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Anthropogenic and natural perturbations on lower Barataria Bay, Louisiana: detecting responses of marsh-edge fishes and decapod crustaceans

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ANTHROPOGENIC AND NATURAL PERTURBATIONS ON LOWER BARATARIA BAY, LOUISIANA: DETECTING RESPONSES OF MARSH-EDGE FISHES AND DECAPOD CRUSTACEANS

A Dissertation
Submitted to the Graduate Faculty of the
Louisiana State University and
Agricultural and Mechanical College
in partial fulfillment of the
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Doctor of Philosophy

in

The Department of Oceanography and Coastal Sciences

by
Agatha-Marie Fuller Roth
B.S., University of Alabama, 1999
M.S., Southeastern Louisiana University, 2003
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DEDICATION

This work is dedicated to Mr. Richard Joseph Roth, Jr.

To My Father who excelled at his profession while maintaining his gentleman's status in society and giving me a fairytale childhood and a royal adulthood.

Thank you for life, love, faith, and drive.
Miss You.

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talents for achieving while still enjoying life kept things in perspective for me. Mrs. Richard J. Roth, Jr. (Cindy, mother), thank you for always believing in me and pushing me to reach my fullest potential no matter what anyone said. At this time I would like to acknowledge those who departed during this endeavor, but remained supportive to the end; Hon. John Hainkel, Jr. (Uncle Johnny), Mrs. Richard Jaubert (Aunt Arie), and Mrs. Hiram Blow (Tía Helen).

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ABSTRACT

Barataria Bay, Louisiana is a dynamic estuary with ongoing disturbances that is in need of restoration. Development and validation of a lower Barataria Bay index of biotic integrity (IBI) for the summer season was the focus of my research. This IBI was created using 2005 data and evaluated with 2006 and 2007 data to demonstrate the feasibility of this approach in coastal Louisiana. The IBI successfully distinguished sites with differing levels of degradation using nine fish metrics. While pursuing this effort, two serendipitous events occurred when an oil spill then a hurricane impacted the study area. This gave me opportunities to examine pulse perturbations in the area. I showed immediate effects from the 2005 oil spill using a before-after-control-impact (BACI) analysis and found that fish abundances were significantly different days after the spill. I examined the recovery path of the nekton community after Hurricane Katrina and found that by the spring the year following the storm there were differences in species composition from pre-Katrina compositions. However, by two years post-Katrina species compositions and environmental variables measurements were similar to pre-storm conditions. I examined the transformation from *Spartina*- to black mangrove- dominated marsh edge (a long-term or press perturbation) and its effects on the nekton community. Nekton abundances were higher in the black mangrove and transition (mixed *Spartina* and black mangrove) vegetation dominated marsh-edge habitat type than the *Spartina* dominated marsh-edge. However, a fisheries species, *Farfantepenaeus aztecus* (brown shrimp), was more associated with *Spartina* than mangrove. By creating loop models of the study area's marsh-edge community, I explored three other press perturbations along with black mangrove encroachment. These other perturbations were freshwater diversions, shrimping pressure, and wetland loss. Models predicted that mangroves encroachment decreased grass shrimp, freshwater diversions increased the water column predators, shrimping decreased wading birds and algae, and wetland loss had a negative effect on algae. Variations to the model showed some differences among the community responses. This dissertation illustrates how resilient the fauna is in Barataria Bay, which along with the proper assessment techniques, makes this area a strong candidate for restoration and management efforts.

CHAPTER I
BACKGROUND: ASSESSING PERTURBATIONS IN BARATARIA BAY

INTRODUCTION

Southeastern Louisiana is a dynamic and productive area with approximately 40% of the United States' coastal wetlands (Boesch et al. 1994). Unfortunately these wetlands are decreasing in area at an alarming rate, with an estimated 1,704 km^2 of wetland lost in the past 30 years (Barras et al. 2004). This loss is devastating because Louisiana coastal wetlands have significant ecological and socioeconomic value. Coastal wetlands provide essential habitat for commercial and recreational fisheries and migratory waterfowl, protect populated southern cities and ports from storm surges, and support significant oil and gas production (Boesch et al. 1994). Louisiana makes a substantial contribution to the country's energy supply with the offshore oil ports handling approximately 13% of the oil for the entire country (Revette 2007). The outer continental shelf combined with inshore production ranks the state as number one in crude oil and number two in natural gas production (Crouch 2007). In 1999, ports of Louisiana created $422.97 million in revenue with approximately 345 million metric tons of waterborne commerce (Ryan 2001). Along with industrial commerce, fisheries are profitable businesses for coastal Louisiana. The Gulf of Mexico has the second highest annual commercial fisheries landings in the United States, with Louisiana contributing over 400, 000 metric tons to the Gulf harvests in 2006 (NMFS 2006). The dominant fisheries species in Louisiana are *Callinectes sapidus* (blue crab), *Litopenaeus setiferus* (white shrimp), *Farfantepenaeus aztecus* (brown shrimp), and *Crassostrea virginicus* (American oyster), with *Brevoortia patronus* (gulf menhaden) contributing significantly to the nation's fisheries for oil and fish meal (Perry and McIlwain 1986, Chesney et al. 2000, Shervette et al. 2004, NMFS 2006). Growth of these industries spurred by increasing human demands for food and energy are growing and expanding the multiple stresses in Louisiana's coastal areas, including altered water quality, habitat disturbance and alteration, modifications in flow regime, and nutrient loading.

These stressors threaten the current structure of the flora and fauna of northern Gulf of Mexico coastal environments. There have been increases in the rate of tropical storm landfalls and their destructiveness in recent decades (Emanuel 2005). Warming trends are also causing northern movement of some species and a decline in others (Oviatt 2004). Local consequences of these changes were evident after the 2005 and 2008 hurricane seasons, when four major storms hit the Mississippi, Louisiana, and Texas coast lines and caused massive wetland and infrastructure destruction and loss of life. Warming trends have also caused a decrease in freeze events that are fostering *Avicennia germinans* (black mangrove) expansion noticeably along their historic northern limit on the Chandeleurs Islands in southeastern Louisiana (Mendelssohn and McKee 2000). These environmental modifications will affect the large number of species that utilize coastal wetlands during their early life history stages or for some throughout their lives.

Most perturbations that alter environmental factors not only affect particular species, life history stages, or populations, but also groups of populations and entire communities occurring in the same area (Crowder 1990). I used a community ecology approach to address questions about the southern portions of Barataria Bay in southeastern Louisiana. This methodology can be difficult as it is important to consider the idea of scale. Changes seen at a site may not be as evident across larger scales in the study area or uniform over the entire Bay. Based on quantitative samples of the marsh-edge nekton community, I explored how the community responded to anthropogenic and natural disturbances.

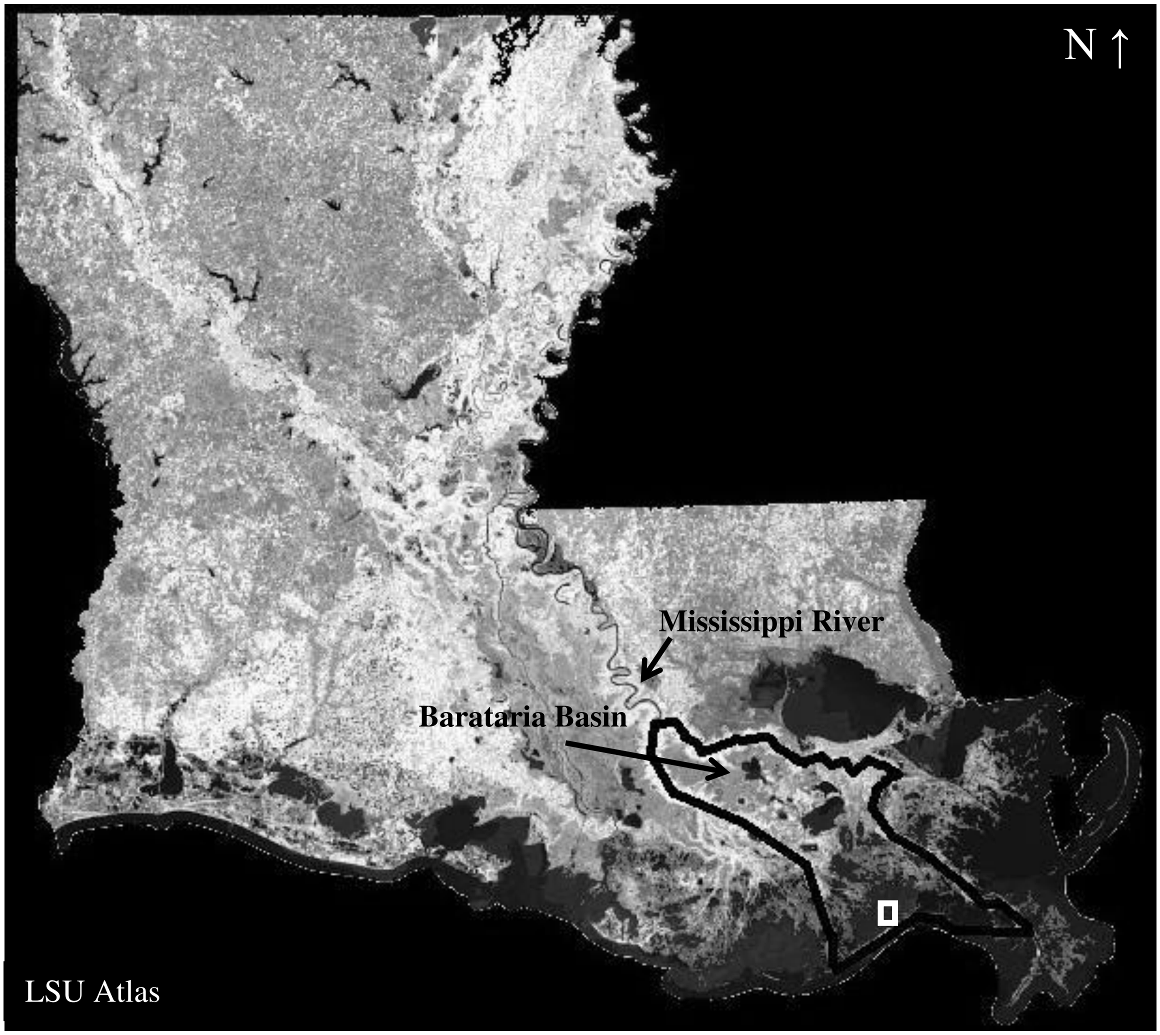

Figure 1.1. Map of Louisiana and the Barataria Basin outlined in black and the study area outlined in white.

MATERIAL AND METHODS

Study Area

Barataria Basin is an inactive deltaic region that lies west of the current mouth of the Mississippi River (Fig 1.1). River input combined with a humid, subtropical climate and other biological and physical gradients have created a specialized and productive system (Bahr and Hebrard 1976, Conner and Day 1987). Approximately 55% of the wetlands that interact with the Gulf of Mexico are included in Barataria Basin (Turner 2003). This estuary is approximately 110 km long and 50 km wide and is characterized by a large salinity gradient. Historically, this system was divided along the salinity gradient into swamp forest, fresh marsh, brackish marsh, saline marsh, and offshore habitats (Bahr and Hebrard 1976), with plant species diversity decreasing with increasing salinity from north to south. The marsh was historically dominated by *Spartina* and still is but with increasing biomass of black mangroves. Areas of high perturbation have other vegetation such as *Batis maritima* (saltwort), *Iva frutescence* (marsh elder), or *Phragmites australis* (common reed).

The southern portion of Barataria Bay has 145,000 ha of salt marsh (Conner and Day 1987), which results from the closure of the Lafourche-Mississippi River connection and an enhanced levee system that has cut off freshwater input to the Basin (Conner and Day 1987). Today, the main source of fresh water for the Basin is a mean precipitation of 1.6 m yr^{-1}. Precipitation, tidal flux, and winds influence Barataria Bay's variable salinity, which ranges seasonally and spatially from 6 to 22 psu (Baumann 1987, Childers et al. 1990, Baltz et al. 1993). The substrate is mainly fine sediment that was deposited by the Mississippi River. Barataria Bay is a microtidal system with a semidiurnal mean tidal range of 0.32 m. The southern portion of the Bay is highly turbid (>10 NTU) and shallow, with depth rarely exceeding 2 m except in the Barataria Waterway and channels (Baltz et al. 1993, Allen and Baltz 1997). Although this is a typical northern Gulf of Mexico estuary, it has seen an accelerating increase of human and natural disturbances in recent years. Because of perturbation and the obvious economic importance of Louisiana's coastal marshes, efforts to study and monitor the Basin have increased.

Field and Laboratory Methods

Because I wanted to successfully estimate densities and population abundances in a shallow estuary, I used a drop sampler (Zimmerman et al. 1984, Baltz et al. 1993, Baltz et al.

1998, Rozas and Minello 1997). This study was targeting the small species and younger life-history stages of larger species found along the marsh edge. Drop sampling is quantitative with high recovery efficiency for these size classes in the marsh (Rozas and Minello 1997, Steele et al. 2006). My drop sampler was a 1.18 m^2 fiber glass cylinder with an aluminum skirt as modified from the Zimmerman et al. (1984) design (Baltz et al. 1993, 1998). It extended approximately 2 m from the bow of a 5.2 m Boston Whaler. The sampler was attached to a 2.4 m tall mast that was connected to a 3.7 m boom (Fig 1.2). A pin through a swivel attached the sampler to a winch on the boom. After quietly maneuvering the boat towards the marsh-edge, I deployed the sampler by pulling the release pin, which caused the sampler to drop into the water and the aluminum skirt to cut into the soft substrate. After the sampler was seated, I noted time of day and marked the longitude and latitude coordinates of the site with a global positioning system (GPS) unit. Sampling occurred at random times during the day to ensure all water levels were represented throughout the study area. This sampler enclosed the water column nekton community, and I pumped the water from the sampler with a Teel Trash Pump and filtered it through a 333 μm mesh plankton net to collect all free swimming fishes and macroinvertebrates. Collected invertebrates were fixed with 10% formalin and fishes were placed in an ice bath on the boat. In the laboratory, fishes were fixed with 10% formalin and all nekton were indentified, counted, and preserved with 70% ethanol. All fishes were measure to standard length (SL) and sex and carapace width (CW) were recorded for all crabs. Because altered water and habitat quality affect nekton community composition and abundances, I measured the following variables at each site (place where the sampler was dropped): salinity (psu), temperature (°C), and dissolved oxygen (mg l^{-1}) with a YSI 85 water quality meter. Mean water velocity (cm s^{-1}) was measured with a Marsh-McBirney Model 2000 Flow-Mate, turbidity (NTU) was analyzed in

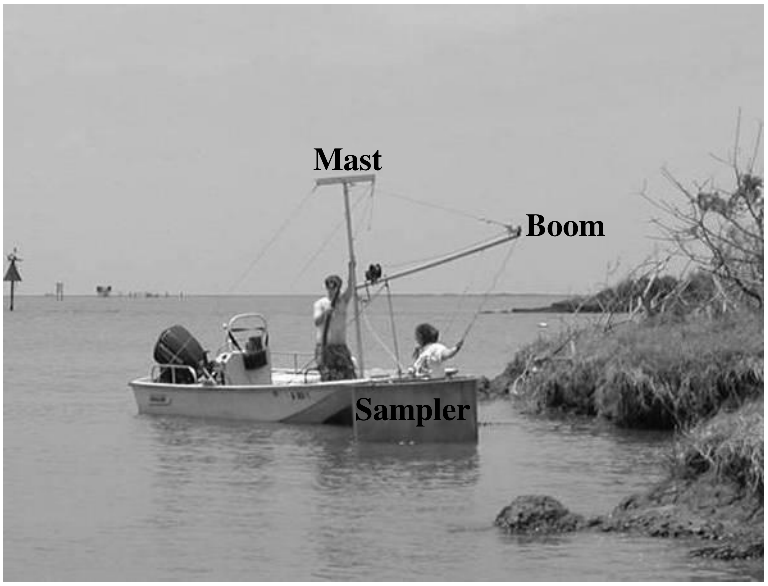

Figure 1.2. Drop sampler near the marsh edge with mast and boom set-up on the Boston Whaler.

the lab with a Hach 2100N, and pore-water toxicity analyzed from sediment cores with a Microtox® Model M500. Mean depth (m) (maximum + minimum depth/2), distance to marsh edge from the sampler (m), dominant and subdominant marsh edge vegetation, and dominant and subdominant substrate type were measured outside of the sampler. Samples were collected in the southern portion of Barataria Bay near Grand Isle (Fig 1.1), Louisiana from March to August in the years 2005, 2006, and 2007.

SUMMARY

My primary goal was to characterize the response of marsh-edge nekton to different levels of degradation and create a biological assessment tool to identify the level of degradation of the lower Barataria Bay marsh. While conducting sampling for the aforementioned objective, there were some serendipitous anthropogenic and natural disturbances on the study area. There has also been a long-term perturbation (a press perturbations sensu Bender et al. 1984) in study area with increases in temperature over the past decade. I was able to test hypotheses about an oil spill, hurricane, and a shift in marsh-edge dominant vegetation affect on the lower Barataria Bay nekton abundances and compositions. I analyzed all data with a MANOVA to ensure there were no interactions among the variables year, season, habitat, and degradation level. The overall MANOVA was significant ($F_{44, 603} = 1.64$, $P = 0.007$) and had significant interactions for season and habitat ($F_{3, 603} = 3.74$, $P = 0.011$), and season, habitat, and degradation level ($F_{3, 603} = 3.32$, $P = 0.020$). These interactions were taken into account in further analyses and seasons were treated as separate data sets for all analyses. While there are seasonal variations in species compositions and abundances within years in estuaries, patterns of a given season across years are predictable in estuaries (Loneragan 1989).

To assess ecosystem health and to monitor restoration efforts, researchers often use an index of biotic integrity (IBI). Chapter 2 developed and validated a Lower Barataria Bay IBI for the months June, July, and August. This preliminary IBI was created with 2005 data and tested with 2006 and 2007 data to demonstrate the feasibility of the approach in coastal Louisiana. The IBI evaluated levels of degradation among sites by combining nine fish metrics created from fish abundance, composition, and life history attributes. The IBI successfully distinguished sites with low, moderate, and high degradation. IBIs can be used by resource managers with varying expertise to monitor the effectiveness of restoration projects or asses areas for future restoration efforts.

After a month into sampling, an oil spill accord near the upper portion of the study area. In Chapter 3, I examined the immediate effects from an oil spill in the study area in spring 2005. I had the unique opportunity to use a before-after-control-impact (BACI) analysis and showed that fishes were displaced days after the spill, but long-term examination of area recovery was thwarted by Hurricanes Cindy and Katrina. Chapter 4 focused on the resilience of the nekton community after category 1 and category 3 hurricanes made landfall near the study area in 2005. I examined the recovery path of the nekton community after the two storm events, and found that although there were community differences a year after the storm nekton abundance and community structure were similar to pre-storm conditions within two years.

Although Chapters 3 and 4 are specific examples of human and natural perturbations, Chapter 5 focuses on the transformation of marsh from a *Spartina* dominated to black mangrove dominated system, and possible effects of this transition on the nekton community. Although *Spartina* still dominates, there are many mixed stands and other areas where black mangrove dominates. I used samples and environment data collected along those two marsh-edge vegetation types and a transitional marsh-edge habitat type (both *Spartina* and mangrove as co-dominant vegetation). I found there was a trend of mangroves occurring with steeper marsh-edge slopes than *Spartina* and that nekton abundances were higher in mangrove dominated than *Spartina* dominated marsh-edge habitat type. An important fisheries species *Farfantepenaeus aztecus* (brown shrimp) that were more closely associated with *Spartina* than black mangrove and this relationship needs to be explored in more detail given the expected changes in habitat structure.

The last chapter of my study concentrated on modeling the biotic and abiotic relationships in the marsh edge habitat. Chapter 5 is a qualitative (Loop) model of the Barataria Bay marsh-edge community, and it modeled possible shifts in trophic dynamics from alterations such as black mangrove encroachment, increased freshwater diversion, shrimping pressure, and wetland loss. The model has nodes for carnivorous wading birds, water column predators, penaeid shrimp, benthic fishes, small crabs, and grass shrimp as the biological variables. Habitat was represented in the model with nodes for turbidity, fine sediment, algae, *Spartina*, and black mangrove. These three models predicted that mangrove encroachment would decrease grass shrimp; freshwater diversion would increase water column predators; shrimp pressure would decrease wading birds and algae; and wetland loss would have a negative effect on algae. This qualitative model can be a useful tool to pinpoint relationships within the community that would be of interest to resource managers and users, especially with regard to the long list of ongoing changes occurring in Barataria Bay. Barataria Bay is a dynamic estuary that has experienced high levels of historical, present, and probably future disturbances, and is desperately in need of restoration. Each of these Chapters examined a different aspect of the perturbations that historically and currently affect the Barataria Bay. Importantly, my dissertation illustrates how resilient fauna in the Bay is, which makes this area a good candidate for restoration and management efforts.

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CHAPTER II
CREATING AN ESTUARINE INDEX OF BIOTIC INTEGRITY (IBI) FOR LOWER BARATARIA BAY, LOUISIANA

INTRODUCTION

Southeastern Louisiana coastal waters have been experiencing increasing human influences for decades, and these include Native American fishing and the arrival of Europeans (Jackson et al. 2001, Jumonville 2002). Major impacts from humans in the system are levee building and canal dredging, oil and gas exploration, water control through impoundment, diversions, commercial and recreational fisheries, introduction of non-indigenous species, and pollution (Chesney et al. 2000). Louisiana's coastal parishes housed over 47% of the total Louisiana population, according to census estimates in 2006 (U.S. Census Bureau 2007). People have developed the barrier islands that protected the decreasing wetlands of Louisiana. The coast also provides the state with economically important industries, including shipping, petro-chemical, and fisheries. Companies have dredged areas which increases erosion and creates spoil bank edge marshes in the historically monospecific *Spartina alterniflora* (*Spartina*) marsh.

The Mississippi River is the world's busiest waterway and the Port of New Orleans has been the center of international trade since 1718 (PORTNO 2009). This has led to levee building and dredging efforts such as the Mississippi River Gulf Outlet. These levees have decreased freshwater, sediment, and nutrient input into the wetlands (Lane and Day 1999). Since its inception in the early 1900s the petrochemical industry has been in Louisiana and continues to expand (Revette 2007, LDED 2009). Approximately 66,498 km of pipelines distribute natural gas and crude oil in, around, and out of Louisiana (LDED 2009). Boat traffic needed to support and monitor these pipelines and ancillary equipment further erode fragile wetlands. There are also numerous platforms and pumps dotting the inshore region of coastal bays. These inshore oil and gas infrastructures have exposed wetlands to oil and gas spills and leaks (DOT 2005). The petrochemical business influences the same coastal areas as the fishing industry.

Recreational and commercial fisheries in Louisiana are some of the most profitable in the country (Jackson et al. 2001). In 2006, commercial fisheries landings exceeded 407, 900 metric tons and there were approximately 1.2 million recreational anglers (NMFS 2006). Commercial trawl fisheries have dredged and changed the biota of large portions of the coastal systems including wetlands in pursuits of white and brown shrimp (*Litopenaeus setiferus* and *Farfantepenaeus aztecus*). These fisheries also produce high rates of bycatch mortality of the estuarine dependent species found on Louisiana's coasts (Rogers et al. 1997). With these anthropogenic influences compounding the effects of sea-level rise and erosion it is necessary to develop a method of easy assessment of marsh-edge health. One way to assess human effects on a system is to measure biological changes along a known gradient of human influences.

Biological assessments of ecosystem health using the multimetric index of biotic integrity (IBI) is effective in documenting changing responses to changes in habitat quality (Karr and Chu 1997, Gibson et al. 2000). The index IBI utilizes biological attributes, which are measureable components of biological systems, to show differences resulting from human caused alterations to water quality, flow regime, energy resources, or biological interactions (Karr and Chu 1997). The most useful metrics are biological attributes that are easy to interpret, change with increasing human influences, are sensitive to a range of biological stressors, and can be used to discriminate between human induced and natural stresses of a system (Karr and Chu 1997). Examples of the latter are difficult to identify for Barataria Bay, as it has a long history of anthropogenic and natural disturbances (i.e. industry and tropical storms). Examples of measurable degrees of human influences are types of disturbances (fisheries or industry), proximity to human occupancy, or type of marsh-edge vegetation (*Spartina* to spoil bank vegetation). Multiple attributes should be tested to validate its merit for the index. As many indexes have been created for freshwater systems (Karr 1981, Miller et al. 1988, Oberdorff and Hughes 1992, Karr and Chu 1997), few have addressed dynamic estuarine systems (Thompson and Fitzhugh 1986, Engle et al. 1994, Deegan et al. 1997, Engle and Summers 1999, Hughes et al. 2002). Most estuarine IBI researchers concentrated on fish community attributes in constructing their multimetric indexes, because fishes respond more predictably to abiotic changes to habitat quality (Thompson and Fitzhugh 1986, Miller et al. 1988, Karr 1991, Oberdorff and Hughes 1992, Deegan et al. 1997, Karr and Chu 1997, Simon and Lyons 1995, Hughes et al. 2002). Also, state and federal agencies use fish community criteria for standards in freshwater ecosystem health (Simon and Lyons 1995, Gibson et al. 2000). Fishes are preferred because they have more literature than other macrofauna written about them, are present throughout aquatic habitats, are easier to identify than small invertebrates, range across multiple trophic levels in the same area, are long-lived, and have high levels of public awareness (Karr 1981, Whitfield and Elliott 2002).

