of Texas and the nation. Population density is relatively high as compared to the rest of the U.S., mainly due to the Corpus Christi metropolitan area (US Census, 2010).

Table 5.1: Model input parameters

Parameter	Variable	Value	Citation
Travel expenditures for recreational fishing value on Texas coast	TE	$466,644,712	Southwick, 2006
Fishing License Sales 2011 (Nueces, Aransas, San Patricio, Kleberg, and Jim Wells Counties)	LS	$5,268,220	TPWD, *Unpublished*
Percent Effort of fishing in bays adjacent to study site (Corpus Christi Bay and Upper Laguna Madre)	%Eff	%16.8	Green and Campbell, 2010
Areal Coverage of habitats	AC	ME: 1,060,916 m^{-2} SG: 40,570,356 m^{-2} OYS: 344,546 m^{-2}	Gibeaut et al., 2010
Percentage of survey respondents and their preferred target species	%Pref	Red drum: 40% Spotted seatrout: 20% Southern Flounder: 8%	Tseng et al., 2006
Mean Density of Red drum	MD	ME: 0.0825 m^{-2} SG: 0.14 m^{-2} OYS: 0.01 m^{-2}	Stunz et al., 2002; Stunz et al., 2010
Mean Density of Spotted Seatrout	MD	ME: 0.048 m^{-2} SG: 0.0025 m^{-2} OYS: 0.01 m^{-2}	Naehr et al., 2010; Stunz et al, 2010
Mean Density of Southern Flounder	MD	ME: 0.0315 m^{-2} SG: 0.0275 m^{-2} OYS: 0.0 m^{-2}	Nanez-James et al., 2009; *No data for OYS*

Recreational Fisheries Preferences:

This study focused on three main recreationally important fish species: red

drum (*Sciaenops ocellatus*), speckled trout (*Cynoscion nebulosus*), and southern

flounder (*Paralichthys lethostigma*). These were chosen from the literature based on

relative economic importance and sufficient data within the sport fishing literature. In

2006, travel expenditures for recreational fishing opportunities associated with these

species were $466,644,712 for the entire Texas coast (Southwick, 2006). A survey study

of saltwater anglers in Texas revealed that 40, 20, and 8% of respondents preferred to

catch red drum, speckled trout, and southern flounder, respectively. The remaining 32%

was distributed among other drum species, red snapper, king mackerel and those

without a preference (Tseng, et al., 2006).

Model development:

The development of a model for recreational fishing value was based on three

main sets of data. The first portion of the model was based on the value of recreational

fishing for the state of Texas which was then parsed out into the value associated with

the specific boundaries of the study site. This represents a revealed-preference

approach that captures only a partial amount of willingness-to-pay for recreational

fishing experiences. The second portion was the mean density of newly-settled

juveniles (red drum, spotted seatrout, and southern flounder) found in each habitat

type as estimated from the literature (Tseng et al., 2006) . Finally, areal coverage data

for the major estuarine habitats were used to express the total number of recreationally

important fish and relative value of each habitat type to recreational fishing services.

To calculate the relative value of each major estuarine habitat, a Site Specific

Recreational Fishing Value (SSRFV) was first calculated (Table 5.1). This was done using

a combination of data found in the literature based on travel expenditures (TE) ($)

corrected for the percentage of effort (%Eff) spent within the bays adjacent to the study

site plus regional license sales (LS) ($) following the form:

$$SSRFV = (TE)\ (\%Eff) + (LS) \qquad (1)$$

Once the SSRFV was determined, further partitioning of this value was

determined by percentage of fish per habitat (%FPH). This was based on published data

which delineates the mean density (MD) (# m^{-2}) of each recreationally important fish

species per habitat type and the areal coverage (AC) (m^{-2}) of each habitat divided by the

total number of fish (TF) (#) from each habitat type in the general form:

$$\%FPH = [(MD)\ (AC)] \div TF \qquad (2)$$

where mean density of fish by habitat type was estimated from published literature

(Stunz et al., 2002; Nanez-James, 2009; Nearh et al., 2010; Stunz et al., 2010).   This

calculation was performed as a combination of each habitat type and recreationally

important species.

