SPATIAL AND TEMPORAL PATTERNS OF *Lycium carolinianum* Walt., THE

CAROLINA WOLFBERRY, IN THE SALT MARSHES OF ARANSAS NATIONAL

WILDLIFE REFUGE, TEXAS

A Thesis

by

RACHEL ELIZABETH BUTZLER

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

May 2006

Major Subject: Wildlife and Fisheries Sciences

SPATIAL AND TEMPORAL PATTERNS OF *Lycium carolinianum* Walt., THE

CAROLINA WOLFBERRY, IN THE SALT MARSHES OF ARANSAS NATIONAL

WILDLIFE REFUGE, TEXAS

A Thesis

by

RACHEL ELIZABETH BUTZLER

Submitted to the Office of Graduate Studies of
Texas A&M University
in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

Approved by:

Chair of Committee,	Stephen E. Davis, III
Committee Members,	R. Douglas Slack
	Fred E. Smeins
Head of Department,	Robert D. Brown

May 2006

Major Subject: Wildlife and Fisheries Sciences

ABSTRACT

Spatial and Temporal Patterns of *Lycium carolinianum* Walt., the Carolina

Wolfberry, in the Salt Marshes of Aransas National Wildlife Refuge, Texas.

(May 2006)

Rachel Elizabeth Butzler, B.S., The Pennsylvania State University

Chair of Advisory Committee: Dr. Stephen E. Davis, III

Understanding the salt marsh ecosystem in the Guadalupe Estuary is needed because wetlands in this system support the endangered whooping crane (*Grus americana*). The marsh plant research and monitoring described herein were based in the salt marshes at Aransas National Wildlife Refuge (ANWR), which are utilized by the cranes each winter. Past research indicates that the Carolina wolfberry (*Lycium carolinianum)* contributes 21-52% of crane energy intake early in the wintering period (Chavez 1996). Beginning in Fall 2003, vegetation transects were sampled along an estuarine gradient at ANWR. Species diversity and composition was similar at the three sites, with all sites containing the same 6-7 common species. While *Spartina alterniflora* is only a minor part of this vegetation community, it dominates the few low inter-tidal, fringe areas present. Species composition exhibited little variability from Year 1 to Year 2 of the study. Densities and biomass of *L. carolinianum* were not significantly different between sites or years. *L. carolinianum,* while important to salt marsh ecology, accounts for only a small portion of the overall productivity. Based on correlation coefficients, *L. carolinianum* was found in association with some of the

common species in the vegetation community, indicating that its growth and survival requirements are typical to the salt marshes at ANWR. Also beginning in Fall 2003, I repeatedly sampled *L. carolinianum* in permanent plots along the estuarine gradient. *L. carolinianum* exhibits strong temporal patterns. Leaf production peaked in early spring and again just prior to peak berry abundance. Flowering of *L. carolinianum* occurred in October and November. Peak berry abundance coincided with the cranes' arrival in late October and early November. Berry production occurred in October, November, and December; berries were virtually non-existent in the marshes for the remainder of the year. Stepwise regression showed stem diameter alone was a good estimator of aboveground biomass of this species in ANWR marshes, accounting for 94% of the variability ($p<0.001$). Changes in aboveground biomass followed no distinct patterns in the year of monitoring, perhaps due to the woody stem of the plant. Spatial patterns in *L. carolinianum* were not explained by water quality parameters alone; it is suggested that soil properties may help to account for the spatial variability.

ACKNOWLEDGEMENTS

First and foremost, I would like to thank my committee chair, Dr. Steve Davis, for taking me on as one of his first graduate students. Without his continual support and encouragement throughout my time at Texas A&M, I would not have had the great experiences in graduate school that I was fortunate to have. I would also like to thank my committee members, Dr. Slack and Dr. Smeins, for their support throughout the course of my research and discussion of my thesis.

I have also received help from many people, without whom I would still be tramping through the marsh collecting plants, or at least running them through the dryer. Tyson Hart – you were essential in getting my project off the ground, and Melissa Romigh in getting me through my first Texas summer and my first semester of graduate school. Other friends and colleagues who helped me with my research are: Kristin LaFever, Tim Assal, Carrie Miller, Peter Krawczel, Matt Driffill, Elizabeth Hebert, Kim Crumpler, Leslie Schwierzke, Michael Parkes, and David Hoeinghaus. I appreciate every single minute that you helped me out.