My objective was to identify metrics to diagnose the condition of locations that have suffered various levels of degradation in a complex estuarine system that has experienced a long history of anthropogenic influence. To do so, I used an iterative process. I first had to detect response metrics from testing, evaluate the metric defined health of sites against previous expectations, and interpret these values in terms of an assessment of the entire area (Karr and Chu 1997). The metrics used can show differences with a range from apparently healthy area with many stress intolerant species in multiple trophic levels to degraded areas with few stress tolerant fish species (Karr 1981). The classification criteria for IBIs are both species richness and ecological factors (Karr 1981). Species richness criteria such as diversity are useful tools to assess system health, but can overlook important rank order shifts in complex ecosystems like estuaries. Using species composition metrics with trophic levels or life history categories and abundance helps to classify the habitat quality of a system (Oberdorff and Hughes 1992, Deegan et al. 1997, Engle and Summers 1999). I sampled different attributes of the system to identify their ability to respond across differing levels of degradation (Karr and Chu 1997). Once appropriate metrics were identified they were scored from 5 (less degraded) to 3 (moderately degraded) and then 1 (highly degraded) (Deegan et al. 1997, Karr and Chu 1997, Engle and Summers 1999). These metric scores are summed to create the index value for each site sampled (Simon and Lyons 1995, Deegan et al. 1997, Karr and Chu 1997). Statistical analyses such as ANOVA and discriminant analyses were then used to detect significant differences among index values for sites in differing states of degradation (Deegan et al. 1997, Engle and Summers 1999). Because these indexes are multimetric, they are sensitive to the changes to a system from increased anthropogenic influences (Karr and Chu 1997). In this study, I used two seasons of data from 2005 to develop and test fish metrics in an effort to create a lower Barataria Bay IBI. I used data from 2006 and 2007 to validate the model.

MATERIALS AND METHODS

Study Area

The study area consisted of marsh-edge habitats surrounding three island groups in the lower portion of Barataria Waterway Bay (Fig 2.1). This area was chosen to reflect uniform water quality measurements (water temperature (˚C), salinity (psu), and dissolved oxygen (mg l^{-1})), depth (cm), and mean water velocity (cm s^{-1}) to minimize these influences on the fish composition of the samples. Based on my assessment of the accumulative influence of human activity, one of three ordinal levels of degradation were assigned to each sampling location within the sampling area. Less degraded locations had sloping marsh edge, were dominated by *Spartina* and/or *Avicenna germinans* (mangrove) vegetation and had silt and organic matter as primary substrate (Fig 2.2.a). These less degraded locations were generally harder to reach and somewhat protected. Moderately degraded locations have more boat and wave action or were areas of past human influences that had been decreased in recent time. While the marsh-edge vegetation is similar between less and moderately degraded locations, moderately degraded location has shell and silt as its dominant substrate (Fig 2.2.b). Highly degraded locations have weedy spoil bank vegetation such as *Myrica cerifera* (wax myrtle) and *Iva frustescens* (marsh elder) or *Phragmites australis* (common reed), a steep and often eroded marsh-edge slope with shell or sand substrate (Fig 2.2.c). These sites were also deeper and were harder to sample along the marsh edge. Most of the highly degraded locations had trash along the marsh edge such as tires, rebar, and cement. At each of the locations, we took three independent samples to fully characterize the marsh edge of that degradation level replicate (Fig 2.1).

Field and Laboratory Methods

Monthly from March to August I randomly deployed the drop sampler at three independent sites within each of the four replicate locations for the three differing levels of degradation (*n* = 216). At each site after the sampler was seated, GPS coordinates, time of day, marsh edge vegetation, and dominant substrate were recorded. I also measured mean water velocity, mean depth, and water quality before evacuating the trap and collecting all fishes that were fixed in formalin and preserved in alcohol. At the laboratory turbidity and porewater toxicity were analyzed from water and sediment samples collected outside the trap. I identified and counted the collected fishes and divided fish taxa into life history categories of estuarine resident or spawner, the trophic category of top carnivore, and spatial category of benthic life-history (Table 2.1). Species were also divided into compositional groups such as the orders Perciformes (perch-like fish) and Plueronectiformes (flatfish) and the families Sciaenidae (drums) and Gobiidae (gobies) as intolerant taxa. The order Clupeiformes (anchovy and menhaden) represented tolerant taxa (Table 2.1). Species richness (number of species in a sample), diversity (Shannon-Weiner diversity index H'), and total abundances were calculated for each sample. All categories were examined for potential use as metrics for development of the IBI.

Figure 2.1. Map of study area with sampling locations of the three levels of degradation boxed with L = less degradation, M = moderate degradation, H = high degradation and representatives of sampling sites within each where L sites are (○), M sites are (□), and H sites are (ø).

Figure 2.2. Examples of the marsh-edge appearance for a) less degraded location, b) moderately degraded location, and c) highly degraded location.

Table 2.1. List of fish taxa, common name, and order with superscript representing the metrics a) estuarine resident, b) estuarine spawner, c) top carnivore, and d) benthic life-history.

Species/Family	Common Name	Order
Anchoa hepsetus	striped anchovy	Clupeiformes
Anchoa mitchilli[a,b]	bay anchovy	Clupeiformes
Ariopsis felis[b,d]	hardhead catfish	Siluriformes
Bairdiella chrysoura[d]	silver perch	Perciformes
Bathygobius soporator[b,d]	frillfin goby	Perciformes
Brevoortia patronus	gulf menhaden	Clupeiformes
Chaetodipterus faber	Atlantic spadefish	Perciformes
Citharichthys spilopterus[c,d]	bay whiff	Plueronectiformes
Ctenogobius boleosoma[a,b,d]	darter goby	Perciformes
Cynoscion arenarius[b,c,d]	sand trout	Perciformes
Cynoscion nebulosus[b,c,d]	spotted trout	Perciformes
Dasyatis americana[c,d]	stingray	Myliobatiformes
Elops saurus	ladyfish	Elopiformes
Gerridae[b]	mojarra	Perciformes
Gobiesox strumosus[b,d]	skilletfish	Perciformes
Gobiidae[a,b,d]	goby	Perciformes
Gobiosoma bosc[a,b,d]	naked goby	Perciformes
Gobiosoma robustum[a,b,d]	code goby	Perciformes
Hypleurochilus geminatus[b,d]	crested blenny	Perciformes
Lagodon rhomboides[a,b,c,d]	pinfish	Perciformes
Leiostomus xanthurus[c,d]	spot	Perciformes
Lutjanus griseus[c]	grey snapper	Perciformes

Table 2.1 (Continued)

Membras martinica[a,b]	rough silverside	Atheriniformes
Menidia beryllina[a,b]	inland silverside	Atheriniformes
Menticirrhus americanus[c,d]	southern kingfish	Perciformes
Microphis brachyurus[b,d]	pipefish	Gasterosteiformes
Micropogonias undulatus[d]	Atlantic croaker	Perciformes
Mugil cephalus[d]	striped mullet	Mugiliformes
Mugil curema	white mullet	Mugiliformes
Myrophis punctatus[d]	speckled worm eel	Anguiliformes
Neoconger mucronatus[d]	ridged eel	Anguiliformes
Opsanus beta[a,b,d]	gulf toadfish	Batrachoidoformes
Paralichthys lethostigma[c,d]	southern flounders	Plueronectiformes
Pogonias cromis[c,d]	black drum	Perciformes
Pomacanthidae[d]	angelfish	Perciformes
Sciaenidae	drum	Perciformes
Sciaenops ocellatus[c,d]	red drum	Perciformes
Sphoeroides parvus[d]	least puffer	Tetradontiformes
Strongylura marina	Atlantic needlefish	Beloniformes
Syacium papillosum[c,d]	dusky flounder	Plueronectiformes
Symphurus plagiusa[d]	blackcheek tonguefish	Plueronectiformes
Syngnathus louisianae[b,d]	chain pipefish	Gasterosteiformes
Syngnathus scovelli[b,d]	gulf pipefish	Gasterosteiformes
Synodus foetens	inshore lizardfish	Aulopiformes

Statistical Methods

My IBI was developed with data collected in 2005 and tested with data collected in 2006 and 2007 using ANOVA and discriminant analysis in SAS (2004). Environmental data were tested with ANOVAs to determine if there were differences among sites in the sample area. This was done to avoid possible differences in fish data among degradation locations were not relicts from time of day or other natural environmental changes. To increase the probability of differences detected by the analyses to be artificially caused and not by natural variations, the best time to run an IBI is during a stable time period in the estuary. I tested samples for differences between months of the spring and summer seasons with ANOVAs to determine which season had no significant temporal differences among fishes.

All life history, trophic, spatial, and taxonomical categories were treated as proportional metrics (number in category/total abundance*100%) for each sample. Total abundance data were log (X + 1) transformed for normality (SAS 2004). For metric development ANOVAs, the main effect factors were the pre-assigned degradation levels less degraded (L), moderately degraded (M), and highly degraded (H). The individual metrics were the dependent variables. Tukey's post hoc pairwise comparisons were used to test differences among degradation levels and $\alpha = 0.05$ was used for significance.

Metrics that were sensitive to changes in habitat quality received index scores based on the highest metric measurements representing best habitat quality. These metrics were divided into thirds and scored based on Karr and Chu (1997). Proportional metrics in the range of 100 to 66.67% representing highest quality and a metric score of 5, 66.66 to 33.33% representing moderate habitat quality with a metric score of 3, and 33.32 to 0% representing the lowest habitat quality with a metric score of 1. Because Clupeiformes contains anchovy and menhaden, which are considered tolerant taxa this metric received scores based on the opposite of the intolerant taxa metric scores. The summation of these metric scores created the sample's index score, and analyses yielded a refinement of site assignments. Metrics were re-analyzed with ANOVAs to detect their sensitivity to the site level degradation assignments.

A discriminant analysis estimated the classification efficiency of the newly assigned groups with the multiple metrics (Deegan et al. 1997, Engle and Summers 1999). This analysis describes algebraically the relationship among individuals in a site and makes the differences in those relationships evident. Populations are then separated into groups based on the observed characteristics (SAS 2004). These derived groupings are compared to the a priori group assignments and error rates are calculated. This analysis utilized many metrics for group verification. These methods were repeated on the 2006 and 2007 data to validate the metrics chosen and verify classifications were appropriate. It is important to use all metrics as one will not show the clear separation among degradation levels especially in a dynamic system like the estuary. This is evident in graphs of the multimetric scores and graphs of the specific metrics such as total abundance or diversity alone. Segregation of metrics among groups is visually evident with multidimensional scaling (MDS). This visually depicts the relationships among sites in three-dimensional space. MDS dimensions are tested for significant differences with MANOVAs to determine clusters created are significant. There is also a goodness of fit test that produces a stress level. This is the inverse of an r^2 where a stress level below 0.2 indicates the MDS is a good representation of these data.

RESULTS

Selection of Metrics and Index Scores

Environmental variables did not significantly differ among degradation levels, so biological changes observed are not from small environmental variable shifts in the study area. Microtox analysis did not determine enough toxic samples to use in analyses. The ANOVAs run on month data found June, July, and August had no significant differences for each metric among these months, while March, April, and May showed significant differences for each metric. Spring samples had higher variation possibly from weather changes and immigration of species into the estuary. Therefore only summer data were used in the rest of the analyses.

From the metrics tested, percent top carnivore, percent Plueronectiformes, and percent Sciaenidae showed no significant differences among pre-assigned groups or the re-assigned index groups (Table 2.2). Estuarine resident, estuarine spawner and percent Clupeiformes did not have a significant difference among pre-assigned groups, but showed significant differences within the new index groupings (Table 2.2). I used the nine remaining metrics for the multimetric analyses and used these for 2006 and 2007 IBI development and validation. The use of multiple metrics instead of total abundance or diversity is to show clear separation of sites in different degradation levels (Fig 2.3).

Discriminate analysis for original grouping of the metrics in the 2005 samples had a high total error rate of 0.42 and had multiple misclassifications for each group (Table 2.3). Site index scores ranged from 45 with all metrics scoring for low degradation (5) to 9 with all metrics scoring for high degradation (1). I re-assigned an index group label of L for sites with summed metric scores between 45-34, an index group label of M for sites with summed metric scores between 33-22, and an index group label of H for sites with summed metric scores between 21-9. With the re-assignment of degradation levels based on summed index scores the number of highly degraded sites increased from 36 to 58 while the number of low degraded sites decreased from 36 to 19 (Table 2.3). The discriminant analysis on the index score groupings came out with a 0.065 error rate and most classifications were correctly distributed across groups (Table 2.3). The difference between the separation of sites based on old and new degradation level assignments is illustrated with significant MDS graphics. The original group assignments had no significant clusters in three-dimensional space, while the new group assignments based on the multiple metric scores have significant clusters (Fig 2.4.a,b).

Validation of Metrics and Index Scores

Metrics were significantly different across degradation levels for both 2006 and 2007 data (Table 2.4) and discriminant analyses had error rates less than 0.070 with most classifications correct for 2006 and 2007 index scores (Table 2.5). The sites that were highly and moderately degraded were higher in number while low degradation sites were less numerous for these years than the original site degradation assignments.

Table 2.2. ANOVA results for all metrics for the original group assignments and new group assignments with * indicating significant p-values ($P < 0.05$).

Original Grouping	F-value	Pr > F
Estuarine Resident	0.79	0.457
Estuarine Spawner	2.82	0.064
Top Carnivore	1.30	0.277
Benthic	2.99	0.054*
Perciformes	5.07	0.008*
Plueronectiformes	0.74	0.480
Clupeiformes	0.35	0.708
Sciaenidae	0.85	0.429
Gobiidae	4.75	0.011*
Species Richness	4.03	0.021*
Diversity	4.17	0.018*
Total Abundance	6.74	0.002*
New assignment	**F-value**	**Pr > F**
Estuarine Resident	6.24	0.003*
Estuarine Spawner	49.92	<0.001*
Top Carnivore	0.18	0.832
Benthic	67.51	<0.001*
Perciformes	184.57	<0.001*
Plueronectiformes	1.70	0.188
Clupeiformes	3.82	0.025*
Sciaenidae	0.74	0.481
Gobiidae	127.20	<0.001*
Species Richness	21.96	<0.001*
Diversity	23.15	<0.001*
Total Abundance	10.30	<0.001*

Table 2.3. Discriminant analysis results for the original group assignments and the new group assignments with misclassifications of samples in each group, total samples included in groupings, and error rates of classifications.

	Low	Moderate	High	Totals
Low	30	2	4	36
Moderate	13	21	2	36
High	20	4	12	36
Error Rates	0.417	0.167	0.667	0.417
	Low	Moderate	High	Totals
Low	19	0	0	19
Moderate	1	29	1	31
High	0	5	53	58
Error Rates	0.000	0.065	0.086	0.065

Table 2.4. Significant ANOVA results of metrics for the validation data sets from 2006 and 2007.

2006	F-value	Pr > F
Estuarine Resident	23.20	<0.001
Estuarine Spawner	78.57	<0.001
Benthic	82.58	<0.001
Perciformes	119.20	<0.001
Clupeiformes	3.43	0.033
Gobiidae	83.31	<0.001
Species Richness	32.68	<0.001
Diversity	34.85	<0.001
Total Abundance	38.17	<0.001
2007	F-value	Pr > F
Estuarine Resident	8.13	<0.001
Estuarine Spawner	37.57	<0.001
Benthic	90.34	<0.001
Perciformes	89.08	<0.001
Clupeiformes	3.51	0.033
Gobiidae	108.66	<0.001
Species Richness	49.16	<0.001
Diversity	36.71	<0.001
Total Abundance	22.09	<0.001

Table 2.5. Discriminant analysis results for the validation data sets 2006 and 2007 with misclassifications of samples in each group, total samples included in groupings, and error rates.

2006	Low	Moderate	High	Totals
Low	21	1	0	22
Moderate	6	39	0	45
High	0	0	41	41
Error Rates	0.046	0.133	0.000	0.065
2007	**Low**	**Moderate**	**High**	**Totals**
Low	14	0	0	14
Moderate	1	41	0	42
High	0	3	49	52
Error Rates	0.000	0.024	0.058	0.037

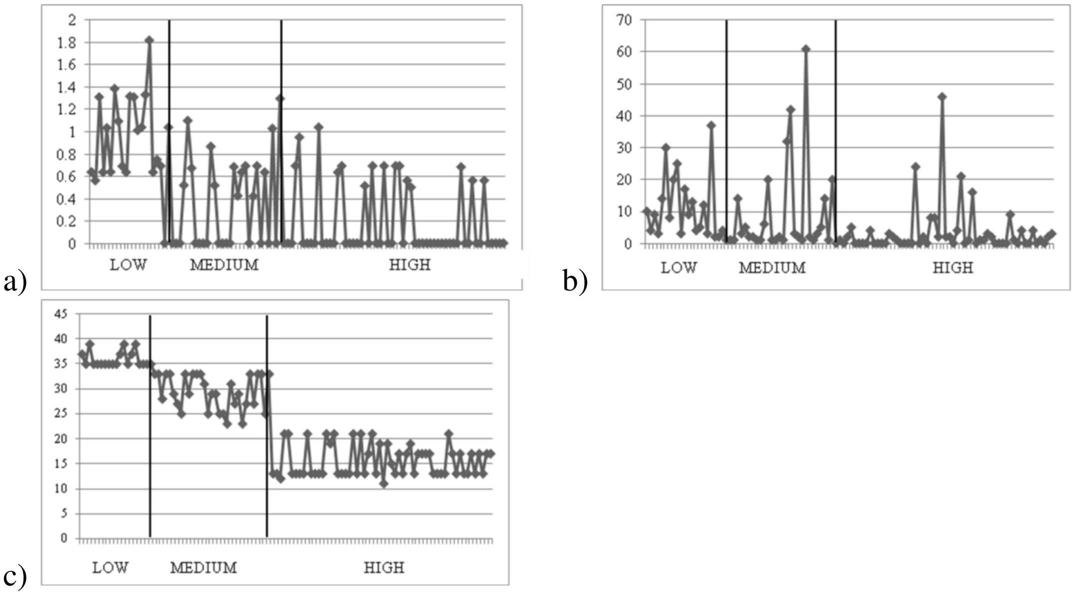

Figure 2.3. Graphs of the different levels of degradation with a) diversity (H'), b) total abundances, and c) multimetric site scores.

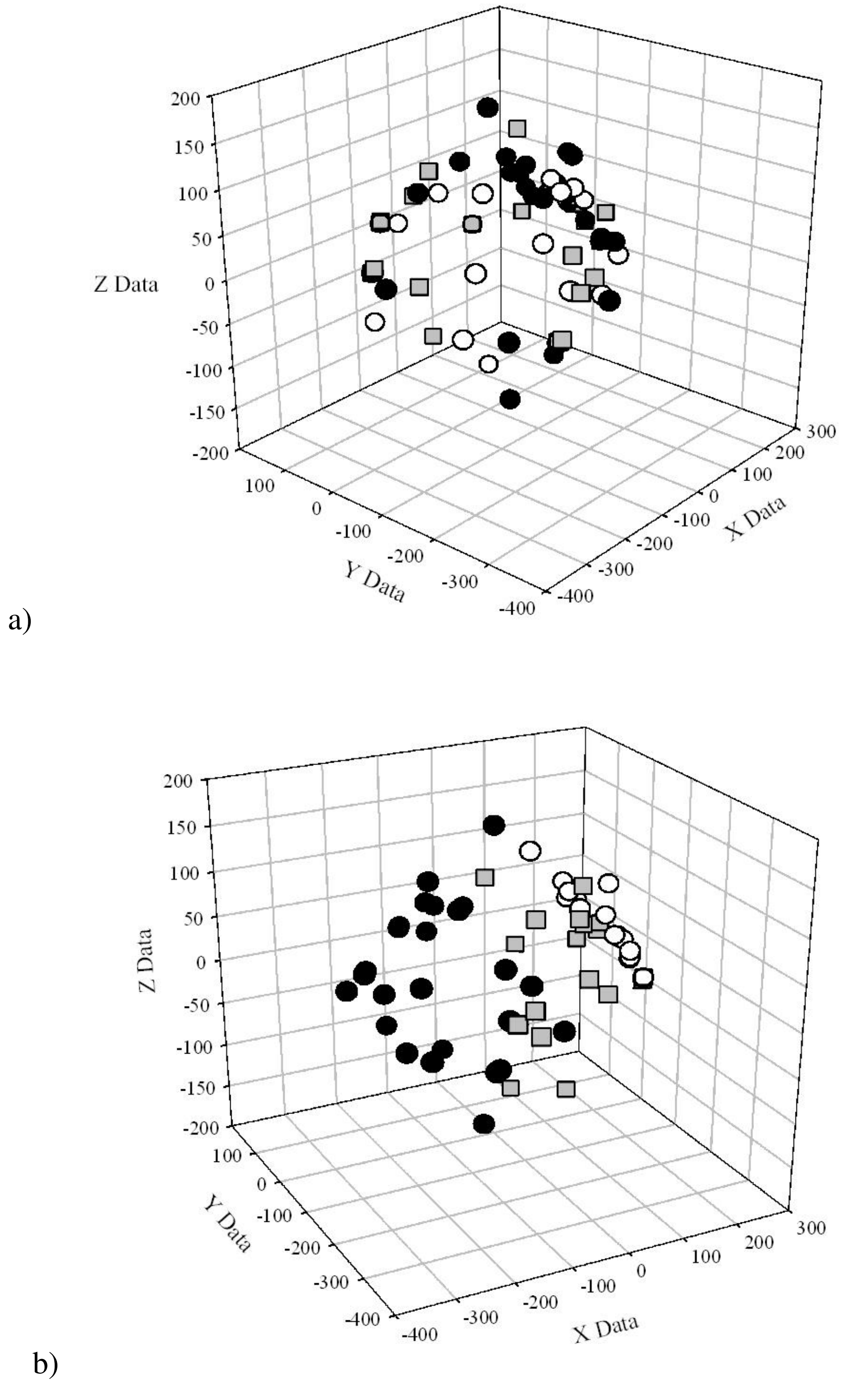

Figure 2.4. Three-dimensional MDS graphics of the sites with less degraded sites (white circles), moderately degraded sites (grey squares), and highly degraded sites (black circles) for a) the original degradation assignments and b) the new degradation assignments. Stress is 0.07 for both MDS graphs indicating they represent the data.

Figure 2.5. Map of study area with sampling locations of the three levels of degradation boxed with M = moderate degradation and H = high degradation. There were no L=less degraded locations. All locations with * were a change from degradation level and the white box indicates the location that was re-assigned as highly degraded from less degraded.

The locations were re-assigned degradation levels based on summation of scores for each site, and there was an increase in moderately degraded locations and no less degraded locations in the study area (Fig 2.5). Most less degraded locations were moderately degraded according to the IBI scores. One less degraded location was considered highly degraded with IBI scores.

DISCUSSION

Monitoring the fish community for habitat quality changes is a better alternative to more in-depth and costly physiochemical evaluation of estuaries. While toxicity and chemical pollutant sampling is useful it is expensive and hard to execute. Other monitoring programs also rely on diversity indexes or presence absence of indicator species, but these methods often overlook the biological integrity of complex ecosystems like estuaries (Miller et al. 1988, Engle et al. 1994). IBI concepts are multimetric and based on community or assemblage data. These indexes can be applied to different fish fauna in a wide variety of habitats ranging from European freshwater streams to northern Gulf of Mexico estuaries (Oberdorff and Hughes 1992, Engle et al. 1994, Engle and Summers 1999). These methods can also be used to determine if restoration actions taken result in improvements to degraded areas (Karr 1991, Oberdorff and Hughes 1992). The IBI approach is easily re-evaluated and metrics re-tested and validated especially when large scale disturbances such as oil spills and hurricanes or significant changes in flow or water temperature due to climate change are observed (Karr 1991, Engle and Summers 1999).

My IBI combined metrics from different fish community attributes used to determine how the community and numbers changed with changes to levels of degradation. While each individual metric did not show an effect of human influences, combined as a multimetric analysis they show site specific changes across differing levels of human caused stresses (Fig 2.3). The summer months were more stable and were good candidates for development of the IBI and this is a similar finding to that of Deegan et al. (1997) in Massachusetts. This was a small-scale study and applies to the lower Barataria Bay in June, July, and August and it indicates the area is moderately or highly degraded with a small amount less degraded sites. Also long-term anthropogenic influences are seen in Barataria Bay can cause continual transitions in estuarine biota as they do in streams (Horwitz 1978). That is why the index must be frequently redeveloped and evaluated to keep pace with the shifting baseline syndrome (Pauly 1995). Sampling, analyses, and development of IBIs are easily implemented. There were shifts seen in the amount of low degradation locations to moderately degraded locations and this suggests that in Barataria Bay has a small amount of high quality habitat and this habitat is difficult to visually categorize by the physical state of the marsh. The moderately and highly degraded habitats are easily evaluated visually based on readily observable marsh-edge characteristics such as marsh-edge vegetation and degree of active human exploitation. Once developed the IBI can be used by people with a variety of backgrounds and expertise to identify degraded habitats or monitor recovery and restoration efforts. The use of area specific IBI combined with other monitoring data can be used for the different portions of Barataria Bay and coastal Louisiana to help with restoration efforts and communication of site health to agencies and the public.