Once the %FPH was determined, the Recreational Fishing Value per Habitat

(RFVPH) was further partitioned by habitat type. This is done by multiplying %FPH and

SSRFV ($):

$$RFVPH = (\%FPH)\ (SSRFV) \qquad (3)$$

The RFVPH was further parsed by incorporating angler preference for the three recreationally important species to calculate Recreational Fishing Value per Species (RFVPS). This was accomplished by multiplying %FPH, the percentage of people preferring one of the given recreationally important species (%Pref), and RFVPH ($) following the general form:

$$RFVPS = (RFVPH)(\%FPH)(\%Pref) \qquad (4)$$

Results:

The SSRFV was calculated to be $83,664,532 yr^{-1} for the Mustang Island study site. The marsh edge habitat type was valued at approximately $2 million, seagrass at $81.5 million, and oyster at $81,000 on an annual basis (Table 5.2). It is important to remember that these values were only determined as a function of license sales and travel expenditures.

Table 5.2: RFVPH attributed to the productive value of each habitat based on the mean density of recreationally important fish species

Habitat	RFVPH ($ yr^{-1})
marsh edge	$2,032,201
seagrass	$81,550,847
oyster reef	$81,484
Total	$83,664,532

The total number of recreationally important fish calculated across all habitats types and species was 7,075,720. The largest number of fish was found in seagrass

(6,896,961), with the greatest number being red drum (5,679,850 individuals) (Table

5.3).   This also resulted in the largest %FPH at 97%. Therefore, the largest percentage

of the habitat value was attributed to seagrass ($81,550,846) (Table 5.2).

Table 5.3: Calculated total number of individuals for each recreationally important species with the corresponding percentage of fish per habitat (%FPH).
Note: No data available for southern flounder on the oyster habitat type

| | Calculated total number of individuals: | | | |
Habitat	Red drum	Spotted Seatrout	Southern Flounder	%FPH
marsh edge	87,525	50,923	33,418	2.4%
seagrass	5,769,850	101,425	1,115,685	97.5%
oyster reef	3,445	3,445	0*	0.1%

The SSRFV was then partitioned on the basis of angler preference for a given

recreationally important species. The RFVPS was approximately $33 million for red

drum, $16 million for spotted seatrout, and $6 million for southern flounder. The sum

of these figures failed to account for approximately $26 million of the total SSRFV of

approximately $83 million. This difference was likely attributed to the undefined

"other" portion of the %Pref. Once again, seagrass was determined the most valuable

habitat type for each recreationally important species (Table 5.4).

Table 5.4: RFVPS for all recreationally important fisheries species used in the current model.

Habitat	RFVPH	RFVPS: *Red drum*	RFVPS: *Spotted Seatrout*	RFVPS: *Southern Flounder*
marsh edge	$2,032,201	$812,881	$406,440	$162,576
seagrass	$81,550,847	$32,650,338	$16,310,169	$6,524,068
oyster reef	$81,484	$32,593	$16,297	$6,519
Total	$83,664,532	$33,465,812	$16,732,906	$6,693,163

Discussion:

The model presented in this study is able to partition recreational fishing value for the state of Texas scaled for a local study site (i.e., Mustang Island). Results from this study generally agree with published data that show that structured, vegetated habitats are important to the recreationally important fish species studied (Stunz et al., 2002; Nanez-James et al., 2009; Neahr et al, 2010; Stunz et al., 2010). The model is also able to partition values based on preferences expressed by the angling public (Tseng et al., 2006). Understanding more closely the interplay between preferences of the angling public and the ability of each habitat to support recreationally important species allows for opportunities to better use these resources. The data used in the model can also be refined and expanded to make the model more descriptive.