Many other people have supported me in one way or another. Drs. Bill Grant, Doug Slack, and Steve Davis, along with Kristin LaFever, Karine Gil de Weir, Magui Mieres, and Amy Snelgrove from the SAGES project provided me positive feedback, suggestions, and technical help during various stages of my research. The staff at ANWR has helped make my times in the field as problem-free as fieldwork can be. The WFSC staff has also been supportive: handling my travel vouchers, having chocolate in your office, and helping with logistical nightmares. Thanks also to Dr. Gatlin who

helped proofread my thesis.

Thanks to my family – without whom I would not be the person I am today; Marge and Bud – for always believing in me; and to my sisters Kathy, Jules, and Mo – for your never-ending support and love, your ability to talk on the phone for hours, and fairly infrequent care packages. And thanks to my brothers, Tom and Stevie, for not forgetting about me even though I am so far away.

Thanks also to my friends – without your help outside of school, Texas would not have been nearly as enjoyable as it was. There will be many memories of my time with you that I will treasure forever.

And finally, special thanks and love to Peter Krawczel. I know I dragged you to Texas, but your emotional, physical, and mental support have been essential to my success in Texas and in life. Thanks for sticking with me (even when the B-pants came out of the closet) and still wanting to marry me….

TABLE OF CONTENTS

LIST OF FIGURES

FIGURE   Page

LIST OF TABLES

CHAPTER I

INTRODUCTION

Tidal salt marshes are commonly found in estuaries, and are driven to a great extent by the balance of freshwater inflows and the tidal flushing of saline water. This balance is critical to species that inhabit the estuary. While they are adapted to a certain degree of variability in water quality, their well-being or survival may be threatened in severe conditions such as drought or extreme flooding. Natural seasonal or inter-annual variability can be exacerbated by anthropogenic effects such as hydrologic manipulations.

Because of an increased freshwater demand for human consumption, agricultural purposes, and other uses, there has been a decrease in freshwater inputs into Texas estuaries (Dunton et al. 2001) and estuaries nationwide (Jassby et al. 1995). In fact, a recent study by the National Wildlife Federation reported that 5 of the 7 major Texas estuaries were rated in the "danger" level; danger predicts at least a 67% increase in periods of low critical flow for at least one of 2 ecological criteria (Johns et al. 2004). The first criteria addresses the freshwater inflows required to maintain salinity levels in a reasonable range for key species, and the second criteria the necessary freshwater pulses that naturally occur in the spring and early summer. This modeling effort assumed that all water rights were completely utilized, and water reuse (common practice in agricultural and industrial practices) increased by 50%. This study predicted the

_____

This thesis follows the style of *Wetlands*.

Guadalupe Estuary will experience a 250% increase in the first criteria and a 26% increase in the second criteria (Johns et al. 2004).

The Guadalupe Estuary is important economically and recreationally to the state of Texas, and is the most important of all Texas estuaries in regards to commercial fisheries (Tanyeri-Abeu et al. 1998). It is primarily fed by the Guadalupe River, which accounts for about 70% of its gauged flow, and the San Antonio River, which contributes about 26% of its freshwater flow (Orlando et al. 1993). The Guadalupe Estuary is shallow, with a mean depth of 1.1 m, and has a relatively small tidal range (0.18 m) compared to other Gulf of Mexico estuaries (Bianchi et al. 1999). In the next 50 years, as the population of Texas is expected to double, upstream water demands will increase (Johns et al. 2004). The NWF (Johns 2004) predicted that all permits will eventually use the total amount of water allowed, and more permits are currently in the application process.

The Lower Guadalupe Water Supply Project (LGWSP) was formed by a group of local water agencies to provide additional water to meet the growing needs of South Texas, while maintaining critical estuarine and spring inflows. The LGWSP proposes a withdrawal of freshwater from just below the confluence of the San Antonio and Guadalupe Rivers to meet projected municipal demands in the rapidly expanding San Antonio metropolitan area. Critical inflows are inflows at a level that supports a healthy estuary which in turn supports a great diversity of species. Probably the most well-known species dependant on the Guadalupe Estuary is the endangered whooping crane (*Grus americana*) that winters on this part of the Texas coast. In order to understand the

potential impacts of this diversion on the crane's wintering grounds—as required under the Endangered Species Act—the agencies of the LGWSP have funded an ecosystem project, the San Antonio Guadalupe Estuary System (SAGES) project, to study the influence of freshwater inflows on the salt marsh ecosystem at ANWR. This thesis project is part of a larger ecosystem project that seeks to identify the environmental drivers of change in the Guadalupe estuarine system, and how various plant and animal species respond to these changes.