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CHAPTER III
SHORT-TERM EFFECTS OF AN OIL SPILL ON MARSH-EDGE FISHES AND DECAPOD CRUSTACEANS*

INTRODUCTION

Louisiana experiences a large amount of oil pollution compared to other states due to the high number and volume of oil storage facilities, production platforms, pipelines, and intense tanker and barge traffic on its waterways (Scott 2007). The Louisiana Offshore Oil Port handles approximately15% of the oil for the United States (OSPR 2003). Louisiana regularly harvests sweet crude oil that is relatively non-toxic and is characterized by high alkane concentrations, low polarity, and moderate concentrations of polycyclic aromatic hydrocarbons (PAHs) (Jackson and Pardue 1999). Sweet crude oil contains many volatile small hydrocarbons that evaporate within the first 24 to 48 hours after a spill (Kennish 1992); however, less volatile hydrocarbon fractions are sequestered by the more adsorbent silt/clay and organic sediments (Knezovich et al. 1987, Kennish 1992, Rozas et al. 2000). Though degradation is much slower for petroleum hydrocarbons that enter the deeper anaerobic zone, petroleum hydrocarbons reach deeper sediments by moving through an oxidized surficial layer where aerobic microbes more rapidly degrade them (DeLaune et al. 1990). Oil composition is important in degradation, but the size of the spill and environmental conditions at the spill site have a greater influence on how the oil will affect that area (Teal and Howarth 1984). While oil spill potential is high in Louisiana, local conditions help mitigate impacts. The highly organic sediments support abundant and varied microbial populations. The warm oxidized water and surficial sediment facilitate microbial activity, increase hydrocarbon degradation, and combine with flushing of estuaries to reduce contaminant concentrations (DeLaune et al. 1990).

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presented in Appendix B

Louisiana also supports 26% of the nation's commercial fisheries catch and has highly successful recreational fisheries dominated by *Callinectes sapidus* (blue crab), *Litopenaeus setiferus* (white shrimp), *Farfantepenaeus aztecus* (brown shrimp), *Brevoortia patronus* (gulf menhaden), and *Crassostrea virginica* (oysters) (Perry and McIlwain 1986, OSPR 2003). These high catches are due to the extensive area of Louisiana wetlands and estuaries. Louisiana fisheries overlap broadly in space and time with oil production and transportation that increases the potential for negative interactions between these economically important industries. Organisms are affected by oil mainly through filtration/ingestion, penetration, and smothering (Suni et al. 2007). Animal densities in salt marshes are reduced by short-term toxicities such as those found with small-scale oil spills (Rozas et al. 2000). Also different species and life history stages react differently to the toxic stress with less mobile benthic fishes and invertebrates suffering increased mortality. Compared to highly mobile pelagic fishes, benthic organisms are more sensitive to environmental disturbance as they are often sedentary, use contaminated food sources, and cycle nutrients and contaminants locally (Rozas et al. 2000, Chapman and Wang 2001). Benthic communities are often used by scientists to monitor effects of marine pollution (Gray et al. 1990); however, animals that are found in stressed environments which may be regularly exposed to oil contamination are likely less sensitive than animals found in more pristine environments (Rozas et al. 2000). Our interest in undertaking this study was to develop a better understanding of nekton responses to small scale oil spills in coastal Louisiana such as Barataria Bay a shallow, well-mixed estuary with strong flushing and oxidation of the sediments, and a long history of petroleum contamination (DeLaune et al. 1990, Jackson and Pardue 1999, OSPR 2003, Scott 2007).

A regularly scheduled sampling event was interrupted on 19 April 2005, by an oil spill of approximately 95.39 m^3 (600 barrels) of crude oil (Department of Transportation 2005, National

Response Center 2005) at the northern range of our study area in Barataria Bay, Louisiana. By 1700 hours on 21 April, most of the floating oil was sequestered by containment booms and removed by skimmers and absorbent pads. This quick clean-up response, and a south-southeast wind with a mean velocity of 3.9 m s^{-1}, gusting up to 8.2 m s^{-1} (NOAA station 8761724) confined the spill effects to the vicinity of Mendicant Island and northward (Fig 3.1). A diurnal tide of small amplitude, 0.24 m, was dominated by the southerly wind and did not disperse the spill to the south. On 22 April when we were able to resume sampling, it was apparent that the effects were generally contained in the northern part of our study area. The spill presented an opportunity to evaluate short-term effects of a confined spill on the abundance of fishes and decapods crustaceans in the marsh-edge nekton community. We sampled around both islands before and after the spill and applied a Before-After-Control-Impact (BACI) analysis (Smith et al. 1993, Smith 2002). We also used nonparametric analyses to explore any larger community structure effects from this oil spill event.

MATERIALS AND METHODS

Study Area

The study area is dominated by *Spartina alterniflora* along the marsh edges (0 to < 3 m from emergent vegetation) on portions of two larger islands (Mendicant and Beauregard) in the lower Barataria Bay Waterway in southeastern Louisiana (Fig 3.1). This micro-tidal estuary is shallow (mean depth is 1.50 m) and highly turbid (> 10 NTU) with seasonal salinities typically ranging from 6 to 22. Sediments vary and include organic matter, clay, silt, sand, and shell (Baltz et al. 1998, Birdsong 2004). This well-mixed estuary rarely stratifies because of its shallow depth, diurnal tides, and regular storm/wind events that govern water movement (Inoue et al. 2008). Barataria Pass is the deepest portion of the Basin (50 m) and is deeply scoured by tidal movement between the bay and the Gulf of Mexico (Krumbein and Aberdeen 1937).

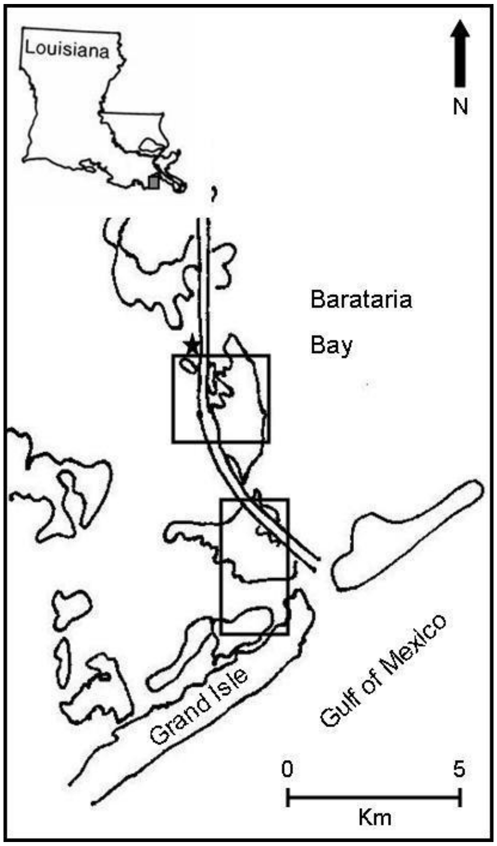

Figure 3.1. Map of the study area in Barataria Bay Waterway in southeastern Louisiana with the impacted Mendicant Island (upper box), and the control Beauregard Island (lower box), and the oil spill origin (★) at 29° 19’ 24.94” N, 89° 59’ 19.05” W.

Dominant fish species are *Anchoa mitchilli* (bay anchovy), *Micropogonias undulatus* (Atlantic croaker), *Ctenogobius boleosoma* (darter goby), and *Gobiosoma bosc* (naked goby). Decapod crustaceans using the estuary are *Litopenaeus setiferus*, *Farfantepenaeus aztecus*, *Palaemonetes* spp. (grass shrimp), *Callinectes sapidus*, and *Clibanarius vittatus* (hermit crab) (Chesney et al. 2000, Jones et al. 2002).

Field and Laboratory Methods

In three monthly sampling events 18-19 March, 9-22 April, and 7-18 May 2005, we deployed a 1.18 m^2 cylindrical fiberglass drop sampler at independent but closely juxtaposed sites (*n* = 108) in the study area. Due to the timing of the oil spill we collected 60 samples before and 48 samples after the event. We randomized sampling order to ensure the same site was not repeatedly sampled at the same time of day each month. At a site we recorded GPS coordinates, time of day, and environmental variable measurements. Water quality data (salinity, dissolved oxygen (mg l^{-1}), and temperature (ºC)) were collected using a YSI 85 meter, and water samples were collected to measure turbidity (NTU). Outside the sampler we collected sediment cores for pore-water contamination analysis, characterized sediment type, and measured mid-water column velocity (cm s^{-1}) with a Marsh-McBirney Model 2000 Flow-Mate. We also measured the distance to marsh edge (m), and we measured maximum and minimum depths in the drop sampler to estimate mean depth (m). We then used a trash pump to remove water from the sampler and filtered that water through a 333 µm mesh plankton net to ensure all fishes and decapod crustaceans were collected. Any remaining animals (e.g. epibenthic fauna) were removed by hand nets and fixed in 10% formalin. In the laboratory, we identified nekton to the lowest possible taxon and counted them. Following identification all animals were preserved in 70% ethyl alcohol. We used the sediment cores to screen for porewater toxicity on a Microtox® Model M500 analyzer, and the water column samples were analyzed for turbidity with a Hach 2100N laboratory meter.

Statistical Methods

A BACI study requires sampling prior to an event to ensure temporal control, knowledge of time and place of the event, and a control to isolate spatial effects (Green 1979, Osenberg and Schmitt 1994). It is also important for impact and control strata to be sampled simultaneously during before and after periods of a specific event (Stewart-Oaten et al. 1986, Smith et al. 1993). Sampling protocol for spring 2005 in Barataria Bay Waterway met these requirements. Hurlbert (1984) argued that impact experiments and analyses constitute pseudoreplication because of non-randomization of the impact and control sampling strata; however, when the impact is handled as a separate treatment that the researcher cannot control it can be justifiably considered random (Smith 2002). With the BACI analysis there is one before-event period, and one after-event period sampling of both impact and control strata. In two-way analysis of variance (ANOVA) evidence of a significant event effect requires a significant temporal-spatial interaction term (Stewart-Oaten et al. 1986, Underwood 1994). All analyses were conducted with the MIXED procedure in Statistical Analysis System package (SAS 2004). After examining the data (*n* = 108) for normality we pooled three closely juxtaposed sites into 36 locations and estimated mean water quality variables for each location (*n* = 36). We set α to 0.1 for all analyses because the event was an isolated incident, this experiment could not be replicated or enhanced, there was a small sample size, and we attempted to detect environmental impacts in a short amount of time (Underwood 1994, Anderson and Talley 1995). The March 18-19 and April 9-10 samples served as before-spill samples (*n* = 20), and the April22 and May 7, 16-18 samples served as after-spill samples (*n* = 16) (Table 3.1). We used the southern Beauregard Island locations as the control (*n* = 18) and the northern Mendicant Island locations as the impact (*n* = 18) treatment samples (Fig 3.1).

Table 3.1. Rank orders of taxa based on the number of individuals collected. Mean abundances (abundance/sample size) of each taxa, total individuals, fishes, and decapod crustaceans for overall (before + after), before, after, control, and impact samples.

Species/Genus	Rank	Overall	Before	After	Control	Impact
Palaemonetes pugio	1	116.61	83.05	33.56	57.72	64.39
Clibanarius vittatus	2	29.81	11.00	18.81	13.06	15.89
Callinectes sapidus	3	14.21	9.15	5.06	5.78	8.89
Ctenogobius boleosoma	4	12.21	10.15	2.06	6.28	6.83
Gobiesox strumosus	5	9.20	2.45	6.75	2.67	6.06
Callinectes similis	6	7.16	0.10	7.06	3.89	2.50
Xanthidae	7	6.10	1.60	4.50	2.89	2.89
Micropogonias undulatus	8	4.16	3.35	0.81	2.06	2.39
Mysidopsis spp.	9	3.56	0.00	3.56	2.67	0.50

Table 3.1 (Continued)

Gobiosoma bosc	10	2.48	1.60	0.88	0.72	1.83
Farfantepenaeus aztecus	11	2.70	0.45	2.25	0.28	2.22
Anchoa hepsetus	12	1.95	1.95	0.00	0.00	2.17
Anchoa mitchilli	13	1.28	0.90	0.38	0.17	1.17
Menippe adina	14	1.33	0.45	0.88	0.83	0.44
Hypleurochilus geminatus	15.50	0.93	0.30	0.63	0.33	0.56
Symphurus plagiusa	15.50	0.88	0.50	0.38	0.06	0.83
Panopeus simpsoni	17	0.66	0.35	0.31	0.39	0.28
Myrophis punctatus	18	0.48	0.10	0.38	0.11	0.33
Citharichthys spilopterus	19.50	0.40	0.15	0.25	0.06	0.33
Rhithropanopeus harrisii	19.50	0.44	0.00	0.44	0.00	0.39
Eurypanopeus depressus	21	0.30	0.30	0.00	0.17	0.17
Syngnathus louisianae	22	0.19	0.00	0.19	0.06	0.11
Synodus foetens	23.33	0.16	0.10	0.06	0.06	0.11
Alpheus spp.	23.33	0.18	0.05	0.13	0.00	0.17
Mugil curema	23.33	0.18	0.05	0.13	0.06	0.11
Brevoortia patronus	26	0.10	0.10	0.00	0.11	0.00
Persephona mediterranea	27.17	0.05	0.05	0.00	0.00	0.06
Syacium papillosum	27.17	0.05	0.05	0.00	0.00	0.06
Gobiidae	27.17	0.06	0.00	0.06	0.00	0.06
Menidia beryllina	27.17	0.06	0.00	0.06	0.00	0.06
Menticirrhus americanus	27.17	0.06	0.00	0.06	0.00	0.06
Penaeid	27.17	0.06	0.00	0.06	0.06	0.00
Total Individuals		218.11	128.30	89.81	124.00	98.39
Total Fishes		34.75	21.75	13.00	18.17	17.56
Total Decapods		183.29	106.60	76.69	105.89	80.72
Sample Size		36	20	16	18	18

Nine separate BACI analyses were run on the total number of individuals (e.g., fishes and decapod crustaceans), the total number of fishes, the total number of decapod crustaceans, and individually on six sensitive species. Treatments were temporal (before and after) and spatial (control and impact) as main effects, and their interactions were tested as the event effects.

The use of nonparametric analyses has increased, to avoid problems with assumptions that occur in most ecological community data (Smith et al. 1993). Because of the dynamic nature of estuarine faunal communities, community structure generally shows a more complete picture of the effect and distribution of a pollutant than number of individuals or species data alone

(Gray et al. 1990). We used the Plymouth Routines In Multivariate Ecological Research (PRIMER) package on the pooled data set (n = 36) to relate the impact of the oil spill to changes in species composition at the community level (Clarke and Warwick 2001). The multidimensional scaling analysis (MDS), analysis of similarity (ANOSIM), BIOENV, and similarity percentages with species composition (SIMPER) procedures were run using Bray-Curtis similarity matrix after square-root transforming the species data. By down weighing the abundant species, this transformation allows similarities to depend on both abundant and rare species (Clarke and Warwick 2001). An MDS is a graphical representation of biological relationships among samples. Closely juxtaposed samples in three-dimensional graphical space have more similar species compositions, while samples farther from each other have more dissimilar compositions (Clarke and Warwick 2001). The ANOSIM tests whether samples differ between treatments and produces a statistic (R), which ranges -1 to 1 and reflects observed differences between treatments contrasted to differences within treatments. A strongly positive R approaching 1 indicates samples within a treatment are more similar than samples between treatments. An R of 0 indicates no difference between treatments. And, a strongly negative R approaching -1 indicates samples between treatments are more similar than samples within the same treatments. When the null hypothesis fails to be rejected and there is no effect between treatments the redistribution of sample treatment labels would make no difference to the R statistic therefore the test would not be significant (Clarke and Warwick 2001). The BIOENV procedure calculates the correlation coefficient between two similarity matrices (species and environment) and identifies the environmental variables that explain the community patterns in the species matrix (Clarke and Warwick 2001). The SIMPER test shows which species contribute to the multivariate patterns detected; species mean abundances are compared and the contribution of species to the Bray-Curtis measures of similarity and dissimilarity are examined

(Ehrich and Stransky 1999). These tests indicate how the nekton community structure and specific taxa responded to the oil spill.

RESULTS

Parametric Analyses

We quantitatively characterized species composition and environmental conditions at 36 locations covering 127.44 m^2 of marsh-edge habitat. Twenty-seven species and five higher taxa comprised the 4001 individuals collected. Decapod crustaceans dominated the collection with 3358 individuals, whereas there were only 643 fishes. The BACI analyses of the total number of individuals, total number of fishes, and total number of decapod crustaceans showed no significant differences for the main effects ($P > 0.10$). There were significant interactions (BA*CI) ($P < 0.09$) for the total number of individuals ($F_{1,32} = 3.09$, $P < 0.09$) and total number of fishes ($F_{1,32} = 3.06$, $P < 0.09$), indicating specific event effects on abundances. Before- and control-treatment samples yielded consistently greater abundances of total number of individuals, total number of fishes, and total number of decapod crustaceans than the after- and impact-treatment samples (Table 3.1). The interaction term abundances show before-control abundances were consistently higher than after-impact abundances. Because six species (*Palaemonetes pugio, Clibanarius vittatus, Ctenogobius boleosoma, Callinectes sapidus, Callinectes similis,* and *Gobiesox strumosus* (skilletfish)) dominated and drove similarities within treatments, we ran separate species-level BACI analyses on each. Significant interactions ($F_{1,32} = 2.90$ and $F_{1,32} = 3.81$, $P \leq 0.10$) indicated potential spill effects for only two species, *P. pugio* and *C. boleosoma*. Tests on the other four species were not significant for interactions or main effects ($P > 0.10$). There was some temporal and spatial variation with seven environmental variables (salinity, dissolved oxygen, temperature, turbidity, mean water velocity, distance to marsh edge, and mean depth) tested with BACI analyses. Distance to marsh edge had the only significant interaction, ($F_{1,32} = 7.37$, $P = 0.01$) (Table 3.2). Microtox analysis on the porewater from sediments was generally non-toxic with only five of 108 samples indicating some contamination. These five samples were spread fairly evenly among all treatments (before = 3, after = 2; control = 3, impact = 2), so trends could not be detected.

Table 3.2. Mean environmental data with standard error for the BACI analysis with * signifying a significant ($P < 0.1$) interaction, † signifying a significant ($P < 0.1$) temporal, and ‡ spatial effect, respectively.

Treatments	Temporal		Spatial	
	Before	After	Control	Impact
Salinity	17.3 ± 1.63	18.4 ± 0.23	18.0 ± 01.19	17.8 ± 1.15
Dissolved Oxygen (mg l^{-1})	7.4 ± 0.29†	8.8 ± 0.58†	8.8 ± 0.51‡	7.4 ± 0.40‡
Temperature (°C)	19.6 ± 0.75†	27.1 ± 0.42†	22.9 ± 0.62	23.8 ± 0.60
Turbidity (NTU)	27.3 ± 3.60	27.3 ± 2.23	23.8 ± 2.51	30.8 ± 3.50
Mean Water Velocity (cm s^{-1})	7.9 ± 0.67†	5.0 ± 1.02†	6.7 ± 1.01	6.3 ± 0.67
Distance to Marsh Edge (m)*	2.5 ± 0.28	2.3 ± 0.24	2.0 ± 0.23‡	2.8 ± 0.28‡
Mean Depth (m)	0.4 ± 0.01	0.4 ± 0.03	0.4 ± 0.03	0.4 ± 0.02

Nonparametric Analyses

The three-dimensional MDS plots of community data showed separation between groupings of before (March and April) and after (April and May) samples with the before samples (circles) generally clumping together and away from the after samples (triangles) (Fig 3.2). There were no separate groupings of control and impact samples as the control samples (black shapes) intermingled with the impact samples (white shapes) (Fig 3.2).

Also this analysis did not show any intermingling of main effects for a visual representation of an interaction among treatments. This agrees with the ANOSIM as it revealed a small but significant ($P = 0.001$) effect on overall community structure from the temporal treatment producing a global R test statistic of 0.25, yet there was no significant R statistic for the

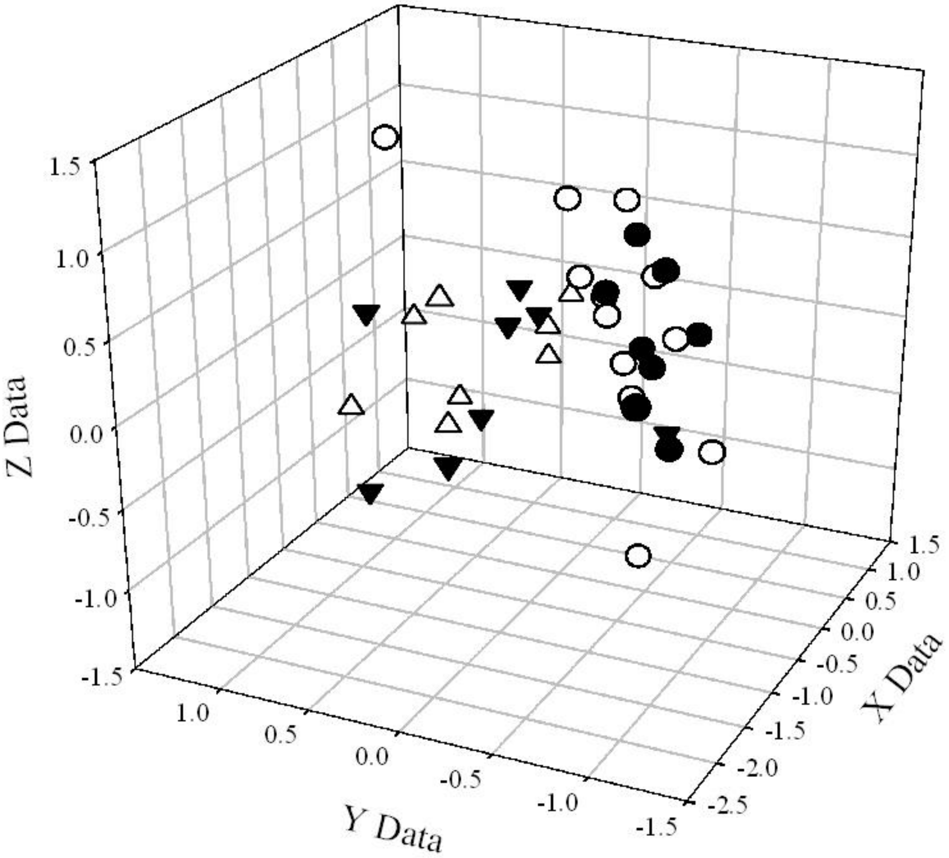

Figure 3.2. Three-dimensional MDS graphics of the sample species data for the temporal (before ○● and after Δ▼) and spatial (control ●▼ and impact ○Δ) treatments. The stress value for the graph is 0.13 indicating this is a good three-dimensional representation of these data (Clarke and Warwick 2001).

spatial treatments. The environmental variables did not explain much of the community data. While dissolved oxygen had the highest correlation coefficient ($p_s = 0.12$), it and the other environmental variables did not significantly explain the variation in the community data (P = 0.55 global BEST permutation test). The SIMPER result of mean similarity within before samples was produced by the mean abundances of *Palaemonetes pugio*, *Clibanarius vittatus,* and *Callinectes sapidus* (79%), and after mean sample similarity was largely due to *P. pugio, C. vittatus,* and *Callinectes similis* (49%). The temporal dissimilarity between treatments was due partly to the after treatment appearance and high abundance of *C. similis* (10%). For the spatial treatments approximately three-fourths of the mean similarity within control samples (77%) and within impact samples (73%) were due to the same three species. The mean dissimilarity found between control and impact treatments was due in part to the abundance of *Ctenogobius boleosoma* (8%).

DISCUSSION

The BACI analyses showed immediate effects from the oil spill on the total number of individuals (fishes and decapod crustaceans), total number of fishes, *Ctenogobius boleosoma*, and *Palaemonetes pugio*. This suggests that many of the less abundant fishes and a few more mobile decapod crustaceans reacted to the oil spill, while many of the more sessile and benthic decapod crustaceans remained in place after the event (Gray et al. 1990, Rozas et al. 2000). *Ctenogobius boleosoma* was primarily responsible for the dissimilarity between the spatial (control and impact) treatments and was significantly ($P = 0.06$) affected by the oil spill event. While *C. boleosoma* is a dominant fish species in lower Barataria Bay, it showed a spill effect and has not shown an ability to acclimate or adapt to petroleum hydrocarbons in sediments (Klerks et al. 1997, Rozas et al. 2000, Klerks 2002). *Ctenogobius boleosoma* was the most abundant fish species and its count data drove the significant ($P < 0.09$) interaction for total number of fishes seen in the original BACI analysis. Overall the most abundant species in the community was *P. pugio*, and it also had a significant ($P = 0.10$) interaction in the single species BACI. Nevertheless, the *P. pugio*'s influence on the decapod crustacean BACI analysis was overwhelmed by the many other abundant decapod crustaceans in the community. *Callinectes similis* was most sensitive to the temporal treatment. Its response may have been driven by a significant increase in temperature ($P < 0.001$) which may be the signal for the lesser blue crab to enter the estuary (Das and Stickle 1994). *Callinectes similis* will recruit to marsh sites from open water areas in late spring to early fall based on temperature and salinity changes (Gibson 1991, Hsueh et al. 1993). Salinity is a primary influence on C. *similis* distribution, but salinity was not significantly different for this study and it was well within the optimal salinity (15-20) range for the species (Gibson 1991). After deleting *C. similis* data, a re-analysis did not affect our conclusions as total number of individuals ($F_{1,32} = 3.30$, $P = 0.08$) and total number of decapod crustaceans ($F_{1,32} = 2.49$, $P = 0.12$) did not change appreciably. While there were temporal differences detected in four of the six species BACI analyses, these possible seasonal shifts reinforce the necessity of finding a significant interaction term to identify a spill event effect (Stewart-Oaten et al. 1986, Underwood 1994).