The ability of the current model to adequately partition recreational fishing value as a function of habitat was primarily based on the relative density of recreationally important fish species. The current model uses those densities from published studies along the Texas coast. Stunz et al. (2002) compared the density of juvenile red drum in Galveston Bay using both epibenthic sleds and enclosure samplers. Their study sampled areas within the Galveston Bay complex with and without seagrass. It was suggested that in the absence of seagrass that marsh edge became an important habitat for newly settled red drum. There was also an effort made to collect red drum from the extensive oyster reefs in Galveston Bay, but none were found. Therefore, the mean density of red drum associated with oyster reef for the present study was based on a later study by Stunz et al. (2010), also in Galveston Bay. This study delineated the relative importance

of oyster reef as structurally complex habitat that was previously more difficult to sample and often omitted from habitat use studies.

Neahr et al. (2010) examined the mean density of newly settled spotted seatrout throughout estuaries along the Texas coast using both long-term data collected by Texas Parks and Wildlife as well as sampling by epibenthic sled. According to maps published by the study, three sites possibly overlap with the modeled study site. Other samples were taken in Aransas Bay and Upper Laguna Madre, systems in close approximated to the present study site. It was found that newly settled spotted seatrout preferred vegetated, structurally complex habitats. Again, this study did not sample oyster reefs in the area. Mean density of spotted seatrout associated with oyster reef used in the present study was based on that of Stunz et al. (2010) from the Galveston Bay.

In 2010, Nanez-James et al. examined the mean density of newly settled southern flounder in the Aransas-Copano bay system. The results of this study also highlighted the importance of vegetated, structured habitats. This study also examined the importance of proximity to Gulf passes and found that the closer to a pass any given habitat is, the more suitable it is for newly settled southern flounder. These samples were taken using a beam trawl and therefore were not used to sample oyster reefs in the area. Stunz et al. (2010) sampled oyster reefs in Galveston Bay, but no southern flounder was sampled. Therefore, no comparable data for mean densities of southern flounder is known. In current model, it is assumed to be zero.

In light of the mean densities used in development of the model, the RFVPH was likely representative of a relatively high value placed on seagrass and marsh edge

habitat types which are reported to be preferred by the recreationally important species used in the present study (Stunz et al., 2002; Nanez-James et al., 2009; Neahr et al, 2010; Stunz et al., 2010). The RFVPH calculated by the model (~$83 million yr^{-1}) was also a conservative estimate considering it is based solely on data collected for travel expenditures and license sales related to recreational fishing. In Texas, saltwater fishing produced an estimated $8.91 billion in retail sales alone (Southwick, 2006). Even though the general trend of preferred habitats by recreationally important species is preserved, it is likely that the oyster habitat type actually has a higher value simply based on the missing data with respect to southern flounder as well as the relative difficulty in sampling this habitat type (Stunz et al., 2010).

The calculation of RFVPS is useful in that it addresses the interplay between angler preference and biological productivity. It holds to reason that red drum, being a more preferred target by the angling public, had a higher value followed by spotted seatrout and southern flounder, respectively. Perhaps the most interesting aspect of development of the model was the ~ $26.7 million yr^{-1} not explained by the current model. A portion of this value could be associated with other species such as black drum (*Pogonias cromis*) and Atlantic croaker (*Micropogonias undulatus*). The portion of value that was not identified by the current model was directly related to motivation for fishing: survey respondents assigned motivation to fish to "to be outdoors", "for relaxation" and, "to experience unpolluted natural surroundings" (Tseng et al., 2006). These responses do not give insight into habitat types preferred by anglers.

Partitioning of RFVPH was based on preference expressed by the angling public. Most often these data are collected by Texas Parks and Wildlife as part of an ongoing creel survey program. This data tracks the amount of time anglers spend fishing and the general area in which their time is spent (Green and Campbell, 2010); however, it does not capture the preference for a given habitat as was identified by the present study. to capture the preferences of the angling public for a given habitat type, new data will need to be collected.