The SAGES conceptual model (Figure 1) illustrates how the various components of the project interact. The environmental drivers affect the system in number of ways. I focused on salinity and inundation, because if there is an effect of a water diversion on the estuarine system, I predict it will become apparent in salinity and marsh inundation patterns. Weather and tidal patterns affect both the salinity gradient and inundation regimes in the system: these in turn affect the trophic components, i.e., survival and productivity of the plant Carolina wolfberry (*Lycium carolinianum)* and blue crabs (*Callinectus sapidus)*, known food sources for the whooping crane. Like any estuarine macrophyte species, *L. carolinianum* may be susceptible to salinities outside of its optimal range, and as a plant, may be restricted by water level.

Salinity and inundation patterns can also affect the cranes behavior. Cranes have limits to water depth for foraging, based on their bill length (Chavez 1996); if the water in the ponds is too deep for the cranes, or berries are more readily available, cranes may spend more time foraging in the marsh vegetation, and less time in the ponds.
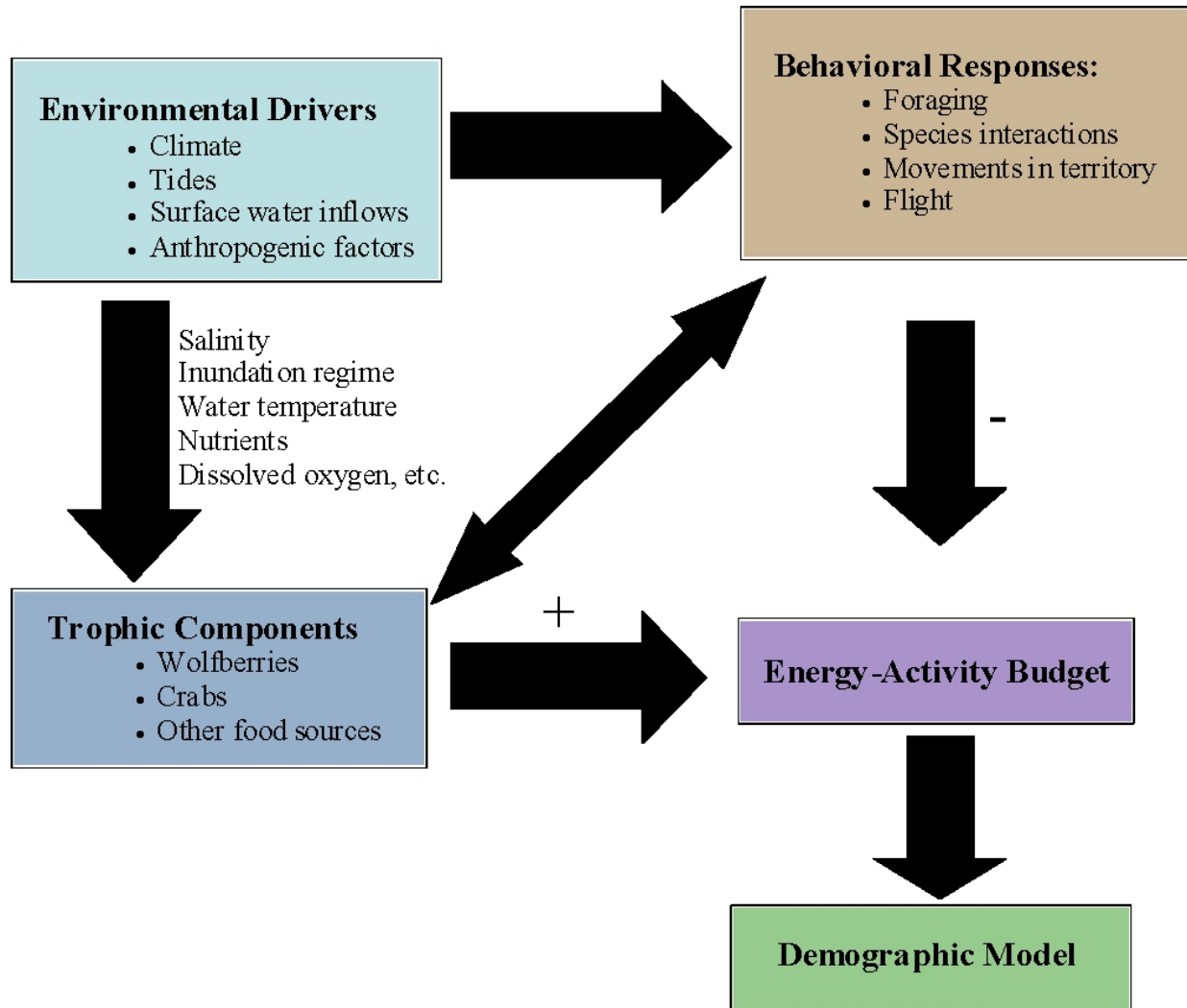
Figure 1: SAGES project conceptual model
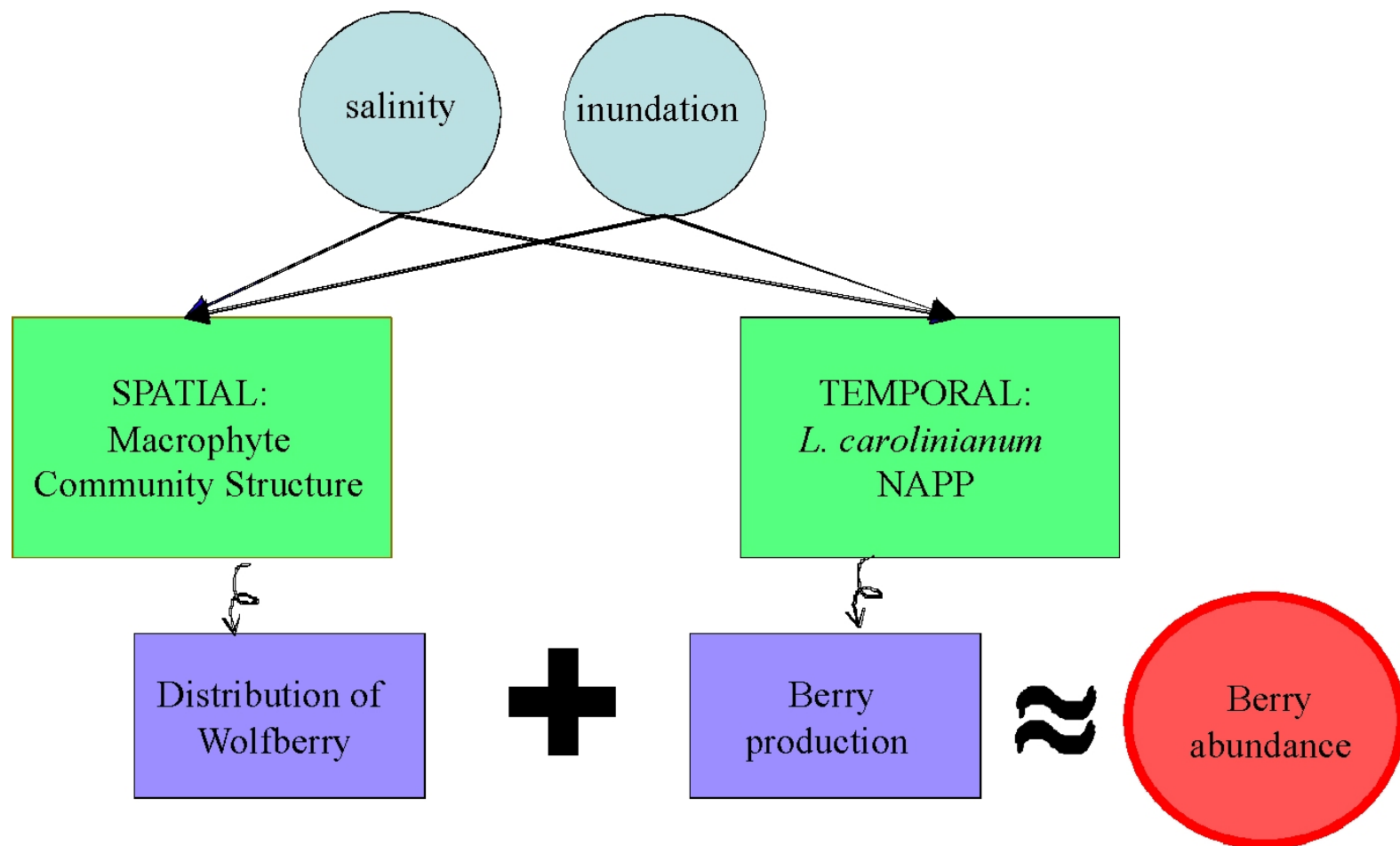
Figure 2: Macrophyte sub-project conceptual model.