The nonparametric analyses on species composition results showed small but significant differences between temporal treatments, but not between spatial treatments. Although the total number of individuals decreased due to the oil spill event (i.e. BACI), the species composition remained relatively constant. The Barataria Basin has a long history of anthropogenic influences and the community structure of the fauna now common to the area could be less sensitive to pollutants by acclimation or adaptation over time (Klerks et al. 1997). Population numbers are more variable while community structure is more stable therefore community structure is more powerful for showing environmental perturbation (Gray et al. 1990, Osenberg et al. 1994); however, to detect clear difference between treatments, these nonparametric analyses require a larger sample size than we had.

The environmental variables (salinity, dissolved oxygen, temperature, turbidity, mean water velocity, distance to marsh edge, and mean depth) were not expected to change significantly in the short-term with an isolated oil spill event as they are physical variables not chemical. Nevertheless distance to marsh edge had a significant event effect, but the mean difference was only 0.8 m and probably not important biologically. Blue crabs move more than a meter in one minute (Das and Stickle 1994), grass shrimp have an escape response greater than 0.4 m (Goddard and Forward 1991), and larger shrimp have response mean velocities on the order of 1 m s^{-1} (Daniel and Meyhöfer 1989). This assessment was also corroborated by the BIOENV procedure results in that the environmental variables did not significantly explain differences in biological data.

An opportunity for longer-term testing of the spill effects was precluded by an active hurricane season with Hurricane Cindy in July, Hurricane Katrina in August, and Hurricane Rita in September. These storms may have decreased the level of water column toxicity and surficial substrate contamination in the study area by widely dispersing or burying the remaining oil constituents deep in the sediments, much as natural accretion would in a building marsh (Jackson and Pardue 1999, Turner et al. 2006). Nevertheless, no long-term effects were detected when we examined data from the same locations in 2006 (Roth, unpublished data). Many simultaneous perturbations occurring in this dynamic estuary making it difficult to identify individual causal factors that might be responsible for differences in abundances or species composition, but this short-lived oil spill allowed us to test effects from a specific perturbation on biological and environmental data. Notably the spill event did have a detectable short-term and localized effect only on numbers of individuals and fishes, and the existing community structure seems to be robust.

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CHAPTER IV
HURRICANE KATRINA: EFFECTS ON AND RATE OF RECOVERY OF A MARSH-EDGE NEKTON COMMUNITY

INTRODUCTION

Tropical storms are intermediate disturbances that flush coastal systems from year to year, and therefore have large influences on water quality and nekton communities in northern Gulf of Mexico wetlands (Hagy et al. 2006). The frequency of tropical storms making landfall in the southeastern portion of the United States has increased since 1996 and this trend is hypothesized to continue for years to come (Goldenburg et al. 2001, Emanuel 2005, Webster et al. 2005, Greening et al. 2006). While hurricanes can change the physiochemical characteristics of wetlands, these disturbances are relatively short-term in natural systems (Tilmant and Curry 1994, Paerl et al. 2006, Stevens et al. 2006). Areas of high anthropogenic influences are more susceptible to increased destruction from the storms, because they are already in an altered state of existence (Mallin and Corbert 2006). This is evident in the still struggling portions of the Florida panhandle from Ivan in 2004 and coastal Mississippi, Louisiana, and Texas from the 2005 Hurricane season.

Hurricane Katrina ravaged the Louisiana and Mississippi coasts in late August 2005. Louisiana’s wetlands are fragile and some of the most economically important areas in Gulf of Mexico for the fisheries and petro-chemical industries (OSPR 2003). Hurricanes can erode marshes, move sediments and vegetation, change salinities, nutrient input, circulation, and cause mortality and displacement of local flora and fauna (Conner et al. 1989, Mallin et al. 1999, Greenwood et al. 2006). Vegetation such as mangroves and submerged aquatic vegetation may remain affected for up to a year after storm passage (Milbrandt et al. 2006, Maiaro 2007). Short-term storm effects on the estuarine fauna can be damaging, but populations are hypothesized to return to normal population numbers with the passage of time (Knott and Martore 1991, Tilmant and Curry 1994, Greenwood et al. 2006). Because of the profitable fisheries industry in southeastern Louisiana, there is a need to understand relationships of the nekton assemblages of fishes and macroinvertebrates to their environment. To understand the possible resilience of community structure following storms in a Louisiana estuary, I used three years of data and quantified changes in species composition before and one and two years after Hurricanes Cindy and Katrina.

MATERIALS AND METHODS

Study Area

Nekton were sampled near two islands (Beauregard Island and Mendicant Island) and an uninhabited portion of eastern Grand Isle, in lower Barataria Bay of southeastern Louisiana (Fig 4.1). While the sample locations are not inhabited by humans, they are constantly perturbed by boat traffic from commercial and recreational fisheries and industries. The study area is shallow with extensive marsh edge habitat dominated by *Spartina alterniflora* (smooth cordgrass) and *Avicennia germanins* (black mangrove) is on Barataria Bay Waterway and near Barataria Pass, which connects the basin to the Gulf of Mexico. The study area is subjected to periodic storm events that bring in sea water from the Gulf and flush the estuary a few times a year.

There were several storm events in 2005 including Hurricane Cindy in July and Hurricane Katrina in August. Hurricane Cindy reduced in intensity and made land fall as a tropical storm on 6 July 2005 west of Grand Isle with 31.3 m s^{-1} sustained winds and a storm surge of 1.2 to1.8 m (Stewart 2006) (Fig 4.2). Hurricane Katrina made land fall as a category 3 hurricane on the Saffir-Simpson scale on 29 August 2005 east of Grand Isle in Plaquemines Parish with sustained winds of 56.6 m s^{-1} and a storm surge ranging 7.3 to 8.5 m (Knabb et al. 2006) (Fig 4.2). Both of these storms physically moved portions of the marsh and deposited human debris on the existing marsh. Because Hurricane Katrina was the stronger storm 2006 and 2007 data are hereby referred to as one year post- and two years post-Katrina.

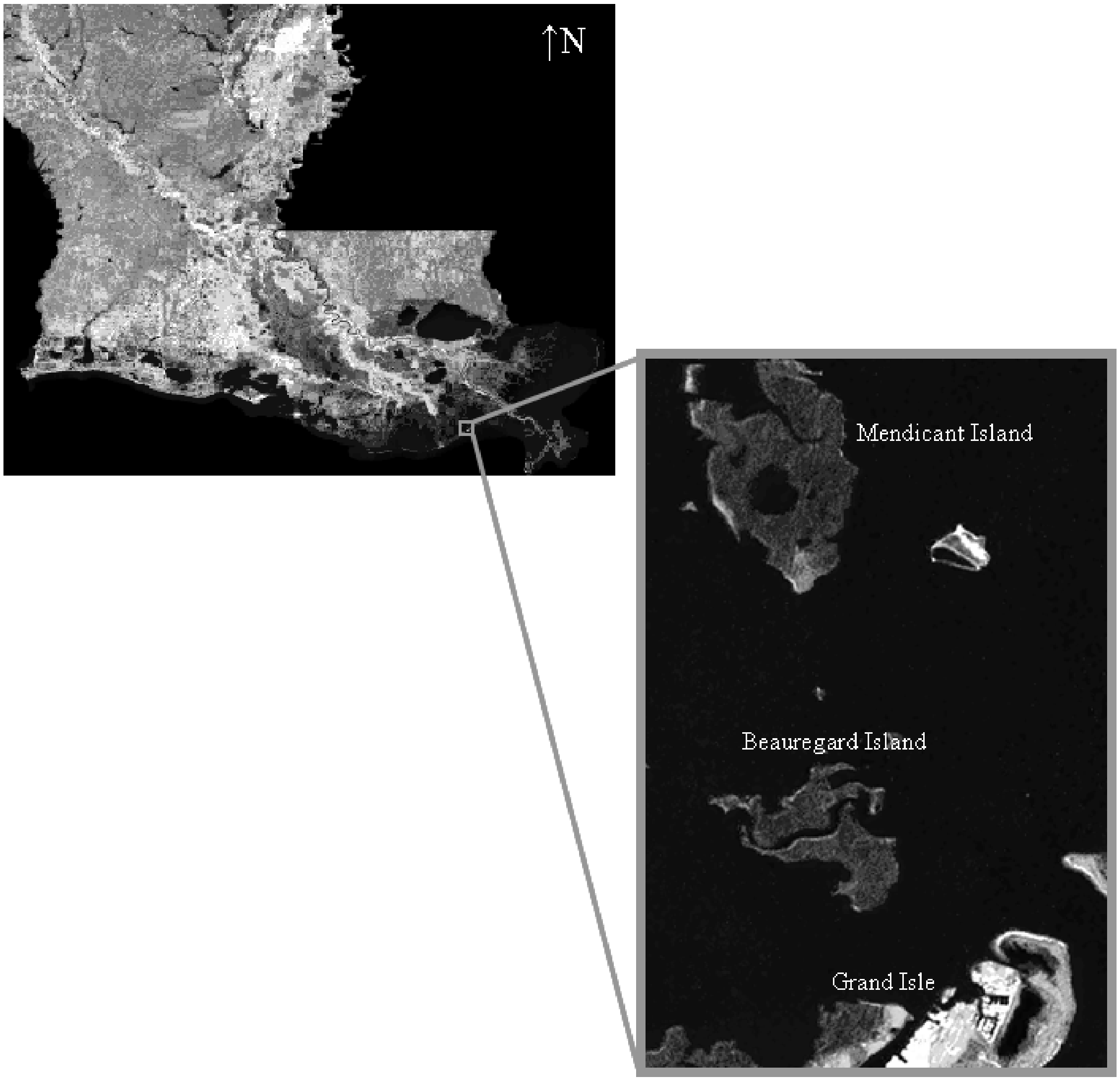


Figure 4.1. Map of Louisiana with study area boxed and enlarged to show Mendicant Island, Beauregard Island, and Grand Isle. Images from LSU Atlas.

Figure 4.2. The paths of Tropical Storm Cindy (dash line) and Hurricane Katrina (white line) with the study area in Barataria Bay in the white box. Image from Google Earth.

Field and Laboratory Methods

In twelve locations, three sites were randomly sampled monthly from March through August (*n* = 216) for each of three years: 2005 (pre-Katrina), 2006 (1 year post-Katrina), and 2007 (2 years post-Katrina) and a total of 648 samples. I deployed a 1.18m^2 cylindrical drop sampler in close proximity to the salt marsh with the purpose to quantitatively characterize the small nekton community of the marsh-edge. This sampling encompassed 764.6 m^2 of the *Spartina* dominated marsh edge habitat. At each site I recorded GPS coordinates, time of day, distance to marsh edge, dominant substrate type, and marsh edge vegetation type. Once the sampler was securely seated, I measured the water quality variables with a YSI 85 meter. Mean water depth (m) was recorded and mean water velocity (cm s^{-1}) was measured. I also collected 50 ml of water from inside the sampler to analyze turbidity (NTU), and 50 ml of sediment was collected just outside the sampler to test for pore water toxicity in the laboratory. I evacuated water from the sampler and collected all remaining nekton with nets. In the laboratory I indentified all individuals to lowest possible taxon and counted them. Turbidity and pore-water toxicity were analyzed with Hach 2100N and Microtox® Model M500, respectively.

Statistical Methods

I related impacts from the Hurricanes to changes in species composition at the community level with nonparametric analyses in PRIMER (Clarke and Warwick 2001) with three procedures. To control for temporal differences, spring and summer data were analyzed separately. I made pairwise post-hoc comparisons among years (2005, pre-Katrina; 2006, one year post-Katrina; 2007, two years post-Katrina) with analysis of similarity (ANOSIM), evaluated differences among samples graphically with multidimensional scaling (MDS), and determined which species contributed to the multivariate patterns indicated with the similarity percent procedure (SIMPER) (Clarke and Warwick 2001). These analyses and procedures were run on Bray-Curtis similarity matrices after root transforming species data. ANOSIM tests whether samples differ between treatments and produces a global R statistic. After a significant ($P < 0.05$) global R indicates differences among all groups, post-hoc pairwise comparisons indicate differences between each group. The R statistic ranges from -1 to 1 and reflects the sample differences among treatments versus the sample differences within treatments. A significant R of 0 indicates no differences in samples among all treatment groups. As R approaches 1 samples show differences among treatments, and as R approaches -1 samples are more similar to samples in other treatment groups than to those in their own treatment group. MDS plots are graphical depictions of the biological relationship among samples. Closely juxtaposed clusters of samples plotted in three-dimensional space are the representation of samples with similar species composition. A stress value is generated as a test of goodness of fit, and if the stress value is less than 0.2 the graphic is a good representation of data. The SIMPER tests which species contribute to the multivariate patterns detected within the samples. Species mean abundances are compared and their contributions to similarity and dissimilarity measurements and are examined (Ehrich and Stransky 1999).

The ten most abundant taxa (cumulatively > 85% of total abundance) were identified as strong contributors to the dissimilarity among years in the SIMPER procedure. Therefore, I ran ANOVAs in SAS (2004) on the log (X+1) transformed total numbers of individuals for those taxa to determine if there were significant differences among pre- and post-Katrina years for the number of individuals in each taxa. Tukey's post-hoc pairwise analyses were performed on these data. PRIMER MDS analyses were followed with parametric MANOVAs to determine if clusters created were significantly different.

RESULTS

Spring

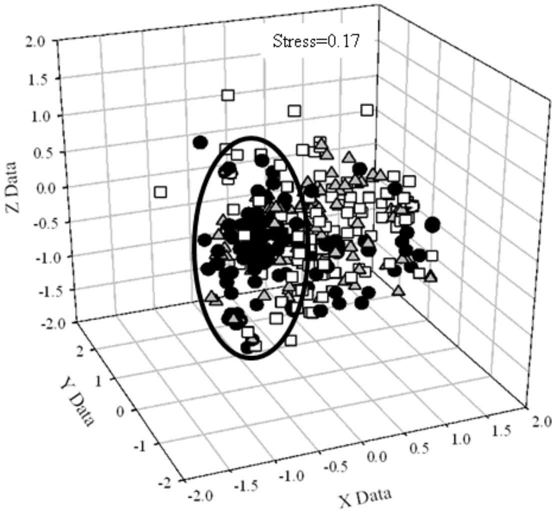
A total of 13,897 individuals in 54 species were collected for March, April, and May in 324 samples. One year post-Katrina (2006) had 5,892 individuals and this was higher than pre-Katrina (2005) and two years post-Katrina (2007) with 3,944 and 4,061 individuals, respectively. These data all showed significant changes in community composition of samples pre- and post- Katrina. The ANOSIM was significant ($P < 0.001$) for total species, fish species, and decapod crustacean composition, and environmental variables. For the pre-Katrina (2005) and one year post-Katrina (2006) the largest separations were for the fish species composition data with R statistics of 0.23 (Table 4.1). The largest year difference ($R = 0.25$) was between pre-Katrina (2005) and one year post-Katrina (2006) for environmental data (Table 4.1).

Table 4.1. Spring ANOSIM pairwise comparison results for all species, fishes, and decapod composition and environmental variables between 2005 (pre-Katrina) and 2006 (one year post-Katrina), and 2005 (pre-Katrina) and 2007 (two years post-Katrina). These were all significant ($P < 0.001$).

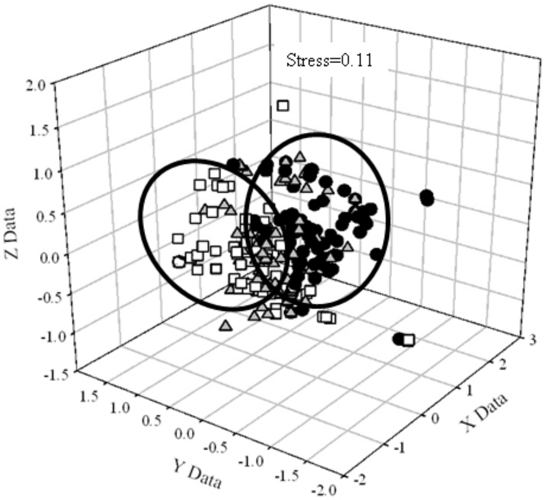
Variables	Years	*R*-statistic
Overall Species Composition	1 year post-Katrina	0.15
	2 years post-Katrina	0.10
Fish Species Composition	1 year post-Katrina	0.23
	2 years post-Katrina	0.09
Decapod Species Composition	1 year post-Katrina	0.13
	2 years post-Katrina	0.10
Environmental Variables	1 year post-Katrina	0.25
	2 years post-Katrina	0.07

The three-dimensional MDS analysis on all species data has pre-Katrina (2005) samples clumped closer and somewhat removed from both groups of post-Katrina (2006/2007) samples (Fig 4.3). This same trend of pre-Katrina (2005) samples separating from other samples was evident in the fish species, decapod crustacean species and environmental variable data (Fig 4.3). All MDS plots had significant ($P < 0.0001$) MANOVAs for dimensions indicating significant clustering of year groupings. Stress values were less than 0.2 indicating the MDS plots were good representations of these data.

The ten most abundant species were *Palaemonetes* spp., *Brevoortia patronus*, *Clibanarius vittatus*, *Callinectes similis*, *C. sapidus*, *Micropogonias undulatus*, *Ctenogobius boleosoma*, *Farfantepenaeus aztecus*, *Gobiesox strumosus*, and *Pagurus longicarpus* (Table 4.2). These ten species alone contributed to 95% of the overall abundance measured in spring. From the SIMPER procedure, the largest dissimilarity among samples (72 %) was between pre-Katrina (2005) and one year post-Katrina (2006), and the abundant species with xanthids contributed to 84 % of this dissimilarity. Dissimilarity of two year post-Katrina (2007) and the other years was each 70% with the same species contributing.



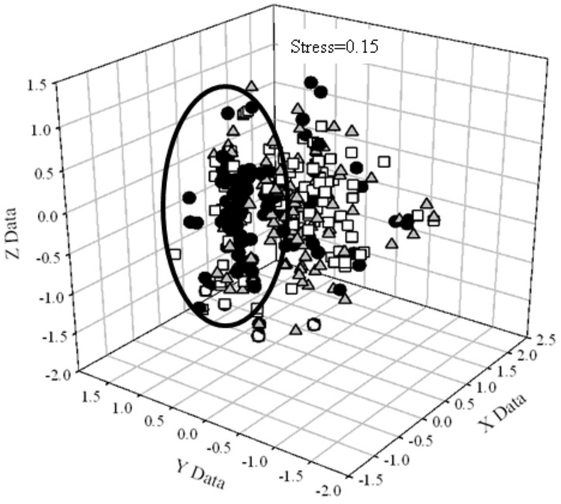
a)



b)

Figure 4.3. Three-dimensional MDS of spring samples with stress values. Samples of pre-Katrina (2005) (●), one year post-Katrina (2006) (□), and two years post-Katrina (2007) (▲) samples for a) all species, b) fishes, c) decapod crustaceans, and d) environmental variables. Tight clusters of samples are encircled in black.

Figure 4.3 (Continued)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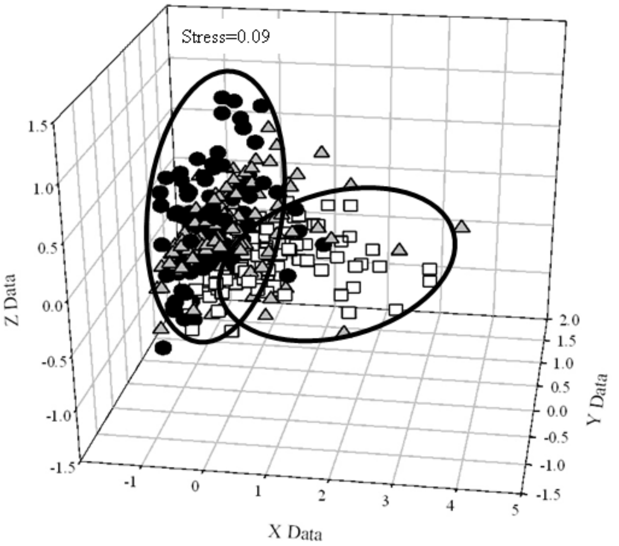
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d)

The ANOVAs on these abundant species from the SIMPER results showed pre-Katrina (2005) means were significantly ($P < 0.05$) higher than post-Katrina for *Palaemonetes* spp., *C. boleosoma*, and *G. strumosus* (Table 4.2). Means were significantly higher one year post-Katrina (2006) than pre- Katrina for *B. patronus*, *C. similis*, *F. aztecus*, and *P. longicarpus* (Table 4.2). *Farfantepenaeus aztecus* and *P. longicarpus* had significantly higher abundances one year post-Katrina but decreased two years post- Katrina (2007) (Table 4.2). *Gobiesox strumosus* were not collected in one year post-Katrina samples, but the species reappeared in smaller abundances than pre-Katrina two years post-Katrina (Table 4.2). Mean water temperature and velocity, and turbidity were significantly higher post-Katrina than pre-Katrina (Table 4.2). Mean water depth and dissolved oxygen significantly decreased one year post-Katrina, but returned to pre- Katrina levels two years post- Katrina (Table 4.2).

Summer

I collected a total of 9,984 individuals in 57 taxa from the 324 samples in June, July, and August with equal efforts across years and months. Again one year post-Katrina (2006) had the greatest number of individuals collected with 4,468. Pre-Katrina (2005) had 2,572 and two years post-Katrina (2007) had 2,944 individuals collected in their samples. According to ANOSIM results, summer analyses were less differentiated among year groupings than spring data. All ANOSIM results were significant ($P < 0.001$), and the largest separation among year groupings was with the fish data and it showed a difference between pre-Katrina (2005) and one year post-Katrina (2006) samples with pairwise R statistics of 0.13 (Table 4.3).

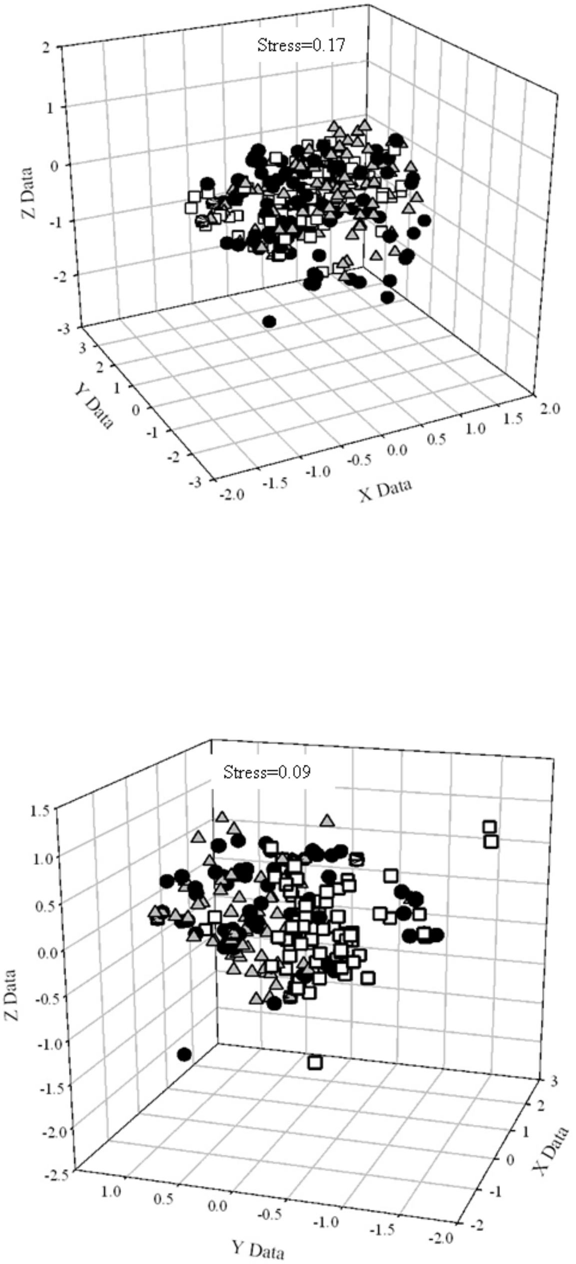
The lack of separation among years is evident with the MDS plots (Fig 4.4). While MANOVA results for all MDS dimensions were significantly different ($P < 0.001$) and stresses were less than 0.2, these samples did not cluster into separate groups among the different years.

Table 4.2. The means and ±2 standard error for the ten most abundant spring species and the continuous environmental variables with a, b, and c indicating significant differences ($P < 0.05$) from the post hoc Tukey's analyses.