In the current model, it is assumed that each habitat is used equally by anglers. It is likely that one or more habitats are preferred over others; just as it is likely that seasonal variation strongly correlates with fishing effort within habitats. The ability to discern this information, as well as possibly coupling it with catch statistics from individual trips, would help refine the model. It would allow individual preferences and actions to delineate the actual amount spent to travel to specific habitats as well as identify where the majority of fish are caught. This is an opportunity to refine the model by coupling catch data with the amount of effort spent in each habitat type. This would also allow for a more refined understanding of how each habitat contributes to recreational fishing services.

It becomes apparent that these habitats contribute to the outstanding fishery in this region. The major estuarine habitat types investigated as part of this study, marsh edge, seagrass, and oyster reefs, have the ability to provide recreational fishing opportunities in the form of habitat and refuge for game fish and other species. Combining socioeconomic data with habitat use and areal coverage showed that the

annual economic value of recreational fishing within the Mustang Island study site was $83,664,532: marsh edge, seagrass and oyster reef at $2,032,201, $81,550,847, and $81,484, respectively. These values were further differentiated by angler preference to delineate the relative annual value of each recreationally important species: red drum – *Sciaenops ocellatus* ($33,465,813), spotted seatrout – *Cynoscion nebulosus* ($16,732,906), and southern flounder - *Paralichthys lethostigma* ($6,693,163). These estimates of the recreational fishing services provided by these habitats are considered conservative on the basis that they were calculated solely on travel expenditures and license sales. This framework allows use of pertinent data derived from other locations. Refinement of the current model with new data could provide further understanding of relationships among habitats, preferences for those habitats, and ability of the habitat to provide recreational fishing services.

Chapter VI:

Summary and Conclusions:

The Research questions this study seeks to address are:

1. Is it possible to create a model that captures the ecological function of the major estuarine habitats (oyster reef, seagrass meadow, intertidal salt marsh) and explicitly link that function to the provision of nitrogen cycling services?

2. Can this "snapshot" of nitrogen cycling services be transferred to similar systems given the appropriate spatial and temporal data?

3. Can this "snapshot" of nitrogen cycling services be used to ask management-relevant questions with alternate outcomes that can be evaluated in side-by-side comparisons?

4. Can a model also be constructed that explicitly links the provision of recreational fishing services based on the productive capacity of each major estuarine habitat?

The objectives of this study are:

1. Model the ecological function and contribution to nitrogen cycling for oyster flats, seagrass meadows, and intertidal salt marshes of the Mustang Island system;

2. Explicitly link ecological function to ecosystem service provision to create a means of valuation of those services;

3. Use replacement cost methodology as a means of valuing nitrogen cycling services provided by these habitats; and

4. Define the contribution of the major marine habitats of this study in terms of recreational fishing services and their value as a function of travel cost and license sales.

The objectives of the study have been met in that a nitrogen cycling model for oyster reefs, seagrass meadows, and salt marshes has been created that explicitly links ecological function and ecosystem service provision. This link was used to delineate the economic value of this service based on the best available data applicable to the Mustang Island system. Also, a similar model was built to delineate the economic value of the contributions of each aforementioned habitat to the provision of recreational fishing services. Ultimately, these models are designed to improve the communication of important ecological functions of each habitat as well as inform the resource management process.

The general design of each model, based in Simile (Simulistics, 2011), used a declarative approach that allows for each of its components to stand alone as a functional unit. This approach is flexible in that each component is able to be modified to represent the conditions of the specific study site. Therefore, the current framework provides a platform to modify the function of the model to answer relevant questions based on local knowledge and actions. This is important to understand in that the goal of this project was not to create a model to represent all habitats in a definitive manner, but instead to create a flexible process that will allow for the ability to gain insight into management relative decisions based on ecological function and the benefits humans receive as a result.