The berries produced by *L. carolinianum* fall under "trophic components" and are a positive contribution to a crane's energy balance. Crane activities, such as flight and aggressive behavior, require energy expenditures (a negative value). In combining the trophic and behavioral components, an energy activity budget will be estimated, which will help in the formation of a population model. Ultimately, the SAGES project will have a better understanding of how various environmental drivers, such as inundation or salinity levels, affect the whooping crane population.

The conceptual model of the macrophyte community (Figure 2) illustrates the type of data collected and how these data will fit into the SAGES model. I collected both spatial (community composition and *L. carolinianum* distribution) and temporal (berry abundance and *L. carolinianum* growth patterns) data to understand how *L. carolinianum* fits into the salt marsh ecology at ANWR.

Lycium carolinianum is a bushy branched shrub in the Solanaceae family (Godfrey and Wooten 1981, Stutzenbaker 1999) (Figure 3). *L. carolinianum* is often included in descriptions of the salt marsh vegetation communities surrounding the Gulf of Mexico (Carlton 1975, McAlister and McAlister 1995, Stutzenbaker 1999). This species is found on the edge of or in brackish marshes where the bases are inundated by the high tides, or just above high tide (Godfrey and Wooten 1981).

This plant can grow up to between 0.9m (Stutzenbaker 1999) and 3.0m tall (Godfrey and Wooten 1981). Main branches often bear shorter thorn-like branches (Godfrey and Wooten 1981, Stutzenbaker 1999) (Figure 4). The simple leaves are small and succulent (Godfrey and Wooten 1981, McAlister and McAlister 1995, Stutzenbaker

Figure 3: Pictures of *L. carolinianum,* the Carolina wolfberry.   Left photo shows *L. carolinianum* in the field. The right photo was taken in the lab to show the plants more clearly. These to plants also illustrate the plants ability to have extensive or little branching.

Figure 4: *L. carolinianum* leaves and thorns. Top photo shows leaves on the plant and thorns.
 Bottom photo illustrates the variability in leaf size.

Figure 5: *L. carolinianum* flowers and fruit.

1999). This plant, unlike species like *D. spicata, S. alterniflora,* and *M. littoralis*, does not form large monotypic stands (Stutzenbaker 1999). It's blue to lavender flowers produce a red ellipsoid berry, which can range between 8 and 15mm long (Figure 5 (Godfrey and Wooten 1981, Stutzenbaker 1999). *L. carolinianum* is a hearty plant able to withstand drought and flooding by salt water (Stutzenbaker 1999).

While there are numerous taxonomic descriptions, and brief mentions in an overall community description, there are very few studies focusing on the plant, and therefore, little in-depth information of the plant. Contrary to it's appearance in other systems (big, bushy, many leaved) this plant's physical appearance in the marshes ANWR does not always conform to the typical description of the plant. It is a "straggly drooping plant" thought to be found amidst stands of *B. frutescens* (McAlister and McAlister 1995). The vegetation community at ANWR is very thick and shrubby; at first glance it was often difficult to discern *L. carolinianum* from the dense community (Figure 3). The plant is easier to pick out when the fruit or flower is present (Figure 5).

The following two chapters will examine spatial and temporal patterns in the salt marsh vegetation community at ANWR, focusing on *L. carolinianum.* The specific objectives are to determine the distribution patterns of *L. carolinianum* in the marshes at ANWR, and to determine the fruit production patterns throughout the year, and along the estuarine gradient.