Taxa	Common Name	2005	2006	2007
Palaemonetes spp.	grass shrimp	2.22±0.14^{a}	1.38±0.14^{b}	1.62±0.14^{b}
Brevoortia patronus	gulf menhaden	0.01±0.01^{a}	0.47±0.12^{b}	0.21±0.06^{c}
Clibanarius vittatus	thin striped hermit crab	1.17±0.10	1.23±0.10	1.04±0.10
Callinectes sapidus	blue crab	0.88±0.08^{a}	0.78±0.08^{a}	1.21±0.10^{b}
C. similis	lesser blue crab	0.41±0.07^{a}	1.23±0.09^{b}	1.35±0.12^{b}
Micropogonias undulatus	Atlantic croaker	0.21±0.06	0.02±0.07	0.24±0.05
Ctenogobius boleosoma	darter goby	0.62±0.08^{a}	0.36±0.05^{b}	0.58±0.07^{a,b}
Farfantepenaeus aztecus	brown shrimp	0.20±0.04^{a}	0.39±0.07^{b}	0.23±0.05^{a,b}
Gobiesox strumosus	skilletfish	0.53±0.07^{a}	0.00±0.00^{b}	0.21±0.05^{c}
Pagurus longicarpus	long wrist hermit crab	0.00±0.00^{a}	0.41±0.07^{b}	0.18±0.05^{c}
Environmental Variables		2005	2006	2006
Mean depth (m)		40.95±1.12^{a}	34.04±1.31^{b}	41.00±1.46^{a}
Temperature (°C)		22.96±0.44^{a}	24.37±0.19^{b}	24.71±0.26^{c}
Salinity (psu)		17.54±0.50^{a}	24.85±0.23^{b}	16.26±0.18^{c}
Dissolved Oxygen (mg l^{-1})		8.09±0.21^{a}	7.10±0.12^{b}	8.28±0.19^{a}
Mean water velocity (cm s^{-1})		2.37±0.16^{a}	6.80±0.42^{b}	4.45±0.58^{c}
Turbidity (NTU)		27.53±1.65^{a}	55.14±4.45^{b}	43.61±4.24^{b}
Distance to marsh edge (m)		1.9±0.13^{a}	1.9±0.22^{a}	1.12±0.11^{b}

Table 4.3. Summer ANOSIM pairwise comparison results for all species, fishes, and decapod composition and environmental variables between 2005 (pre-Katrina) and 2006 (one year post-Katrina), and 2005 (pre-Katrina) and 2007 (two years post-Katrina). These were all significant ($P < 0.001$).

Variables	Years	*R*-statistic
Overall Taxa Composition	1 year post-Katrina	0.07
	2 years post-Katrina	0.04
Fish Taxa Composition	1 year post-Katrina	0.13
	2 years post-Katrina	0.02
Decapod Taxa Composition	1 year post-Katrina	0.04
	2 years post-Katrina	0.04
Environmental Variables	1 year post-Katrina	0.02
	2 years post-Katrina	0.07

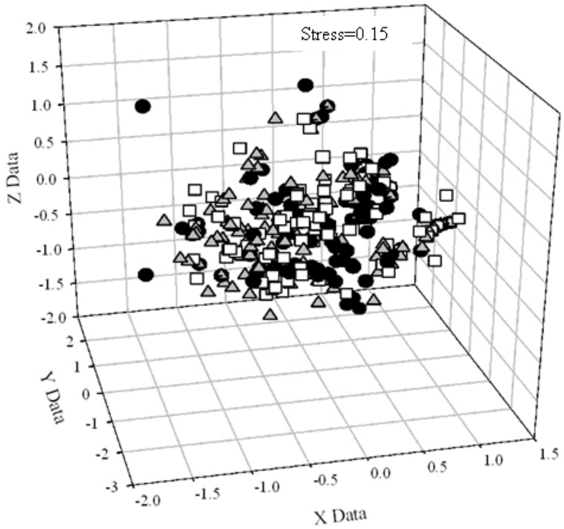


a)

b)

Figure 4.4. Three-dimensional MDS of summer samples with stress values. Samples of pre-Katrina (2005) (●), one year post-Katrina (2006) (□), and two years post-Katrina (2007) (▲) samples for a) all species, b) fishes, c) decapod crustaceans, and d) environmental variables.

Figure 4.4 (Continued)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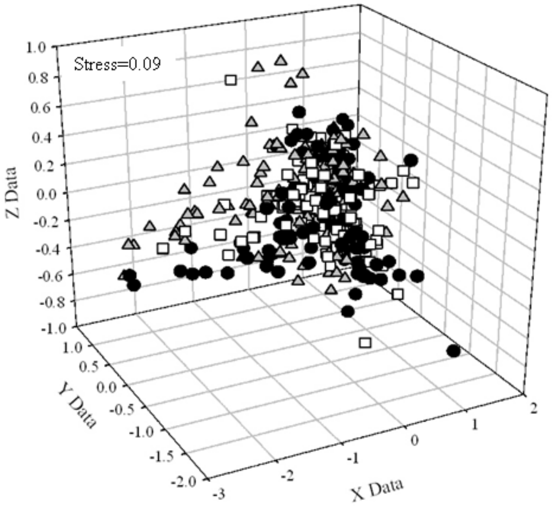
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X Data
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d)

Only the fish taxa samples showed a slight clustering and separation of one year post-Katrina (2006) samples from the other samples (Fig 4.4.b).

The dissimilarities between pre-Katrina (2005) and one and two years post-Katrina were nearly the same, 75 and 76% respectively. The ten most abundant taxa plus penaeid shrimp contributed to 79% of the dissimilarity found between pre- and post- Katrina samples. *Palaemonetes* spp., *C. vittatus*, *C. sapidus*, xanthid, *Anchoa mitchilli*, *C. boleosoma*, *C. similis*, *G. strumosus*, *Panopeus simpsoni*, and Gobiidae abundances were 88% of the total of individuals collected in the summer samples. Because the SIMPER procedure showed these species to drive the multivariate patterns, I ran separate ANOVAs on them.

The ANOVA results indicated the greatest difference was between pre-Katrina (2005) and one year post-Katrina (2006). There were significant ($P < 0.05$) increases in the mean abundances for *Palaemonetes* spp., *C. vittatus*, *C. sapidus*, xanthids, *C. boleosoma*, *P. simpsoni*, and Gobiidae (Table 4.4). *Anchoa mitchilli* and *G. strumosus* significantly decreased in mean abundances one year post-Katrina (Table 4.4). With the exception of *Palaemonetes* spp., all taxa returned to pre-Katrina abundances two years after the passage of the storm (Table 4.4). *Palaemonetes* spp. remained at significantly higher levels post-Katrina. Summer was characteristically a more stable period in the estuary as for environmental variables. Mean salinity had a steady significant increase for each year after the storm (Table 4.4). Mean water temperature significantly decreased while mean turbidity significantly increased two years post-Katrina (Table 4.4). Other environmental variables did not change significantly between pre- and post- Katrina.

DISCUSSION

Spring showed the strongest differences between pre- and post-Katrina years for the fish species composition and environmental variables. It seems these mobile species were more

Table 4.4. The means and ±2 standard error for the ten most abundant summer species and the continuous environmental variables with a, b, and c indicating significant differences ($P < 0.05$) from the post hoc Tukey’s analyses.

Taxa	Common Name	2005	2006	2007
Palaemonetes spp.	grass shrimp	0.78 ± 0.11^{a}	1.40 ± 0.16^{b}	1.15 ± 0.14^{b}
Clibanarius vittatus	thin striped hermit crab	1.25 ± 0.11^{a}	1.64 ± 0.10^{b}	1.03 ± 0.09^{a}
Callinectes sapidus	blue crab	0.59 ± 0.07^{a}	0.96 ± 0.09^{b}	0.81 ± 0.07^{ab}
Xanthids	mud crab	0.45 ± 0.09^{a}	0.80 ± 0.10^{b}	0.30 ± 0.06^{a}
Anchoa mitchilli	bay anchovy	0.45 ± 0.09^{a}	0.16 ± 0.05^{b}	0.28 ± 0.06^{ab}
Ctenogobius boleosoma	darter goby	0.29 ± 0.05^{a}	0.57 ± 0.08^{b}	0.38 ± 0.06^{ab}
C. similis	lesser blue crab	0.38±0.05	0.46±0.07	0.44±0.07
Gobiesox strumosus	skilletfish	0.45 ± 0.08^{a}	0.03 ± 0.01^{b}	0.26 ± 0.51^{a}
Panopeus simpsoni	oystershell mud crab	0.17 ± 0.05^{a}	0.37 ± 0.06^{b}	0.17 ± 0.04^{a}
Gobiidae	goby	0.01 ± 0.01^{a}	0.35 ± 0.07^{b}	0.00 ± 0.00^{a}
Environmental Variables		2005	2006	2007
Mean depth (m)		43.24±1.32	42.46±1.53	42.18±1.36
Temperature (°C)		31.59 ± 0.18^{a}	30.98 ± 0.19^{ab}	30.41 ± 0.21^{b}
Salinity (psu)		20.46 ± 0.32^{a}	22.32 ± 0.16^{b}	23.72 ± 0.38^{c}
Dissolved Oxygen (mg l^{-1})		7.21±0.25	6.09±0.14	9.89±2.83
Mean water velocity (cm s^{-1})		3.09±0.27	3.96±0.51	3.66±0.23
Turbidity (NTU)		36.07 ± 2.81^{a}	32.26 ± 2.19^{a}	50.57 ± 3.51^{b}
Distance to marsh edge (m)		3.24 ± 0.19^{a}	3.97 ± 0.19^{b}	3.24 ± 0.19^{ab}

readily re-distributed and re-assembled once environmental conditions returned to pre-Katrina conditions. *Brevoortia patronus* and *C. similis* had a large increase and *Pagurus longicarpus* first appeared post-Katrina, so these species may have been associated with the significant increase in salinity one year post-Katrina. This is a case of species responding to environmental cues that were altered by this disturbance. Also, fishing pressure was considerably less in 2006 with a loss of $1.1 billion in landing for Louisiana, and many of the abundant species are important fisheries species (*Brevoortia patronus*, *Callinectes sapidus*, *Farfantepenaeus aztecus*) (Buck 2005). *Gobiesox strumosus* was not present in either spring or summer samples one year post-Katrina (2006), but re-appeared two years post-Katrina (2007). This species is estuarine dependent and utilizes hard substrates (oyster shells) that were buried by sediment or removed and redistributed by the storm surge. *Gobiesox strumosus'* return two years post-Katrina may be from the increase, re-aggregation, or exposure of suitable habitat in this system.

The spring season is a naturally dynamic one in the estuary with passages of northern fronts and species entering into the estuary from the bays and the Gulf of Mexico. Differences caused by Katrina were more dramatic during this variable season. Though the stronger tropical storm events tend, in recent years, to occur in late summer and early fall with Ivan on 16 September 2004, Katrina on 29 August 2005, Rita on 24 September 2005 (whose effects were included in post-Katrina data), Gustav on 1 September 2008, and Ike on 13 September 2008. Summer was a more stable period during my sampling efforts. These storms were all after the last sampling trip for my summer season, so any immediate effects from Hurricanes were not tested.

Coastal plants and animals have developed with tropical storms as episodic events (Conner et al. 1989, Tilmant and Curry 1994), but individual basins can be without hurricane activity for decades. Because there is no congruent long-term data for this area, it is hard to surmise long-term effects from these current hurricane data (Switzer et al. 2006). Drastic changes such as marsh developing into open water will change nekton abundance and composition (Boesch and Turner 1984), but more subtle changes as those associated with Hurricane Katrina are harder to decipher especially across areas like our well mixed estuary. Hurricanes are short duration (pulse) perturbations (Bender et al. 1984) and their effects on natural areas are temporary with vegetation regenerating to normal levels within a year (Chabreck and Palmisano 1973, Valiela et al. 1996). This area is more open to the Gulf of Mexico than leveed areas like the Mississippi River Gulf Outlet and the Mississippi River, where Gulf water movement is facilitated to fresher regions in these channeled areas (Stokstad 2005). Hurricanes cause immediate morality and it was unfortunate we could not sample just after Hurricane Katrina but it was logistically impossible. There is also evidence of dispersal of fauna from marine to fresher areas in other wetlands (Valiela et al. 1996, Paerl et al. 2006), but in this highly saline area and the increased salinities measured after the storms this was not an issue for lower Barataria Bay. This study area and its fauna had long-term responses similar to other natural wetlands to storm events (Greenwood et al. 2006, Paperno et al. 2006, Switzer et al. 2006), but with continued anthropogenic alterations its resilience to natural perturbations may decrease.

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CHAPTER V
NEKTON ABUNDANCES IN *SPARTINA*, BLACK MANGROVE, AND TRANSITIONAL (BOTH SPECIES) MARSH-EDGE HABITAT TYPES

INTRODUCTION

Climate changes are expected to modify many aspects of Louisiana's coastal systems. Increased air and water temperatures have lead to population shifts in the poleward distribution of tropical species such as mangroves (Snedaker 1995). The decade between 1995 to 2006 ranks as the warmest on record, and this warming trend combined with human activity is decreasing wetland area around the world (IPCC 2007). Sea-level rise, a major contributor to wetland loss, increases with rising temperatures, and globally total sea-level has risen approximately 3.1 mm per year from 1993 to 2003 (IPCC 2007). Coastal Louisiana has experienced a 0.3 to 1.5 cm yr^{-1} rate of relative sea-level rise (subsidence plus sea-level rise), and a cumulative wetland loss of approximately 4,921 km^2 in the last century (USGS 2005). These numbers are particularly alarming for Louisiana where coastal marsh-edge habitat types are extremely productive and important to commercial and recreational fisheries. Coastal habitat types have been historically dominated by thick stands of *Spartina alterniflora* Loisel (smooth cordgrass, *Spartina*). However, with the current trend of increasing temperatures in southern Louisiana that has decreased the freeze events that historically resulted in black mangrove diebacks. The dominant *Spartina alterniflora* now grows in co-dominant stands with black mangrove (Scavia et al. 2002).

Spartina alterniflora of North America is a stiff perennial grass that is capable of growing in high saline marshes (Godfrey and Wooten 1979). Depending on salinity and tidal action, which can decrease species richness in marshes when salinity is increased, S*partina* is usually found in nearly monospecific stands. This saltmarsh grass is a food source for consumers found in the saltmarsh habitat such as the abundant *Palaemonetes* spp. (grass shrimp) (Mitsch and Gosselink 2000). Salt marshes are beneficial habitats because they serve as a food source and refuge for estuarine-dependent organisms (Teal 1962, Zimmerman et al. 1990), and are productive nursery habitats for resident and transient estuarine species (Minello and Webb 1997, Minello et al. 2003). Coastal salt marshes facilitate growth and recruitment of marine fauna by providing food through detritus, and refuge (Boesch and Turner 1984, Hettler 1989, Minello and Webb 1997). *Spartina* marsh-edge habitat supports high densities of nekton and this relationship has been quantitatively tested (Zimmerman and Minello 1984, Baltz et al. 1993). Due to increased vegetative habitat suitability (as a refuge) as compared to open water, many species and different life history stages utilize vegetated habitat, be it emergent marsh vegetation or seagrass, in greater densities than open water areas (Zimmerman and Minello 1984, Baltz et al. 1993). *Spartina* marshes also facilitate the accumulation of organic matter and sediment thereby increasing accretion of the marsh surface, but even this accretion is unable to keep up with current marsh loss rates (Perry 2007). *Spartina* is the dominant coastal vegetated habitat for numerous species in the southern portion of Barataria Bay.

Avicennia germinans Loisel (black mangrove, referred to as mangrove in analyses and results) is a woody species that can grow to 15 m in the tropics. It is established in the Caribbean Islands and in North America in Florida and Texas (Mitsch and Gosselink 2000). Mangroves inhabiting fringes of estuaries and tidal channels in thick stands (Mendelssohn and McKee 2000) and *Spartina*-dominated salt marshes serve the nekton community by providing food and refuge in a similar capacity (Sheriden and Hays 2003). As with all mangroves, black mangrove is sensitive to low temperatures and is not found in more temperate climates. However, black mangrove has a lower temperature tolerance and more low temperature resistant populations than other mangrove species found in the tropics (Lugo and Patterson-Zucca 1977). Black mangrove reaches its northern limit between 29° and 30° N in southeastern Louisiana’s Chandeleur Islands and marshes (Woodroffe and Grindrod 1991, Mendelssohn and McKee 2000). Large diebacks of black mangrove that occurred in the tropics in 1961-62 and 1962-63 due to freezes of -3 to -11°C suggest that mangroves do not establish where there is a 5% decrease from 20°C for the mean low air temperature or three to four nights of frost a year (Lugo and Patterson-Zucca 1977, Woodroffe and Grindrod 1991). Some question remains as to whether local nekton use of mangroves is universally high. Some qualitative studies indicate the regular utilization of mangrove habitat by juvenile decapod crustaceans and fishes, but many studies did not quantitatively compare mangrove habitat use to other fringing habitat types (Laegdsgaard and Johnson 1995, Sheriden and Hays 2003). Most mangrove comparisons have been between non-vegetated and/or seagrass habitat types, but not emergent marsh-edge grasses (Thayer et al. 1987, Chong et al. 1990, Laegdsgaard and Johnson 1995). Mangroves are severely affected by hurricanes and typically achieve larger sizes in areas of protected waters (Cronk and Fennessy 2001) such as those found in Louisiana's wetlands.

Southeastern Louisiana coastal estuaries are dominated by *Spartina*, but due to the recent decrease in freezing events, small strands of mangroves that occurred since the 1700s have become more widely established and appear to be increasing in dominance creating a marsh/mangrove ecotone (Peterson and Turner 1994, Caudill 2005). When *Spartina* and mangrove co-occur, mangroves often shade *Spartina* shoots. Both species generally thrive in areas with wave-protection, low relief topography, and fine sediments. However, mangroves can be found in sand when there are adequate amounts of organic matter in the sediment (Mendelssohn and McKee 2000). While they currently appear to be keeping pace with sea-level rise, increased sea-level rise may lead to degradation to open water, as *A. germinans* is more susceptible to this than other mangrove species (Ellison and Stoddart 1991, Field 1995). Because many of the estuarine-dependent species in southeastern Louisiana are important commercial and recreational fisheries, concern about the encroachment of mangroves and a possible habitat shift from *Spartina* marsh to mangrove stand salt marshes are high. Perry (2007) examined changes in elevation, organic matter, and carbon assimilation with changes in the physiochemical composition of the wetlands as they transform from *Spartina* to mangrove dominated. Others studies observed density and community differences of nekton among *Spartina*, black mangrove, and transition (*Spartina* and black mangrove) edge habitat type (Caudill 2005). Mangroves, due to their thicker stems, should facilitate accretion at a similar rate to *Spartina* marshes, and in Louisiana there was no difference between the two rates (Perry 2007). Regardless of the cause, Louisiana's wetlands are decreasing and open water areas are increasing.

Because of their importance, salt marsh and nekton interactions have been extensively studied (Boesch and Tuner 1984, Baltz et al. 1993, Rozas and Reed 1993, Peterson and Turner 1994, Zimmerman et al. 1990, Minello and Rozas 2002, Birdsong 2004). These studies concentrated on specific marsh edge vegetation or different bottom sediment types. There have been few studies comparing *Spartina* and mangrove edge, and the ones that exist were relatively short-term studies only lasting 10 to 13 months (Caudill 2005, Perry 2007). This study was longer term in spring and summer seasons in lower Barataria Bay, Louisiana, collected small fishes and decapod crustaceans, and used densities of selected species and total individuals along with the fish and decapod crustacean community structure to test whether patterns differed among *Spartina* dominated, black mangrove dominated, or transitional habitat type. I also compared the physiochemical properties of the water column and bottom substrate type to the potential causal relationships among vegetation and nekton species.

MATERIALS AND METHODS

Study Area

In the study area of southern Barataria Bay, *Spartina* and mangrove habitats are interspersed; therefore, I was able to sample the separate habitat types during the same sampling events (Fig 5.1). This is a microtidal estuary where silt is the dominant sediment and most wave action occurs from climatic events not tidal surges. Dominant nekton species are *Anchoa mitchilli* (bay anchovy), *Ctenogobius boleosoma* (darter goby), *Gobiesox strumosus* (skilletfish) along with some important fisheries species such as *Micropogonias undulatus* (Atlantic croaker) and *Brevoortia patronus* (gulf menhaden). Macroinvertebrates using the estuary include *Clibanarius vittatus* (hermit crab) and *Palaemonetes* spp. (grass shrimp) and the commercially important *Callinectes sapidus* (blue crab) and *Farfantepenaeus aztecus* (brown shrimp) (Chesney et al. 2000, Jones et al. 2002). I sampled near and around Mendicant and Beauregard Islands just north of the mouth of Barataria Waterway (Fig 5.1).

Field and Laboratory Methods

Because estuarine fauna are highly seasonal in abundance and community structure (Day et al. 1989, Rundle et al. 1998), I sampled seasonally in 2005, 2006, and 2007 for spring (March, April, May) and summer (June, July, August) periods. I sampled quantitatively by deploying a cylindrical 1.18 m^2 drop sampler at sites in *Spartina* (spring n = 111; summer n = 124), mangrove (spring n = 56; summer n = 42), and transitional (spring n = 56; summer n = 54) habitat types. At each site I recorded longitude and latitude with GPS, distance to marsh edge (m), and dominant substrate type and measured mean water depth (m), mean current velocity (cm s^{-1}) temperature (°C), salinity (psu), and dissolved oxygen (mg l^{-1}). I collected 50 ml of water to measure turbidity (NTU) and 50 ml sediment cores for Microtox® analysis back at the laboratory. A trash pump removed the water from the sampler and it was filtered through 333μm mesh plankton net. A sample was complete once sampler was well seated, pumped of water, and swept with nets to ensure removal of all macrofauna. I preserved, identified, measured, counted, and ran statistical analyses on all nekton.

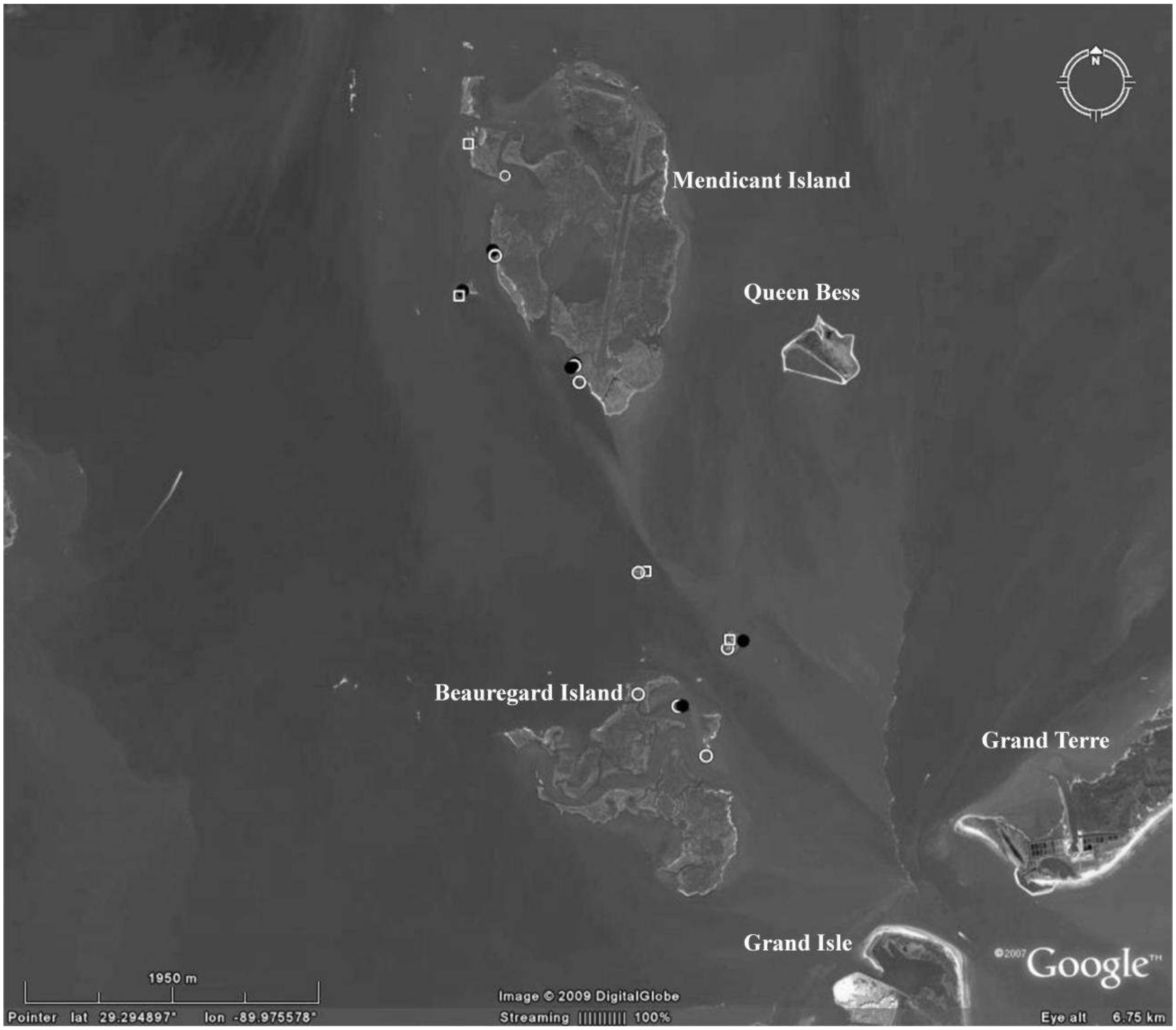


Figure 5.1. Map of study area that includes Mendicant and Beauregard Islands with examples of *Spartina* marsh sites (○), mangrove marsh sites (●), and transitional marsh sites (□).