The oyster reefs of the Mustang Island system cover a relatively small area (35 ha), but are important based on ecological and economic roles they play wherever they may be found (Dame et al., 1989; Peterson et al., 2003; Newell and Koch, 2004; NMFS, 2012). The contribution of this habitat to nutrient cycling is based on the living biomass of the reef, the potential ability to filter phytoplankton (a proxy for nitrogen), and incorporation in to soft-tissue. Using these metrics at the replacement cost price $4.90 kgN-1 (Smith, 2008), it was found that the nitrogen cycling contribution from this reef was valued at $173,000 yr-1. It was also shown that the drivers of this value include water temperature, total suspended solids and chlorophyll a concentration. Therefore, the model is able to provide insight into management relevant questions that will affect any one of these drivers such as dredging or warm-water release near the oyster reef.

Seagrass meadows, as a habitat, have the largest areal extent within the system (4,057 ha) and also have the highest estimated annual value, $12,054,095 yr^{-1}. Much like the other models, physical parameters such as light reaching the canopy, change in seagrass shoot density, whole plant biomass, and nitrogen acquisition were used to calculate the contribution to nitrogen cycling services of the seagrass meadows. It is often recognized that seagrass meadows are areas of high productivity and ecologically important as habitat for commercially important species (Heck et al., 2003; Sheridan and Minello, 2003; Stunz, 2010) while also being recognized as a habitat that is being degraded by human activity (Greening and Janicki, 2006; Orth et al., 2006; Russell et al., 2011). Therefore, management of seagrass meadows is a topic that has received much attention. The current model is a platform to answer questions such as how will

increased nutrient loading affect the phytoplankton communities and in turn light regimes that reach to seagrass canopy? This is particularly important in Texas where a persistent brown tide in the previous decade was shown to decrease belowground seagrass biomass throughout much of the Lower Laguna Madre (Dunton, 1996).

Intertidal salt marshes of the Mustang Barrier Island system are under two specific threats, relative sea level rise and the encroachment of the Black Mangrove, that make understanding the level of ecosystem service provision both timely and relevant. Intertidal salt marsh, dominated by *Spartina alterniflora*, covers approximately 925 ha of the study site and contributes $1,491,000 yr-1 to the nitrogen cycling service. The model is driven by the Normalized Difference Vegetation Index (NDVI), which is often used to monitor salt marsh species on large spatial scales (Hardisky et al., 1983; Hardisky et al., 1984; Gross et al., 1991; Jensen et al., 2002; Adam et al., 2010). Salt marshes are recognized around the world as providing multiple ecosystem services, but much of the study has focused on the provision of recreation services and storm abatement (GecoServ, 2011). Few ecosystem service studies focus on the provision of nitrogen cycling services, while the ecological literature has spent considerable more attention elucidating these processes (Tobias, 2009; GecoServ, 2011). This imbalance would imply that current management frameworks could benefit from reliable, science-driven, economic metrics to help set, communicate, and reach salt marsh management goals.

In the State of Texas, recreational fishing is an important economic driver with an estimated economic impact $8.91 billion in annual retail sales state wide. The goal of

the valuation in this study is to delineate the contribution of the three main estuarine

habitats (marsh edge, seagrass meadows, and oyster reef) to the production of

recreational fishery services based on license sales and travel costs. The three estuarine

habitats in the study site cover approximately 4,145 ha and contribute $83.8 million

dollars (Marsh edge: $2.0 million; Seagrass: $81.5 million; Oyster: $81 thousand) in

recreational fishing services. It is likely that these figures are conservative due to the

fact that only license sales and travel costs to this region were included in the valuation.

Although the value is conservative, the process of attributing recreational fishery

services to a specific habitat is becoming increasingly important. Currently, the National

Resource Damage Assessment (NRDA) process is being revamped and is becoming more

focused on replacing functional units of a given ecosystem rather than a one-to-one

replacement of fish that may have been lost due to human activity (Strange et al, 2004;

Parsons and Kang, 2010). This will increase the need for valuing ecosystem services

based on the habitat-associated ecological functions that provide them.