Statistical Methods

Spring and summer seasons were analyzed separately with SAS (2004) software. I first used the MIXED procedure ANOVA to discern differences among marsh habitat types and normalized total number of individuals, total number of decapod crustaceans, and total number of fishes along with species richness, Shannon-Weiner diversity, and physiochemical variables. To ensure results were not from changes in year or month, I analyzed habitat type (*Spartina*, black mangrove, and transition), year, and months as main effects for a three-way ANOVA. Main effects differences were tested post hoc with Tukey's pairwise comparisons to detect differences among habitat types (*Spartina*, mangrove, transition). A canonical discriminant analysis was used to facilitate visual interpretation of the differences identified in the ANOVAs. Canonical discriminant analysis calculates the linear combination of quantitative variables (taxa or environmental parameters) that have the highest multiple correlations with the pre-assigned factor (*Spartina*, mangrove, and transition) to provide the maximum separation among those groups. While singular variables may not show differences among groups, linear combinations may do so (SAS 2004). It also provides MANOVAs to test if there are significant canonical correlations and separation among groups. This procedure is preferable to discriminate analysis when there are multiple correlated variables in the data set (SAS 2004).

Lastly, I used factor analyses to understand which environmental variables (distance to marsh edge, mean water depth, mean water velocity, water quality variables, turbidity, and dominant substrate) contributed to differences seen between *Spartina*, mangrove, and transitional habitat types. The analysis generated factor scores for each sample. Mean un-rotated factor scores for each habitat type and the ten most abundant species were calculated and used to plot habitat types and species in three-dimensional environmental space. Balloons were used to represent ±2 standard errors radii around group centroids.

RESULTS

Spring

I sampled 130.98 m^2 of *Spartina* and 66.08 m^2 each of mangrove and transition habitat type and collected 10,348 total individuals. The ten dominant species that accounted for 94% of the nekton community were *Palaemonetes* spp., *Brevoortia patronus*, *Clibanarius vittatus*, *C. similis*, *Callinectes sapidus*, *Micropogonias undulatus*, *Ctenogobius boleosoma*, *Farfantepenaeus aztecus*, *Gobiesox strumosus*, and *Pagurus longicarpus* (Table 5.1).

Table 5.1. Ten most abundant spring species, common names, and their total number and percent of the total abundance.

Taxa	Common Names	Total Number	Percent
Palaemonetes spp.	grass shrimp	4084	39.5%
Brevoortia patronus	gulf menhaden	1588	15.3%
Clibanarius vittatus	thin striped hermit crab	1270	12.3%
Callinectes similis	lesser blue crab	857	8.3%
C. sapidus	blue crab	612	5.9%
Micropogonias undulatus	Atlantic croaker	483	4.7%
Ctenogobius boleosoma	darter goby	300	2.9 %
Farfantepenaeus aztecus	brown shrimp	211	2.0%
Gobiesox strumosus	skilletfish	189	1.8%
Pagurus longicarpus	longwristed hermit crab	102	1.0%

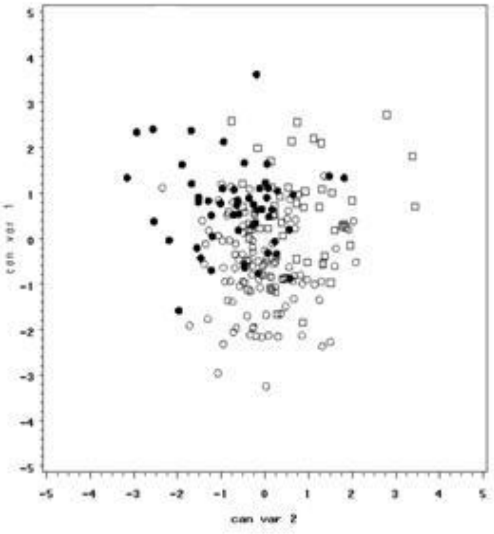
All significant results for habitat type main effects did not have significant interactions ($P > 0.05$). Total three-way ANOVAS of number of fishes, species richness, and diversity were not significant for main effects. The ANOVAs for total number of individuals and total number of decapod crustaceans were significantly different ($F_{2,\,196} = 4.45$, $P = 0.013$ and $F_{2,\,196} = 5.23$, $P = 0.006$) for mangrove, *Spartina*, and transitional habitat types. From the Tukey's post hoc analyses, the total number individuals was significantly higher in transitional ($P = 0.022$) than in *Spartina* habitat types. Also the number of decapod crustaceans was higher in mangrove ($P =$

0.021) and transitional ($P = 0.039$) than in *Spartina* habitat types. Analysis of the separate environmental variables showed a significant increase in mean water velocity for transitional ($F_{2,196} = 3.7$, $P = 0.027$) than *Spartina* marsh edge. The Canonical discriminant analyses showed significant difference between habitat classifications for total number of individuals ($F_{104, 334} = 1.32$, $P = 0.034$, Fig 5.2.a), decapods crustaceans ($F_{36,402} = 1.94$, $P = 0.001$, Fig 5.2.b), and the ten dominant species ($F_{20,418} = 2.28$, $P = 0.001$, Fig 5.2.c).

The factor analysis identified four orthogonal axes with eigenvalues greater than one that together explained 68.2% of the total variation in the system. Factor 1 explained 23.3% of variation with large positive loadings for temperature**,** salinity and mean water velocity. Factor 2 explained 18.6% of variation with positive loading for distance to marsh edge and turbidity and a negative loading for temperature. Factor 3 explained 14.6% of variation with a positive loading of mean water depth. Factor 4 explained 11.7% of variation with a positive loading of dominant substrate (Table 5.2). Habitat types fell out separately across Factor 2 with the *Spartina* samples having higher temperature, lower turbidity and sampled closer to the marsh edge, mangrove samples having lower temperature, higher turbidity, and sampled farther from the marsh edge, and the transition samples falling between them (Fig 5.3.a). *Callinectes similis* and *F. aztecus* were associated with *Spartina* habitat within environmental space. *Callinectes sapidus*, *C. vittatus*, *C. boleosoma G. strumosus M. undulatus*, and *Palaemonetes* spp. were associated with transitional habitat; while *Brevoortia patronus* and *P. longicarpus* were associated with mangrove habitat environmental parameters (Fig 5.3.b).

Summer

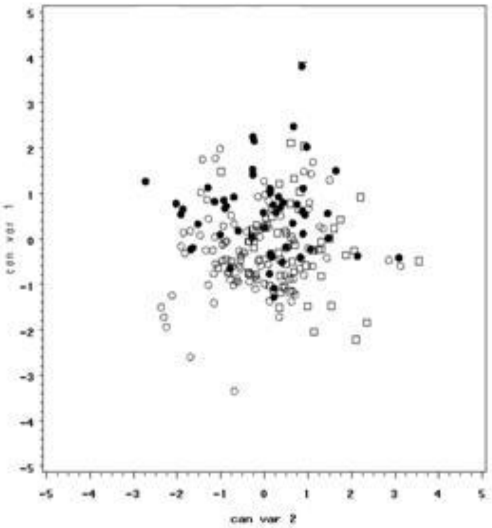
In the summer samples included 151.04 m^2 of *Spartina*, 49.56 m^2 of mangrove, and 63.72 m^2 of transitional habitat types with a total abundance of 7,480 individuals. Ten dominant taxa comprised 87% of the individuals and were *Palaemonetes* spp., *C. vittatus*, xanthids, *C. sapidus*, *G. strumosus*, *Anchoa mitchilli*, *C. boleosoma*, *C. similis*, *P. simpsoni*, and *Alpheus* sp. (Table 5.3).



a)

Figure 5.2. Graphical representation of spring canonical coefficients with a) total number of individuals, b) total number of decapod crustaceans, c) the ten abundant species; *Spartina* (○), mangrove (●), and transition (□).

Figure 5.2 (Continued)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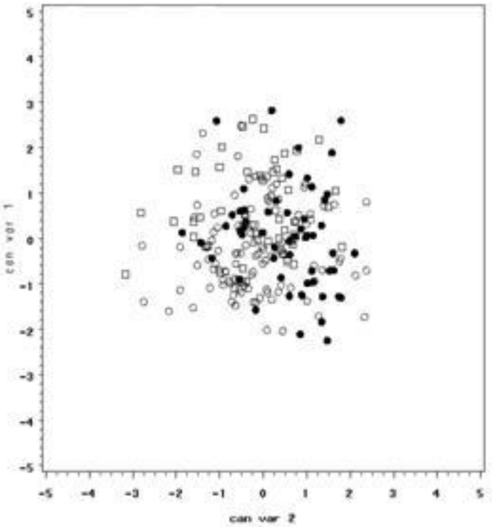
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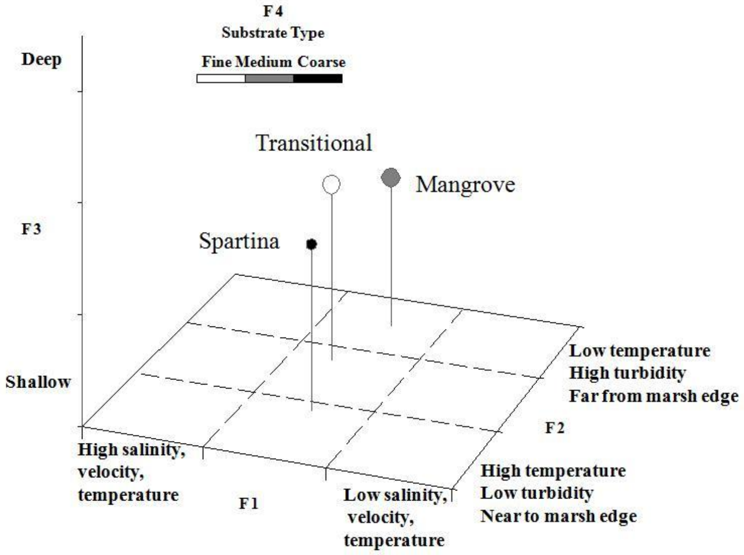
Table 5.2. Factor scores for the spring environmental variables with the highest scores in bold.

Variables	Factor 1	Factor 2	Factor 3	Factor 4
Salinity	**0.74**	0.20	0.05	0.33
Mean Water Velocity	**0.63**	0.06	-0.37	-0.05
Temperature	**0.54**	**-0.60**	0.14	0.11
Turbidity	0.23	**0.61**	0.17	-0.43
Distance to Marsh Edge	-0.44	**0.56**	0.25	0.10
Mean Depth	-0.24	-0.45	**0.71**	0.09
Dominant Substrate	-0.43	0.07	-0.46	**0.62**
Dissolved Oxygen	-0.35	-0.44	-0.46	-0.47
Eigenvalue	1.87	1.49	1.17	0.94
% Variance Explained	23.3	18.6	14.6	11.7
Cumulative % Variance Explained	23.3	41.9	56.5	68.2



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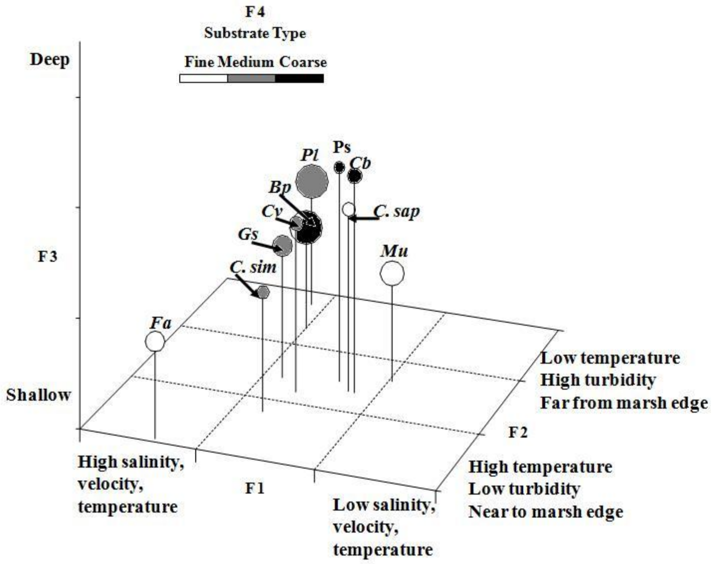


b)

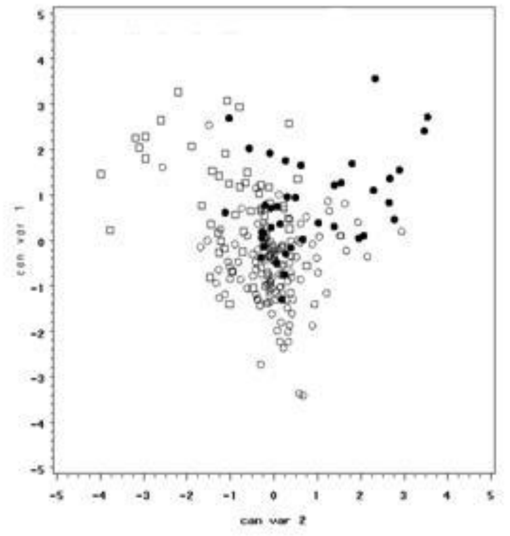
Figure 5.3. Plot of spring a) habitat types and b) ten most abundant species with Ps: *Palaemonetes* spp., *Bp*: *B. patronus*, *Cv*: *C. vittatus*, *C. sim*: *C. similis*, *C. sap*: *C. sapidus*, *Mu*: *M. undulatus*, *Cb*: *C. boleosoma*, *Fa*: *F. aztecus*, *Gs*: *G. strumosus*, *Pl*: *P. longicarpus* in three-dimensional environmental space. Habitat centroids are the factor means with balloon radii representing ± 2 standard error.

Table 5.3. Ten most abundant summer taxa, common names, and their total number and percent of the total abundance.

Taxa	Common Name	Total Number	Percent
Palaemonetes spp.	grass shrimp	2747	36.7%
Clibanarius vittatus	thin striped hermit crab	1456	19.5%
xanthid crab	mud crab	610	8.2%
Callinectes sapidus	blue crab	423	5.7%
Gobiesox strumosus	skilletfish	298	4.0%
Anchoa mitchilli	bay anchovy	274	3.7%
Ctenogobius boleosoma	darter goby	273	3.7%
C. similis	lesser blue crab	190	2.5%
Panopeus simpsoni	oystershell mud crab	153	2%
Alpheus sp.	snapping shrimp	96	1.3%

Variables with significant ANOVA results did not have significant interactions ($P > 0.05$). The ANOVAs for total number of individuals ($F_{2,193} = 4.88$, $P = 0.009$), total number of decapod crustaceans ($F_{2,193} = 3.41$, $P = 0.035$), total number of fishes ($F_{2,193} = 4.14$, $P = 0.017$), species richness ($F_{2,193} = 4.95$, $P = 0.008$), diversity ($F_{2,193} = 4.30$, $P = 0.015$) and were significantly different across all habitat types. Means of all biological variables were significantly higher in mangrove ($P < 0.05$) than in *Spartina* habitat types. The ANOVA conducted on the environmental variables for summer showed no significant difference among habitat types. The canonical discriminant analysis showed a significant differences among habitat classifications for total number of individuals ($F_{112,324} = 1.39$, $P = 0.015$, Fig 5.4.a), total number of fishes ($F_{76,360} = 1.32$, $P = 0.050$, Fig 5.4.b), and environmental variables ($F_{14,422} = 1.98$, $P = 0.018$, Fig. 5.4.c).

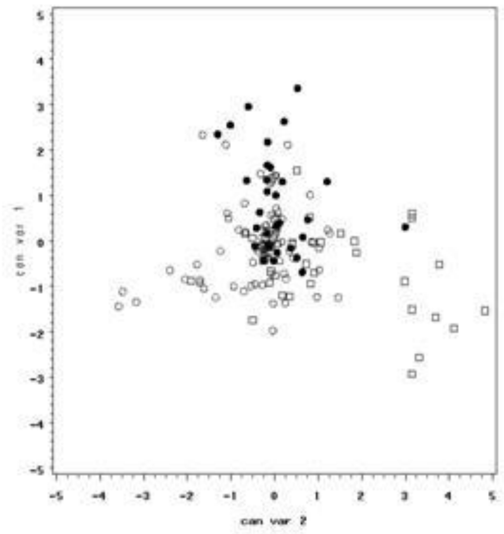
Factor analysis depicted four orthogonal axes with eigenvalues greater than or equal to one and explained 62.5% of the variation. Factor 1 explained 17.4% of variation with large negative loadings for DO and temperature. Factor 2 explained 16.8% of variation with a positive loading for mean water depth and a negative loading for turbidity. Factor 3 explained 15.2% of variation with a positive loading of dominant substrate and a negative loading for mean water velocity. Factor 4 explained 13.1% of variation with a positive loading for distance to marsh edge (Table 5.4). Habitat types separated the most across the Factors 1 and 3 where *Spartina* samples had higher temperature, DO, mean water velocity, finer sediments, and we were able to sample closer to the marsh edge. Mangrove samples had lower temperatures, DO, mean water velocity, turbidity, higher mean water depth, coarser dominant substrate, and we sampled farther to the marsh edge. The transition samples were intermediate (Fig 5.5.a). *Alpheus* sp., *A. mitchilli*, and *P. simpsoni* were more closely associated with *Spartina* habitat in environmental space. *Callinectes similis*, *G. strumosus*, *Palaemonetes* spp., and xanthid crabs were associated with *Spartina* and transition habitats, while *C. sapidus*, *C. vittatus*, and *C. boleosoma* associated with transitional and mangrove habitat types (Fig 5.5.b).



a)

Figure 5.4. Graphical representation of summer canonical coefficients for a) total number of individuals, b) total number of fishes, c) environmental variables; *Spartina* (○),mangrove (●), and transition (□).

Figure 5.4 (Continued)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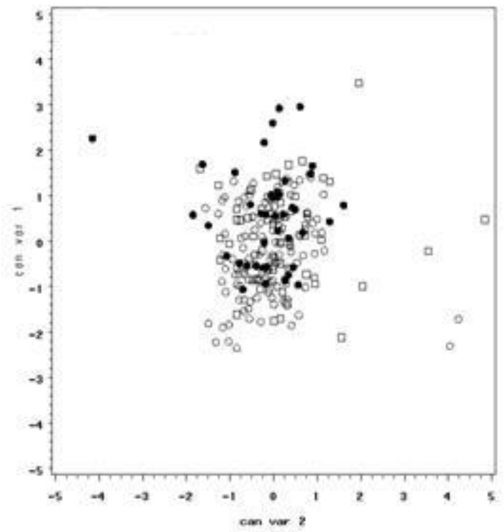
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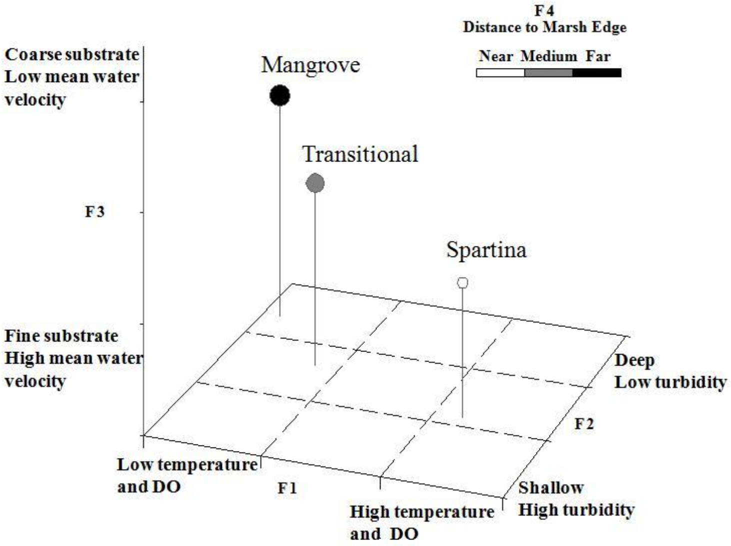
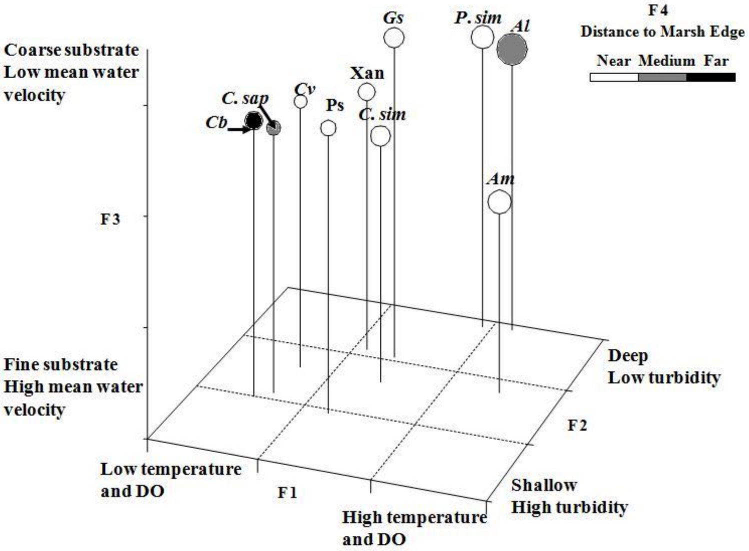
Table 5.4. Factor scores for the summer environmental variables with the highest scores in bold.

Variables	Factor 1	Factor 2	Factor 3	Factor 4
Temperature	**-0.62**	0.22	-0.25	-0.37
Dissolved Oxygen	**-0.52**	0.38	0.07	-0.11
Mean Depth	0.41	**0.63**	-0.20	0.32
Turbidity	0.23	**-0.62**	0.09	-0.32
Dominant Substrate	0.07	0.22	**0.81**	0.20
Mean Water Velocity	0.35	0.13	**-0.59**	0.22
Distance to Marsh Edge	-0.34	-0.27	0.13	**0.69**
Salinity	0.38	0.48	0.30	-0.36
Eigenvalue	1.39	1.35	1.22	1.05
% Variance Explained	17.4	16.8	15.2	13.1
Cumulative % Variance Explained	17.4	34.2	49.4	62.5



a)



b)

Figure 5.5. Plot of summer a) habitat types and b) ten most abundant species with Ps: *Palaemonetes* spp., *Cv*: *C. vittatus*, Xan: xanthid, *C.sap*: *C. sapidus*, *Gs*: *G. strumosus*, *Am*: *A. mitchilli*, *Cb*: *C. boleosoma*, *C. sim*: *C. similis*, *P. sim*: *P. simpsoni*, *Al*: *Alpheus* sp. in three-dimensional environmental space. Habitat centroids are the factor means with balloon radii representing ± 2 standard error.

DISCUSSION

Current and hypothesized climate changes that increase temperatures and decrease the frequency of freeze events will expand the dominance of mangrove stands in coastal southeastern Louisiana. While mangroves are a form of estuarine habitat, they were not historically prevalent in North America, nor have they previously expanded at their current rate. Louisiana *Spartina* salt marshes are decreasing with the increased sea-level rise, major storm events, and continued anthropogenic perturbations. This increases unvegetated marsh areas for the seeds of the black mangrove to germinate and establish thick mangrove stands. While other studies have found little change with surface soils and vegetated impacts (Perry 2007), this chapter looked at effects on nekton community components that utilize the *Spartina* marsh edge as refuge and habitat.

Spring data showed the total number of individuals, driven by the abundant decapod crustacean community, was higher in mangrove and transitional habitat types than *Spartina* and this was similar to Caudill (2005) findings. There were also trends of *Spartina* habitats having lower turbidity than mangrove habitats. *Spartina* in dense stands has more numerous stems that grow farther from the emergent vegetated edge than mangrove pnuematophores. These *Spartina* stems can slow down water velocity and trap suspended sediments from the water column, similar to sea grasses. Mangroves are not in dense stands in Louisiana. The transient and economically important species that utilize estuaries as nurseries is *F. aztecus* (Beck et al. 2001) and it was strongly associated with *Spartina* habitat, in this study. Thus a shift in dominance to mangroves may result in declines for this species in this heavily fished area, and the enhancement of other species more closely associated with mangroves (Fig 5.3).

In the summer all biological variables were significantly higher in black mangrove than *Spartina* dominated marsh-edge habitat type. There was a trend of *Spartina* habitats occurring in higher temperature and turbidity than mangrove, but this is because of the lower mean depth and higher mean water velocity along *Spartina* edge. The more shallow water was warmer and the turbulent flowing water re-suspended the fine sediments. *Spartina* also was associated with finer sediments like silt and clay which made this habitat more suited for the resident *A. Mitchilli*, and mangrove habitat was associated with the transient *C. sapidus*. Mangroves are complex structures on the marsh edge and although the stems and pnuematophores are coarser than *Spartina* stems they still act as complex structure for aquatic fauna providing refuge and food from the epiphytic algae (Laegdsgaard and Johnson 2001, Caudill 2005). Mangroves occur at higher elevations than *Spartina* in southeastern Louisiana and these were usually on the steeper banks of channels and creeks (Perry 2007). Many of the species in the summer were associated with the transitional habitat types.

Nekton utilized marsh edge along all habitats in Barataria Bay as these three vegetated habitat types were in close proximity to each other. Mangrove habitats attract juvenile fishes and decapod crustaceans like other vegetated habitat in estuaries (Sheridan 1997). While mangrove infringement may be a new phenomenon in southeastern Louisiana, even this vegetative marsh edge is more beneficial than unvegetated open water habitat (Zimmerman and Minello 1984). There still needs to be research on the four components of nursery habitat (Beck et al. 2001) in each of the open water, mangrove, *Spartina*, and transition habitat types to better understand how our fisheries species are adapting or utilizing these areas as nurseries in Barataria Bay (Sheriden and Hays 2003). Both seasons had high decapod crustacean numbers in mangrove as they seem to exploit that marsh edge (Caudill 2005). However, this study showed two fisheries species' response to shifts in dominant habitat type and there is a need for more detailed examination of this relationship, and the use of black mangroves as suitable nursery habitat in southeastern Louisiana.

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CHAPTER VI
ANALYSIS OF A LOUISIANA SALTMARSH-EDGE COMMUNITY FOODWEB: SHIFTS IN FAUNAL INTERACTIONS FROM PRESS PERTURBATIONS

INTRODUCTION

Louisiana coastal marshes accommodate a range of taxa in multiple trophic levels such as wading birds (Ciconiiforms), bottlenose dolphin (*Tursiops truncatus*), red drum (*Sciaenops ocellatus*), gulf menhaden (*Brevoortia patronus*), bay anchovy (*Anchoa mitchilli*), southern flounder (*Paralichthys lethostigma*), blue crab (*Callinectes sapidus*), oystershell mud crab (*Panopeus simpsoni*), brown shrimp (*Farfantepenaeus aztecus*), American oyster (*Crassostrea virginica*), and early life history stages of numerous other fishes and macroinvertebrates. Red drum, gulf menhaden, blue crab, and brown shrimp are commercially important species and use coastal marshes for larval and juvenile life history stages (Nelson 1992). The marsh edge is dominated by *Spartina alterniflora* (smooth cordgrass) and is nursery habitats for many estuarine-dependent species (Boesch and Turner 1984). In southeastern Louisiana, the lower Barataria Bay estuary is directly connected to the Gulf of Mexico, and is in a constant state of change from multiple long-term anthropogenic perturbations (press perturbations sensu Bender et al. 1984). Examples of these disturbances are oil industry presence (Roth and Baltz 2009), wetland loss (Evers et al. 1992), and active and planned freshwater diversions to combat wetland loss (Lane and Day1999), mangrove encroachment from a decreased frequency of freeze events due to climate change (Mendelssohn and McKee 2000), and a long history of shrimping and fishing activities (Shervette et al. 2004). These influences are pressures for change in relationships between and among abiotic and biotic ecosystem variables and can result in shifts within the food web. To understand the effects of these presses on the trophic web of Barataria Bay and to pinpoint areas of needed research, I utilized qualitative loop modeling. This approach allowed me to examine the upper trophic levels of the marsh-edge habitat of the bay and look at interactions among floral, faunal, and specific abiotic environmental variables. I also used the models to predict the directions of change in community food-web constituents from increased biomass of black mangrove, freshwater input by diversions, increased shrimping pressure, and wetland loss.

MATERIALS AND METHODS

Study Area

Barataria Bay in southeastern Louisiana is shallow, and in the spring and summer it has high salinity, dissolved oxygen, and turbidity. The substrate is dominated by fine sediments, including silt and clay with high organic matter. The lower Bay (Fig 6.1) includes the same species associated with most Louisiana coastal wetlands and estuaries, and is strongly influenced by petroleum and fishing industries. I used natural history literature sources, my own observations, and faunal databases to create several loop models characteristic of non-degraded saltmarsh edge communities for this study.

Loop Model

For more detailed methods and formulas of my general loop model refer to Dambacher (2001). Ecological loop models are qualitative signed diagraphs that graphically represent direct interactions between major biological and abiotic variables in a system. These direct interactions define a community's structure based on connections between major components and the transmission of effects directly and indirectly from one component to another through changes of interaction signs between variables within the model. The signs of direct interactions are positive (represented by an arrow) if x_1 increases from the influence of x_2 and negative (represented by a circle) if x_1 decreases from the influence of x_2 (Fig 6.2. a,b,c) (Lane and Levins 1977). There are also self-dampening negative feedback loops that account for natural interactions outside the scope of a model (Fig 6.2.a) (Lane and Levins 1977). A feedback loop is a directional pathway

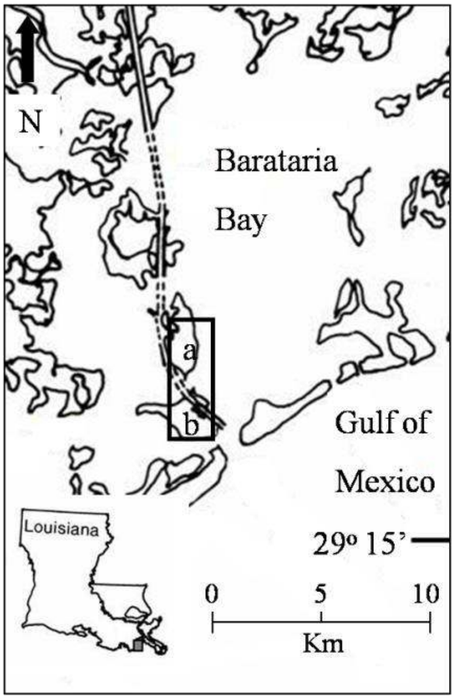

Figure 6.1. Map of the study area in Barataria Bay in Southeastern Louisiana where data were collected and observations made a) Mendicant Island and b) Beauregard Island.

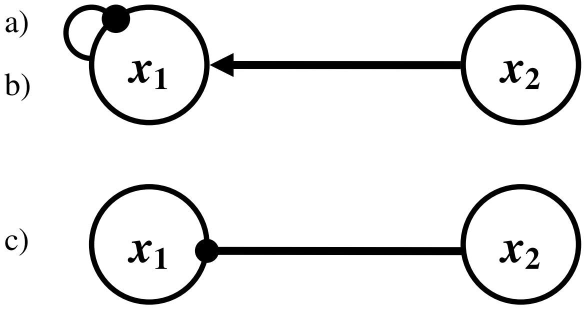

Figure 6.2. Loop model interactions a) negative self damping loop, b) positive interaction of x_2 on x_1, and c) negative interaction of x_2 on x_1

running from one node through others and back to the node of origin without repeating any portion of that pathway. Feedback loops are positive if the sign is unchanged and negative if the sign changes (Dambacher 2001). Direct interactions including predator-prey (+/-), interference competition (-/-), mutualism (+/+), commensalism (+/0), and amensalism (-/0) relationships can all be represented in the diagraph. Once the diagraph is completed, a community interaction matrix of 1, -1, and 0's (where 0 indicates no direct interaction) is created based on direct interactions between variables. This matrix shows the direct effects of all variables on each other in the model (Dambacher et al. 2003). From the community interaction matrix, the adjoint matrix is calculated. The adjoint matrix represents the net number of positive or negative feedback loops acting on any variable in the model, where any change in a column variable exerts a positive or negative influence on each row variable in the column. There are several diagnostic tools available to assess the stability of a given model. Feedback at each level is used to predict the overall stability of the model. System stability depends on three criteria: 1) overall feedback must be negative, 2) the net feedback at all levels must be negative, and 3) the net negative feedback at lower levels must be stronger than feedback at higher levels (Dambacher et al. 2003). The adjoint matrix is used to predict net changes to constituents in the system when one or more other variables or nodes are altered (Dambacher and Ramon-Jiliberto 2007). To assess perturbations to the system, I employed presses on the adjoint matrix to simulate disturbances to system equilibrium.

Presses are persistent disturbances in near-equilibrium systems to one or more variables and can affect other variables through direct or indirect interactions (Bender et al. 1984). Presses can be a positive or negative and are evaluated via the adjoint matrix as the effects of positive or negative changes to one or more column variables on all row variables. Negative presses are generated by reversing the sign at all nodes in a column. For example, a positive press on nutrients and a negative press on salinity could simulate a freshwater diversion to a system, and if the fresh water is turbid it would be included in the press. The combined effects of two or more presses of column variables on row variables are expressed through the algebraic summation of row variables of the corresponding columns in the adjoint matrix (e.g. ∑ of + nutrients, - salinity, and + turbidity).

Individual loop models cannot be manipulated in the manner that quantitative models can by changing interaction rates between nodes, but rather direct interactions between nodes must be made or broken to examine similar models. Thus several similar alternative models can be compared by modifying the direct links between nodes or variables to evaluate their affects on the outcome of particular presses. These manipulations allow one to test whether the basic model is the best representation of the study system and whether the conclusions are general across an array of similar models (Lane and Levins 1977). Four presses were assessed with each of these models in an effort to uncover a general and realistic model for Barataria Bay marsh-edge food webs. All models were stable systems that did not collapse with changes to the interactions or links.

Initial Model Variables and Interactions

My initial model (Fig 6.3) is a representation of the primary food sources and macrofauna on the marsh edge. Top level carnivores are represented by piscivorous wading birds (WB) such as Herons, Egrets, and Ibises that prey upon nekton at the shallow marsh edge and their prey consist of anchovies, small sciaenids, tonguefishes, eels, shrimps, and crabs (Wambach and Emslie 2003). High nekton density and vulnerability are necessary for successful predation by wading birds (Gawlik 2002). They leave an area if prey densities, or interactions with the environment change prey availability (Gawlik 2002). Water column predatory fauna (P) such as sciaenids and larger blue and stone crabs (Fig 6.3) use the marsh edge as a nursery ground until they are well into the juvenile life history stage, then they move into the open bay or up the estuary (Currin et al. 1984). These species are opportunistic feeders and prey upon the smaller life history stages of many of the fishes and macroinvertebrates in the area. Larger sciaenids such as drum and Atlantic croaker feed on juvenile transient fishes, resident fish like bay anchovy, blue and mud crabs, and shrimp (Stickney et al. 1975, Scharf and Schlight 2000, Simonsen 2007). Larger blue and stone crabs prey on most of the large macrofauna in the estuary including clams, fishes, and other crabs (Gibson 1988, Hines et al. 1990).

Brown and white penaeid shrimp (PS) in southeastern Louisiana's estuaries support important fisheries (Fig 6.3) (Chesney et al. 2000, Shervette et al. 2004). Juvenile shrimp utilize the marsh edge as nursery grounds (Minello and Zimmerman 1985). The *Spartina* vegetated edge has been shown to decrease predation of brown shrimp by species like Atlantic croaker (Minello et al. 1987). Penaeid shrimp consume detritus and small caridean shrimp. The node for benthic fishes (BF) represents a guild of small and juvenile stages of many flatfishes, gobies, and eels that live on or in the sediment (Fig 6.3). Larger flatfishes eat young-of-year sciaenids and other benthic fauna (Rice et al. 1993). Gobies and small flatfishes prey on zooplankton and meiofauna that live on stems of *Spartina* (Carle and Hastings 1982, Fitzhugh and Fleeger 1985, Switzer 2003). The remaining faunal nodes are macroinvertebrates.

Numerous small crabs (SC), represented by lesser blue crab, mud crabs, and earlier life history stages of the large crabs (Fig 6.3), are smaller animals, and molt frequently making them vulnerable to many predators. These crabs eat detritus and benthic algae (Currin et al. 1995). Mud crabs are can be prevented from accessing essential resources (food and refuge) by the large stone crabs in the estuary (Brown et al. 2005). Grass shrimp (GS) are small but highly abundant animals at the marsh edge (Fig 6.3), and interact with vegetation as they eat meiofauna associated with *Spartina* stems and consume the epiphytic and benthic algae in the system (Gregg and Fleeger 1998). They also feed on live *Spartina* stems and detritus (Welsh 1975).

Two additional nodes represent environmental variables in the estuary. Turbidity (NTU) (Fig 6.3) is a measure of the amount of suspended particles in the water column. Turbidity affects predation by epibenthic and visual predators (like many in this loop model), because it modifies visibility and their effectiveness to capture prey (Cyrus and Blaber 1992, Chesney et al. 2000). Higher abundances of marsh-edge fishes are associated with turbidity above 10 NTU and water depth great than 30 cm, where the bottom is not visible and prey are less visible to their predators (Baltz et al. 1993). The study area had a mean turbidity of 41 NTU and the bottom was rarely visible except at the water's edge. The dominant fine-grained sediment (FS) found in much of coastal Louisiana (Fig 6.3) is from 0.06 mm to 0.004 mm on the Udden-Wentworth Grade scale, and of the type historically deposited by the Mississippi River. This sediment is the last to fall out of suspension, it is good burrowing substrate and can easily be resuspended by burrowers, storm events, or trawling.

The last three nodes are primary producers found on the marsh edge. *Spartina alterniflora* (*Sa*) (smooth cordgrass) is the dominant saltmarsh vegetation in Louisiana (Fig 6.3 and 6.4.a,c). It is an important form of structure that is utilized as refuge by small estuarine nekton, and it is the primary producer of detritus (Welsh 1975, Currin et al. 1995, Gregg and Fleeger 1998). *Spartina* is a major primary product in the marsh and is associated with higher densities of nekton than open water habitat (Minello and Zimmerman 1985, Baltz et al. 1993, Mitsch and Gosselink 2000). Epiphytic and benthic algae (A) found in the Bay (Fig 6.3) grow on either plants (epiphytic) or substrate (benthic) and are another primary food source for many species in the estuary (Sullivan and Moncreiff 1990, Currin et al. 1995). Because of the high turbidity in lower Barataria Bay, light is often unable to penetrate to the sediment and affects the amount of algae grown on substrates. *Avicennia germinans* (black mangrove) (*Ag*) is the dominant mangrove species in higher latitudes and lower temperatures (Fig 6.3 and 6.4.b,c) (Woodroffe 1982). It lives in monospecific and mixed stands and performs the same functions as other wetland plants (Mendelssohn and McKee 2000, Sheriden and Hays 2003). Due to a decreasing frequency and intensity of freeze events, black mangrove has been expanding in southeastern Louisiana marshes creating a *Spartina*-mangrove ecotone at many points on the marsh edge (Fig 6.4.c) (Peterson and Turner 1994, Caudill 2005), and becoming the dominant vegetative species in some marshes.

New Model Variants

The initial model had predator-prey interactions for most trophic levels (Fig 6.3). There were two direct one-way negative interactions of turbidity (NTU) on wading birds (WB) and on algae (A), because in turbidity reduces the bird's ability to detect prey and algal photosynthesis is reduced. Most of the other environmental variables had direct positive interactions on the faunal nodes. To explore the generality of the models, I examined two basic variations on the initial model (Levins 1966) the first by reversing s direct link, and the second by adding a direct two-way faunal interaction.

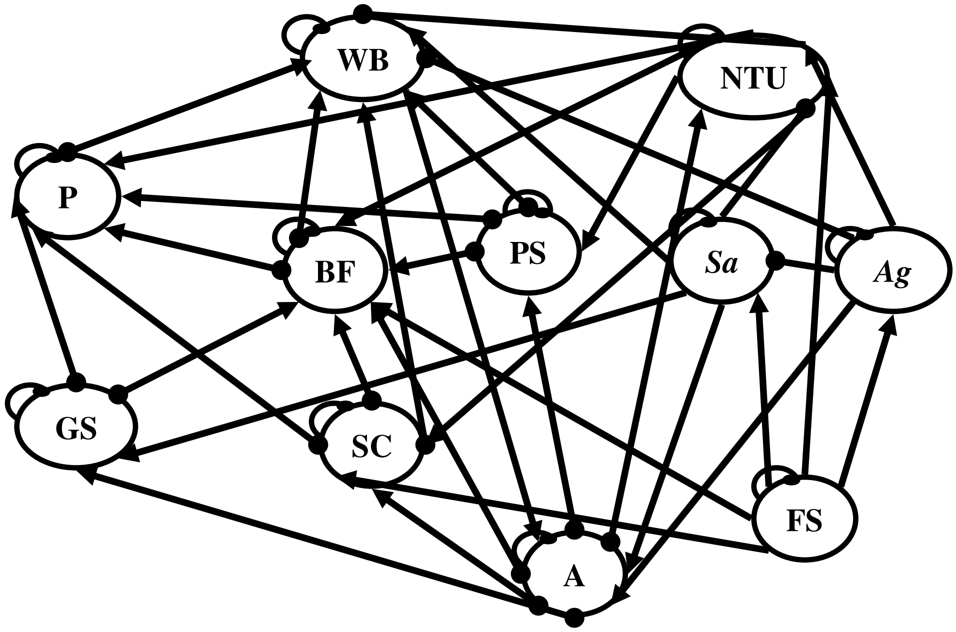

Figure 6.3. Initial loop model of direct interactions between primary producers and consumers of higher trophic levels and environmental variables at the Barataria Bay marsh edge. Individual nodes are described in the text in more detail: A = algae, FS = fine sediment, GS = grass shrimp, SC = small crabs, P = water column predators, BF = benthic fishes, PS = penaeid shrimp, *Sa* = *Spartina*, *Ag* = black mangrove, WB = wading birds, and NTU = turbidity.

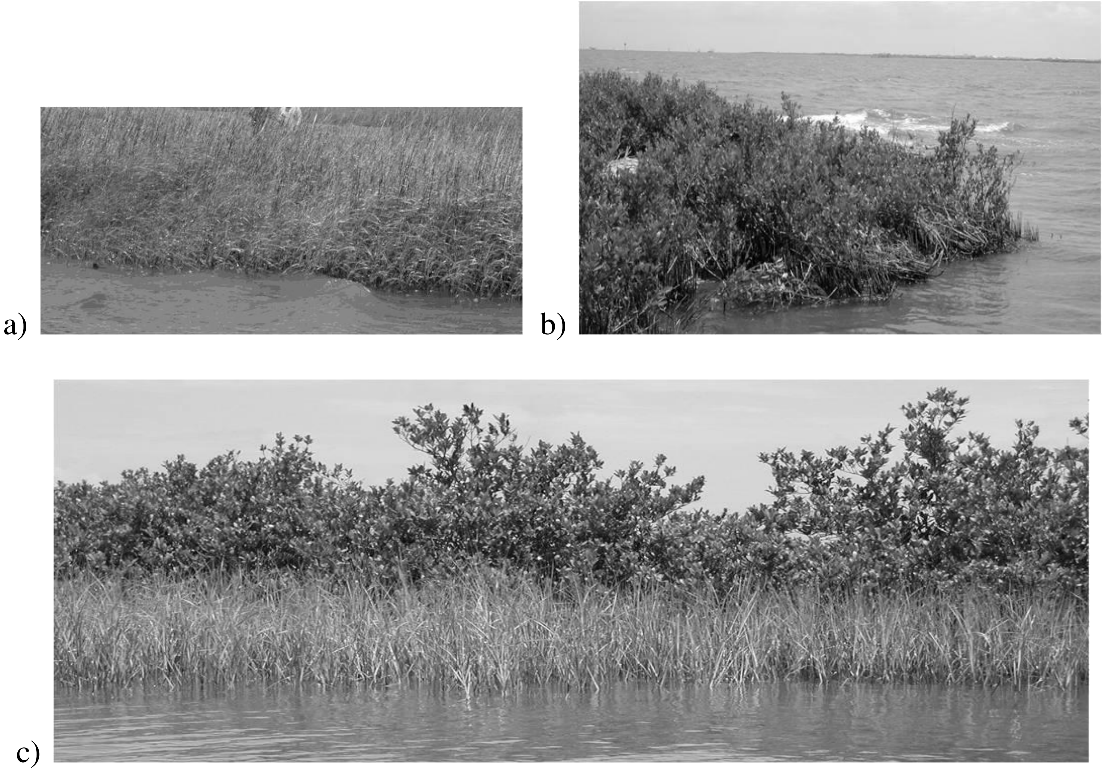

Figure 6.4. Photos of a) *Spartina alterniflora* marsh edge, b) *Avicennia germinans* marsh edge, and c) the ecotone in the study area.

Several benthic fishes (BF), like gobies and eels, can evade predators by burrowing into the sediment, while other species like blackcheek tonguefish have dark or cryptic coloration to blend into the background. For these fishes, turbidity is not an important refuge characteristic but may hinder their ability to locate and capture prey (Walsh et al. 1999). This relationship was represented by a change to the initial model from that reversed positive direct interaction between benthic fishes and turbidity to a negative in the second model (Fig 6.5). The third model (Fig 6.6) differed from the initial model, by the addition predator-prey interaction between penaeid shrimp and grass shrimp. Penaeid shrimp (PS) will prey upon the early life history stages of grass shrimp (GS) (Minello and Zimmerman 1983). There were no environmental alterations in this model.

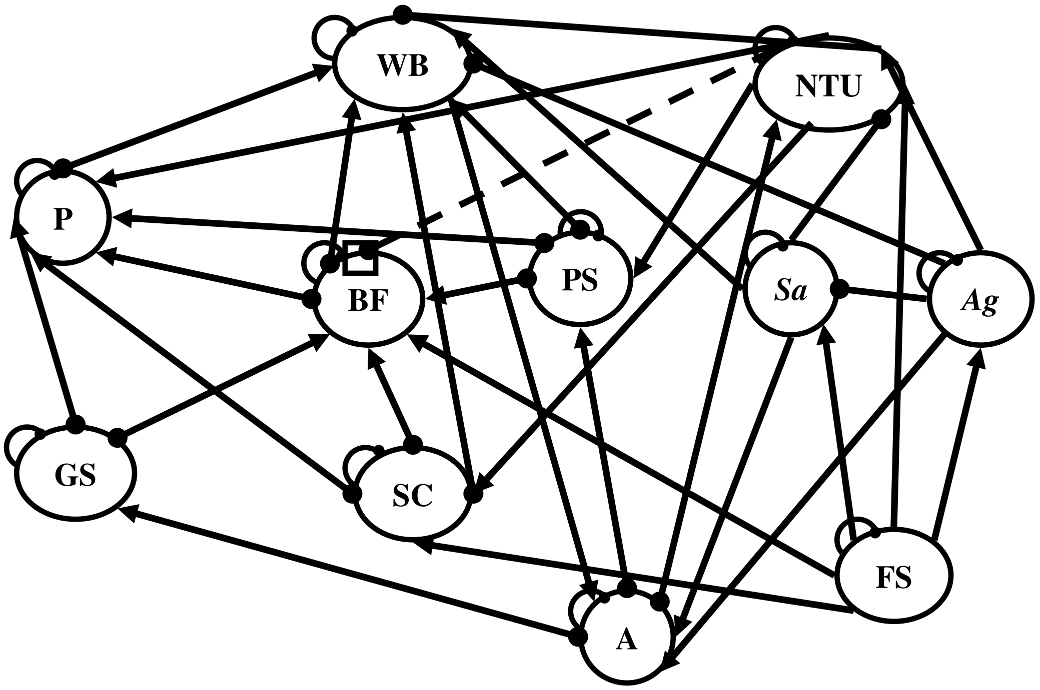

Figure 6.5. Modified loop model of direct interactions between primary producers and consumers of higher trophic levels and environmental variables at the Barataria Bay marsh edge. Individual nodes are described in the text. The changed link is represented by a dashed (---) line and the changed interaction sign is boxed. Justification for the change is in the text.

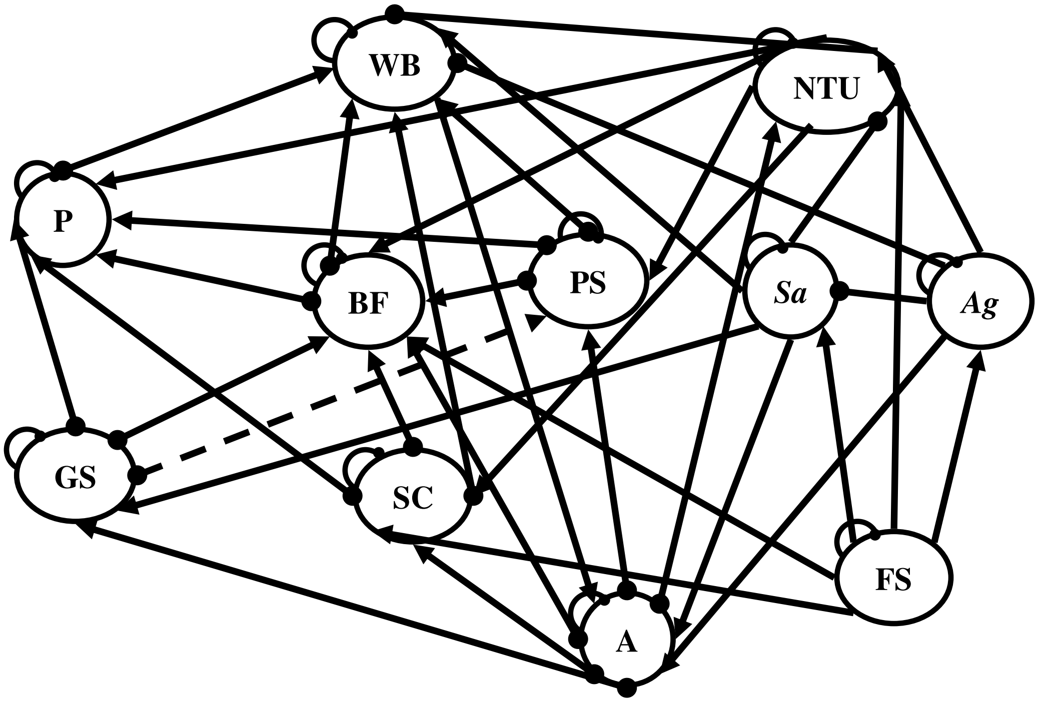

Figure 6.6. Modified loop model of direct interactions between primary producers and consumers of higher trophic levels and environmental variables at the Barataria Bay marsh edge. Individual nodes are described in the text. The changed link is represented by a dashed (---) line. Justification for the change is in the text.