The driving force behind the current project was to create a set of

communication tools that could effectively translate the language of ecological function

in to the more widely understood realm of economic benefit. Farber et al. (2006)

wrote…"Full ecological-economic models may be the gold standard for establishing the

full range of ecosystem service possibilities and management options." Therefore, this

is part of a growing body of work from the roots of Deep Ecology (Naes, 1972) to

EMERGY accounting (Odum, 1997) to the current push for Ecosystem Based

Management. This history of merging natural science principles to societal norms is the

beginning of a deeper understanding of where society fits into the natural world and what that responsibility entails. Society has used economic metrics as a method of communicating value. Nature on the other hand, has no use for money. Therefore, it is important to understand that these values reflect the importance humans place on the benefits they receive. It is with caution that this statement is made, because it is equally important to understand that economic value does not wholly encompass the needs or desires of individuals in a society. Rather, it helps provide society a collective ability to express priorities in the face of the complex connections with ourselves, each other, and nature.

This complexity has driven the effort to communicate meaningful connections between ecological function and ecosystem service provision in as direct a manner as possible. An effort to focus on the essential elements of the model was intentional. A model focused solely on the essential elements is likely to be able to be applied to other study sites and also more likely to be reliable in its' output. Conversely, increased complexity will allow for more fine scale determination of inputs, flux, and loss of nitrogen. While this may be useful, it is not the goal of this project. Currently, the goal is to take a snapshot of nitrogen cycling services associated with each habitat to communicate the benefits humans receive in return. The current framework is a tool to measure relative benefit in comparison to varied management decisions so that benefits can be maximized and/or losses minimized as each decision. Focusing on only those most basic elements is used as a tool to boil down current knowledge of each habitat in

a manner that will facilitate both discussion and understanding of the factors that drive those benefits received but remain meaningful to the largest share of stakeholders.

The current models are also an opportunity for further refinement and improvement. One way in which these may be improved, is to connect them in a functional manner. Currently, each model stands alone representing a singular habitat. But as ecosystem based management approaches become more prevalent, it will require that management decisions be made in such a way that all habitats will have to be seen holistically as an ecosystem instead of individual units. For example, both seagrass meadows and oyster reefs are spatially correlated (Heck et al., 2003). This may be due to the movement of nutrient rich matter from the water column into the benthic sediments and/or the reduction of water velocity as both create friction allowing suspended particles to reach the bottom. Regardless of reason, these explicit connections are necessary to be able to fully communicate the importance of management actions on these foundational habitats.

It will also be necessary to improve the economic data available for given habitats and services. Currently, it is common practice to transfer values from one study site to the next by adjusting for local conditions. But, a stronger understanding of what drives economic value for ecosystem services can only be obtained through original valuation studies designed to do so. Demographics also constrain the ability to pay for these services, therefore knowledge of economic value and what motivates people will have bearing on how to effectively communicate the importance of the benefits we receive from nature. Ultimately, reinforcing the connections between humans and

nature will allow for opportunities to embrace a more complete understanding of our

living planet and its' ability to provide the necessary components for humans to

continue to thrive and prosper.

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Biographical Sketch:

Jeff began his love for Marine Biology in the first grade after reading a Jacques Cousteau book. His passion for the oceans has never faded even though the first 15 years of his life were spent surrounded by the Sonoran Desert. Jeff received a Bachelor of Science in Ecology and Evolutionary Biology from the University of Arizona where he helped teach undergraduate Marine Ecology labs for Dr. Donald Thomson. Shortly after graduating, he entered a Master's of Science program in Mariculture at Texas A&M University – Corpus Christi where he worked on phenotypic plasticity of hatchery-reared red drum with Texas Parks and Wildlife. Upon completion, Jeff started teaching science at Miller High School in Corpus Christi, Texas, which he considers to be very formative years in his career. He taught in the public school for four years during which time the Harte Research Institute became operational at Texas A&M University – Corpus Christi. He then entered the newly formed Marine Biology Doctoral program where he studied Ecosystem Services under Drs. David Yoskowitz and Joe Fox.