Presses

After a community interaction matrix is developed based on direct interactions and determined to be stable, effects from sustained alterations (presses) to one or more variable(s) on the other nodes in the model can be examined through the adjoint matrix. Presses can be from a single node on the rest of the variables like the black mangrove influences on *Spartina* and nekton, or a combined press of multiple nodes to simulate an events' influence on the model. I explored a marsh-edge vegetation shift and three anthropogenic actions with four different combination presses to determine how the marsh-edge community may respond. These were a shift from *Spartina* to black mangrove dominated marsh edge, freshwater diversion, shrimping pressure, and wetland loss. Each of these presses was used to predict changes in the food web interactions of the Barataria Bay marsh-edge community with different anthropogenic disturbances.

First, an influence of the warming trend associated with climate change is the decrease of freeze events and consequential increase in abundance of black mangrove (Caudill 2005 and Perry 2007). The increased black mangrove biomass decreases biomass of the dominant *Spartina* and will affect the marsh-edge faunal community (Chapter 5). I simulated this marsh vegetation alteration by summing a negative press on the *Spartina* (*Sa*) and a positive press on the black mangrove (Ag). I then compared the response of the other variables to this combination of presses.

Secondly, several freshwater diversion projects have been developed in southern Louisiana in an effort to input fresh water, suspended sediments and nutrients into the coastal system to offset subsidence and sea level rise to save wetlands (Scavia et al. 2002). These diversions are intended to recreate the flooding events from the Mississippi River that historically deposited nutrient rich sediments and helped with accretion to keep the marsh accretion rates ahead of sea level rise (Lane and Day 1999). Most of the nekton in lower Barataria Bay are marine species, and there is concern that a shift in the water regime will affect food-web interactions (Drinkwater and Frank 1994). I pressed environmental and habitat variables to see how faunal populations would change. Positive presses on turbidity (NTU), fine sediment (FS), *Spartina* (*Sa*), algae (A), and black mangrove (*Ag*) simulated a freshwater diversion on the lower Barataria Bay saltmarsh edge habitat.

Thirdly, shrimping has been an important industry in Louisiana for decades (Chesney et al. 2000). In 2006 the industry in Louisiana landed 61.19 metric tons of shrimp valued over $110 million (Isaacs and Lavergne 2007). Shrimp trawling efforts have been a long-term activity on the coasts, and shrimping has caused changes to the sea floor such as the reduction of large bivalves and other macrobenthic organisms (Dayton et al. 1995, Chesney et al. 2000). Even with the increased regulations and use of bycatch reduction devices (BRDs), there is still a large biomass of non-target bycatch associated with shrimp trawling (Rogers et al. 1997, Chesney et al. 2000). The bycatch is usually larger fishes and blue crabs that are occupying the same parts of the bay as penaeid shrimp (Shervette et al. 2004). The influences of shrimping and bycatch were modeled by negative presses on penaeid shrimp (PS) and water column predators (P). The shrimp fisheries use bottom trawls to collect shrimp and other demersal organisms. These trawls make contact with bay substrates resuspending fine sediments and significantly increasing turbidity, so a positive press on turbidity was included in the assessment of shrimping.

Finally, from 1945-1985, 60% of Louisiana's wetlands were converted to open water (Evers et al. 1992) by various forms of land loss. Over a 20 year period (1995-2015) Barataria Bay is projected to lose 28% of its marsh (LaCoast 2008). Figure 6.7 shows the shift of three small islands from marsh to open water in seven years. Wetland loss is a major concern for coastal Louisiana as we have 41% of the continental wetlands in United States (Turner and

Gosselink 1975). Climate change increases sea level rise by expanding oceans and melting ice caps (IPCC 2001). Wetlands cannot keep up or adapt to increases in sea level due to anthropogenic influences (Scavia et al. 2002). To understand the implications of marsh conversion to open water for the marsh-edge food web, I modeled a positive press on fine sediment and turbidity as they would increase with less vegetation to secure sediment or slow the water to allow sediment to settle. I also applied negative presses on the marsh-edge vegetation, *Spartina* and black mangrove.

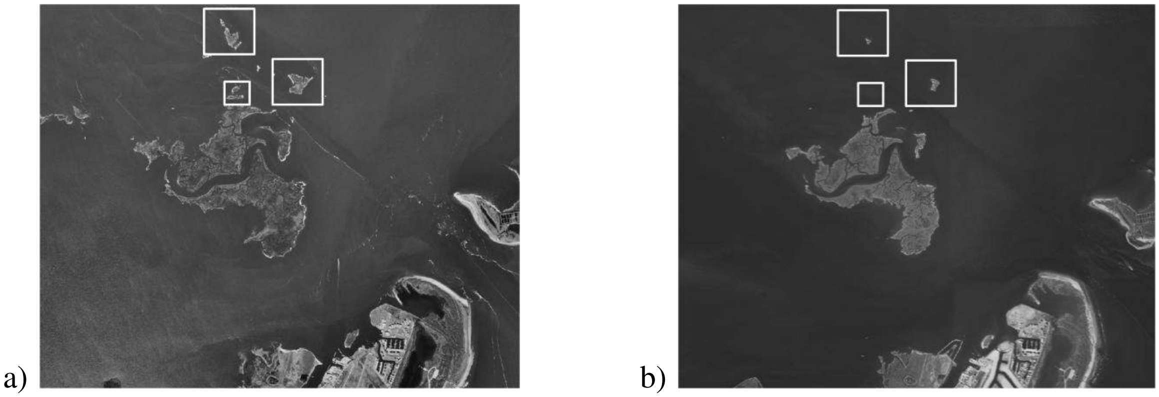

Figure 6.7. Map of Beauregard Island a) 1998 and b) 2005. The three islands in boxes show the increased wetland loss over the years.

RESULTS

I used the initial models and its two variations, to examine four basic presses simulating long-term perturbations on the marsh-edge community. All models met system stability criteria with overall feedback being negative for all levels, and the negative feedback was strongest at lower levels. The most notable response from presses on individual variables in the initial model were from a press on fine sediments (FS) and black mangrove (*Ag*) (Table 6.1). A press on fine sediment increased water column predators (P) and turbidity (NTU), but decreased grass shrimp (GS). A press on black mangroves resulted in strong negative response by grass shrimp with a positive response from turbidity. A press on *Spartina* (*Sa*) had a strong positive response from grass shrimp. Penaeid shrimp (PS) and small crabs showed no response from the other nekton variables. In the second model, the direct positive interaction of turbidity on benthic fishes was reversed to negative. Fine sediments and black mangrove resulted in the most notable responses when they were individually pressed (Table 6.2). Benthic fishes (BF) responded negatively to the positive press on fine sediment. Faunal variables had more effects on the environmental variables in this model than the initial model. In the third model, the revision was adding a predator-prey link between grass shrimp and penaeid shrimp, and turbidity and algae responded to every other variable in this model. All responses in this model were generally weaker than those in both other models. All three models indicated the same directional changes for presses on individual variables with primary exceptions being the negative response of benthic fishes in the second model and the responses of turbidity and algae in the third model.

Using each model's adjoint matrix, the community responses to four simulated events (i.e., combinations presses) were compared. A negative press on *Spartina* combined with a positive press on black mangrove simulated the encroachment of black mangroves in the *Spartina* dominated marsh-edge. Combined presses showed similar responses of the variables as the single press of black mangrove for all models (Table 6.1, 6.2, 6.3, 6.4.a). There was a change in the response sign between model one and model two from a positive to a negative for penaeid shrimp (Table 6.4.a).

Freshwater diversions were simulated by simultaneous positive presses on turbidity and fine sediment (from the increased sediment deposition of the diverted river waters) and on the vegetation variables (from the freshwater and nutrient input of the river waters). Combined presses predicting the effects of freshwater diversion showed an increase in water column predators and small crabs, but a decrease for grass shrimp in all three models (Table 6.4.b). In model two there was a positive response by penaeid shrimp, and a negative response by benthic fishes (Table 6.4.b). Model three showed decreases in wading birds and penaeid shrimp (Table 6.4.b). There were two strong changes in sign responses between the initial and other models.

I represented shrimping pressure in the Bay negative presses on water column predators and penaeid shrimp and a positive press on turbidity. Shrimping had a positive influence on small crabs and a negative influence for wading birds and algae in all models (Table 6.4.c). Model two showed a negative response of benthic fishes to shrimping pressure (Table 6.4.c). Model two also showed responses from the vegetation variables, *Spartina* and black mangrove, while the other models had no responses for these variables (Table 6.4.c). Model three produced a positive response by benthic fishes (Table 6.4.c). There were some differences among models realized with shrimping simulations, but the finding of wading birds, water column predators, benthic fishes, small crabs, grass shrimp, turbidity, fine sediment, and algae were general across all models.

Wetland loss was simulated by the combined negative presses on the *Spartina* and black mangrove variables and a positive press on turbidity. Wetland loss resulted in a negative response of algae for all three models (Table 6.4.d). Model two also had positive responses from water column predators, penaeid shrimp, and small crabs (Table 6.4.d). Models three differed from models one and two in that it showed positive responses by benthic fish and grass shrimp (Table 6.4.d). All models had different responses to the combined wetland loss press.

DISCUSSION

Qualitative loop modeling is an effective method for gaining insight into factors that influence marsh-edge communities, and is generally useful for exploring management scenarios and discovering the unintended consequences of perturbations (Dambacher et al. 2003,

Table 6.1. The adjoint matrix for the initial model (Fig 6.3) with variables listed. These values are the sum of all feedback loops that influence each node in the model. Presses from one or more column variables are examined by their effect on row variables.

Variables	WB	P	PS	BF	SC	GS	NTU	FS	*Sa*	A	*Ag*
Wading Birds (WB)	18	5	11	-2	11	-7	-3	-5	24	10	-35
Water Column Predator (P)	-6	24	0	-3	0	6	12	42	3	15	30
Penaeid Shrimp (PS)	-3	1	22	-7	-11	-8	6	-1	-15	2	26
Benthic Fishes (BF)	3	-12	0	18	0	-3	-6	12	15	9	-15
Small Crabs (SC)	-3	1	-11	-7	22	-8	6	32	-15	2	26
Grass Shrimp (GS)	9	-3	0	-12	0	24	-18	-63	45	-6	-78
Turbidity (NTU)	6	9	0	3	0	-6	21	57	-3	18	36
Fine Sediment (FS)	0	0	0	0	0	0	0	33	0	0	0
Spartina alterniflora (*Sa*)	0	0	0	0	0	0	0	0	33	0	-33
Algae (A)	6	9	0	3	0	-6	-12	-9	30	18	-30
Avicennia germinans (*Ag*)	0	0	0	0	0	0	0	33	0	0	33

Table 6.2. The adjoint matrix for the second model (Fig 6.5) with variables listed. These values are the sum of all feedback loops that influence each node in the model. Presses from one or more column variables are examined by their effect on row variables.

Variables	WB	P	PS	BF	SC	GS	NTU	FS	*Sa*	A	*Ag*
Wading Birds (WB)	28	0	16	-2	26	-12	6	18	32	10	-44
Water Column Predator (P)	-14	29	-8	1	16	6	26	107	-16	24	80
Penaeid Shrimp (PS)	-2	0	32	-4	-6	-24	12	36	6	20	28
Benthic Fishes (BF)	-4	-29	6	21	-12	10	-34	-73	12	-18	-60
Small Crabs (SC)	-6	0	-20	-12	40	-14	36	108	-40	2	84
Grass Shrimp (GS)	22	0	-4	-14	8	32	-16	-48	50	12	-76
Turbidity (NTU)	4	0	-6	8	12	-10	34	102	-12	18	60
Fine Sediment (FS)	0	0	0	0	0	0	0	58	0	0	0
Spartina alterniflora (*Sa*)	0	0	0	0	0	0	0	0	58	0	-58
Algae (A)	4	0	-6	8	12	-10	-24	-14	46	18	-56
Avicennia germinans (*Ag*)	0	0	0	-0	0	0	0	58	0	0	58

Table 6.3. The adjoint matrix for the third model (Fig 6.6) with variables listed. These values are the sum of all feedback loops that influence each node in the model. Presses from one or more column variables are examined by their effect on row variables.

Variables	WB	P	PS	BF	SC	GS	NTU	FS	*Sa*	A	*Ag*
Wading Birds (WB)	26	5	13	-7	14	1	-11	-31	48	10	-75
Water Column Predator (P)	-8	30	-4	-1	2	6	16	60	1	19	42
Penaeid Shrimp (PS)	3	-1	22	-15	-11	8	-6	-43	15	-2	-26
Benthic Fishes (BF)	4	-15	2	21	-1	-3	-8	11	20	11	-21
Small Crabs (SC)	-6	2	-3	-11	22	-16	12	45	-30	4	52
Grass Shrimp (GS)	9	-3	-16	-4	8	24	-18	-47	45	-6	-78
Turbidity (NTU)	8	11	4	1	-2	-6	25	63	-1	22	40
Fine Sediment (FS)	0	0	0	0	0	0	0	41	0	0	0
Spartina alterniflora (*Sa*)	0	0	0	0	0	0	0	0	41	0	-41
Algae (A)	8	11	4	1	-2	-6	-16	-19	40	22	-42
Avicennia germinans (*Ag*)	0	0	0	0	0	0	0	41	0	0	41

Table 6.4. Combined press results for each model as calculated from the adjoint matrices of each model a) black mangrove encroachment with - *Sa* and + *Ag,* b) freshwater diversion with + NTU, FS, *Sa*, and *Ag*, c) shrimping pressure with – P and PS and + NTU, and d) wetland loss with – *Sa* and *Ag* and + NTU. Pressed variables are in bold and are not used for response results.

a)

Mangrove Encroachment	Model 1	Model 2	Model 3
Wading Birds (WB)	-59	-76	-123
Water Column Predator (P)	27	96	41
Penaeid Shrimp (PS)	41	22	-41
Benthic Fishes (BF)	-30	-72	-41
Small Crabs (SC)	41	124	82
Grass Shrimp (GS)	-123	-126	-123
Turbidity (NTU)	39	72	41
Fine Sediment (FS)	0	0	0
***Spartina alterniflora* (*Sa*)**	**-66**	**-116**	**-82**
Algae (A)	-60	-102	-82
***Avicennia germinans* (*Ag*)**	**33**	**58**	**41**

b)

Freshwater Diversion	Model 1	Model 2	Model 3
Wading Birds (WB)	-9	-185	-59
Water Column Predator (P)	102	67	137
Penaeid Shrimp (PS)	18	170	-62
Benthic Fishes (BF)	15	-135	13
Small Crabs (SC)	51	5	83
Grass Shrimp (GS)	-120	-213	-104
Turbidity (NTU)	**129**	**55**	**149**
Fine Sediment (FS)	**33**	**-22**	**41**
***Spartina alterniflora* (*Sa*)**	**0**	**-143**	**0**
Algae (A)	-3	-138	-15
***Avicennia germinans* (*Ag*)**	**66**	**94**	**82**

Table 6.4 (Continued)

c)

Shrimping Pressure	Model 1	Model 2	Model 3
Wading Birds (WB)	-19	-70	-29
Water Column Predator (P)	**-12**	**64**	**-18**
Penaeid Shrimp (PS)	**-17**	**100**	**15**
Benthic Fishes (BF)	6	-82	21
Small Crabs (SC)	16	4	11
Grass Shrimp (GS)	-15	-102	1
Turbidity (NTU)	**12**	**66**	**10**
Fine Sediment (FS)	0	0	0
Spartina alterniflora (*Sa*)	0	-66	0
Algae (A)	-21	-54	-31
Avicennia germinans (*Ag*)	0	54	0

d)

Wetland Loss	Model 1	Model 2	Model 3
Wading Birds (WB)	8	-17	16
Water Column Predator (P)	-21	19	-27
Penaeid Shrimp (PS)	-5	18	5
Benthic Fishes (BF)	-6	-13	9
Small Crabs (SC)	-5	7	-10
Grass Shrimp (GS)	-141	-21	15
Turbidity (NTU)	**54**	**11**	**-14**
Fine Sediment (FS)	0	0	0
***Spartina alterniflora* (*Sa*)**	**0**	**-11**	**0**
Algae (A)	-12	-4	-14
***Avicennia germinans* (*Ag*)**	**-33**	**26**	**-42**

Dambacher and Ramos-Jiliberto 2007). Using these models, researchers can unravel direct and indirect interactions among biotic and abiotic variables to formulate hypotheses, design experiments, or identify specific data needs. Models clarify important and interesting relationships among environmental and faunal variables such as turbidity's influences on the nekton community. The strongest influences in Louisiana marsh-edge systems were related to environmental and floral variables on faunal variables (Tables 6.1, 6.2, 6.3). This illustrates how these estuarine-dependent species are tied to their environments and dominant vegetation and how changes to the environment result in direct or indirect effects on the fauna. The environmental variable with the most influence on other variables was turbidity. It was also involved in combination presses to simulate freshwater diversion, shrimping pressure, and wetland loss demonstrating the significant role it plays in the Barataria Basin system. Nevertheless, it has been unappreciated in Louisiana and no long-term records appear to exist across the northern Gulf of Mexico (Turner 2001). Positive presses on turbidity caused positive responses of the nekton as that have been corroborated in nature (Cyrus and Blaber 1992). This strong influence was expressed across all of the models and warrants a closer examination of the hypothesis that high turbidity is a trait of high quality nursery habitat in southeastern Louisiana (Chesney et al. 2000).

The models were a good representation of the food-web dynamics of the marsh-edge community of lower Barataria Bay. While the changes in links and connections showed some variations in responses to presses, they were most numerous for model two. Model two had the most differences in responses for the freshwater diversion and shrimping presses, but was similar to the other models for the other presses. The commonalities among the responses of variables to the complex presses for all of models indicate that they are general and realistic representations of the marsh-edge community in Barataria Bay, Louisiana. The mangrove encroachment press showed the greatest agreement in responses among all models. With the exception of a negative response of penaeid shrimp in model three, all models showed grass shrimp having a strong negative response to increased mangrove and decreased *Spartina*. Chapter 5 (Fig 5.3.b) indicates grass shrimp were less associated with black mangroves. For freshwater diversions the models anticipate an increase in water column predators and small crabs and a decrease in grass shrimp. Grass shrimp are a considerable food source for many species that showed a positive response to the freshwater diversion (Anderson 1985, Kneib 1987).The shrimp fisheries had a common response of increases in small crabs and decreases in wading birds and algae. Wetland loss is expected to cause a decline in algae across all models. This press showed the largest disconnect of the three models. The next step in making general and realistic predictions is to develop a final basic model and explore interesting variations around that model to discover basic truths about the system and indentify critical data needs to build good quantitative models.

The Bay has multiple long-term perturbations, but there were four changes that have been ongoing for a while and will continue in the study area. I have examined these individually as complex presses of sets of individual variables. It is also apparent that those and many other perturbations are occurring simultaneously, and may be additive as positive and/or negative effects, or result in synergistic effects. Habitat changes in coastal Louisiana and the northern Gulf of Mexico are numerous and the effects of some perturbations may be overlooked while other negative effects are attributed solely to a major perturbation without appreciating the additive and synergistic nature of multiple insults. Jackson et al. (2001) reveal that fishing is an early and long standing influence that is widely unappreciated. It has preceded habitat destruction, pollution, the introduction of exotic species and climate change (Jackson et al. 2001, Fig 3). Louisiana has had and will continue to endure multiple perturbations in the form of fishing, pollution, or sea-level rise (Chesney et al. 2000). These perturbations increase patchiness in the wetlands and in turn increase marsh-edge habitat, which is a valuable nursery feature for many estuarine-dependent species (Baltz et al. 1993). Because of the increased nursery function, nekton densities remain high while other habitat qualities decrease. With these changes it is important to consider the shifting baseline syndrome with long-term data (Pauly 1995). The environmental conditions we are currently experiencing are the result of multiple interacting press perturbations not necessarily the specific result of current resource management.

Qualitative modeling of multiple long-term perturbations will help to determine which interactions have been important in causing responses and how alternative management options will influence systems.

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APPENDIX A
LIST OF ALL NEKTON ABUNDANCES BY YEAR

Species/Genera/Family	Common Name	2005	2006	2007
Alpheus spp.	snapping shrimp	23	65	35
Anchoa spp.	anchovy	7	2	0
Anchoa hepsetus	striped anchovy	44	101	7
Anchoa mitchilli	bay anchovy	301	54	123
Ariopsis felis	hardhead catfish	0	1	0
Bairdiella chrysoura	silver perch	1	3	12
Bathygobius soporator	frillfin goby	5	16	1
Brevoortia patronus	gulf menhaden	2	1885	111
Callinectes sapidus	blue crab	429	583	743
Callinectes similis	lesser blue crab	196	616	846
Chaetodipterus faber	Atlantic spadefish	0	0	1
Citharichthys spilopterus	bay whiff	12	17	0
Clibanarius vittatus	thin striped hermit crab	1137	1425	799
Ctenogobius boleosoma	darter goby	305	266	270
Cynoscion arenarius	sand trout	0	9	0
Cynoscion nebulosus	spotted trout	2	6	12
Dasyatis americana	stingray	0	0	3
Elops saurus	ladyfish	0	0	1
Eurypanopeus depressus	flatback mud crab	13	6	55
Farfantepenaeus aztecus	brown shrimp	70	187	93
Gerridae	mojarra	2	0	1
Gobiesox strumosus	skilletfish	333	187	129
Gobiidae	goby	2	5	2
Gobiosoma bosc	naked goby	92	24	31
Gobiosoma robustum	code goby	0	0	14
Hypleurochilus geminatus	crested blenny	45	25	40
Lagodon rhomboides	pinfish	0	2	0
Leiostomus xanthurus	spot	0	7	61
Litopenaeus setiferus	white shrimp	14	61	15
Lutjanus griseus	grey snapper	1	4	10
Majoidae	arrow crab	0	59	65
Membras martinicas	rough silverside	0	60	9
Menidia beryllina	inland silverside	1	0	1
Menippe adina	gulf stone crab	44	44	23
Menticirrhus americanus	southern kingfish	1	1	4
Microphis brachyurous	pipefish	0	5	4
Micropogonias undulatus	Atlantic croaker	85	431	62
Mugil cephalus	striped mullet	2	1	99

Mugil curema	white mullet	4	0	0
Myrophis punctatus	speckled worm eel	15	14	8
Neoconger mucronatus	ridged eel	0	0	1
Opsanus beta	gulf toadfish	1	5	3
Ovalipes floridanus	Florida lady crab	0	2	1
Pagurus longicarpus	longwristed hermit crab	0	142	48
Palaemonetes spp.	grass shrimp	2858	3483	2962
Panopeus obesus	saltmarsh mud crab	2	1	12
Panopeus simpsoni	oystershell mud crab	58	137	85
Paralichthys lethostigma	southern flounder	0	5	16
Penaeidae	penaeid shrimp	7	0	16
Persephona spp.	purse crab	1	0	0
Pogonias cromis	black drum	1	1	1
Pomacanthidae	angelfish	0	0	2
Rhithropanopeus harrisii	estuarine mud crab	8	30	13
Sciaenidae	drum	4	6	7
Sciaenops ocellatus	red drum	0	1	2
Sphoeroides parvusd	least puffer	0	0	3
Strongylura marina	Atlantic needlefish	0	1	0
Syacium papillosum	dusky flounder	1	0	0
Symphurus plagiusa	blackcheek tonguefish	30	39	26
Syngnathus louisianae	chain pipefish	7	2	0
Syngnathus scovelli	gulf pipefish	0	1	0
Synodus foetens	inshore lizardfish	4	9	4
Xanthidae	mud crab	339	392	116

APPENDIX B
PERMISSION REQUEST AND RESPONSE

I am writing to ask for permission to put my article (Estuaries and Coasts, Article 9135; Short-term effects of an oil spill on marsh-edge fishes and decapod crustaceans) in my dissertation as a chapter.
I have graduate school deadlines and need an email giving me the permission to have it as a chapter.
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Arie

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Agatha-Marie (Arie) Fuller Roth was born in October 1975, in New Orleans, Louisiana, to Mr. and Mrs. (Cindy) Richard J. Roth, Jr. She graduated from the Academy of the Sacred Heart in 1994. She earned her Bachelor of Science in biology and marine sciences with a concentration in botany from the University of Alabama, Tuscaloosa, in 1999. She was granted a Master of Science in biology by the graduate faculty at Southeastern Louisiana University, Hammond, in 2003. She will receive her degree of Doctor of Philosophy with the major of oceanography and coastal sciences and marry Matthew John Kaller May 2009.