CHAPTER II

DISTRIBUTION OF *Lycium carolinianum* Walt.,

THE CAROLINA WOLFBERRY, IN THE SALT MARSHES OF

ARANSAS NATIONAL WILDLIFE REFUGE, TEXAS

SYNOPSIS

Understanding the implications of a proposed freshwater diversion from the

Guadalupe Estuary is needed because the wetlands in this system support the endangered

whooping crane (*Grus americana*). My marsh plant research and monitoring efforts have

targeted the salt marshes at Aransas National Wildlife Refuge (ANWR) utilized by the

cranes each winter. Past research indicates that Carolina Wolfberry (*Lycium

carolinianum)* contributes 21-52% of crane energy intake early in the wintering period

(Chavez 1996). Beginning in Fall 2003, I sampled vegetation via transects along an

estuarine gradient at Aransas National Wildlife Refuge (ANWR). Species diversity and

composition is similar at the three sites, with all sites sharing 6-7 common species.

There are distinct "low marsh" and "high marsh" zones throughout the marsh landscape;

while *Spartina alterniflora* is only a minor part of this vegetation community, it

dominates the few low inter-tidal, fringe area areas present on the landscape. Species

composition exhibited very little variability from year one to year two of the study.

Biomass values were not significantly different between sites or years. Densities and

biomass of *L. carolinianum* were not significantly different between sites or years either.

L. carolinianum, while important to salt marsh ecology, accounts for only a small

portion of the overall productivity. *Batis maritima, Borrichia frutescens,*

Monanthochloa littoralis, and in the second year, *Distichlis spicata* were species more likely to be found with *L. carolinianum.* Based in these correlations, *L. carolinianum* is found in association with some of the common species, indicating that its growth and survival requirements are typical to the salt marshes at ANWR.

INTRODUCTION

Tidal salt marshes are a common component of estuaries which are formed and maintained by a number of factors, including the balance of freshwater inflow and the tidal flushing of saline water (Odum 1988, Jassby et al. 1995, Baldwin et al. 2001, Kennish 2001). The resulting spatial and temporal patterns in estuarine water quality brought about by this mixing are important in structuring biotic communities. Salinity, temperature, and dissolved oxygen vary seasonally and inter-annually in estuarine systems, and can significantly affect the plant, animal, and planktonic communities (Odum 1988, Jassby et al. 1995, Gough and Grace 1998, Kennish 2001). Salinity, inundation patterns, and nutrient limitation exert strong control over species distribution, growth patterns, and productivity of salt marsh plants (Smart and Barko 1980, Webb 1983, Pennings and Callaway 1992, David 1996, Dunton et al. 2001,).

Both short and longer term studies (from 35 days to 2 years) in the field and greenhouse have shown that changes in salinity can affect the estuarine macrophyte community (Zedler 1983, McKee and Mendelssohn 1989, Howard and Mendelssohn 2000). These effects vary according to species and the magnitude and duration of salinity change (Zedler 1983, McKee and Mendelssohn 1989, Howard and Mendelssohn

2000). Significant and extended changes in salinity levels can result in shifts in species composition, mass plant mortality (Kennish 2001), decreased species diversity (Gough and Grace 1998, Howard and Mendelssohn 2000) and a reduction in plant community biomass (Gough and Grace 1998). In a California estuary under flooded and high flow conditions, Zedler (1983) found that short term changes in salinity affected productivity with no major changes in species composition, and a quick return to pre-flood conditions. The long-term freshwater inundation (throughout the entire growing season) caused mortality and a shift to freshwater species, and was experiencing a very slow recovery to pre-flood conditions (Zedler 1983).

Oftentimes, the effects of salinity seem more pronounced due to an interaction with other variables, such as inundation (McKee and Mendelssohn 1989), elevation (Pennings and Callaway 1992) or soil properties (Silvestri et al. 2005). In the Nueces River Estuary (TX), Dunton et al. (2001) found significant changes in plant biomass, percent cover, and aboveground:belowground biomass in several marsh plant species with increased inundation and reduced salinity resulting from above average precipitation and upland runoff. In the salt marshes of the Venice Lagoon, Italy—a site characterized by both low and high marsh zones—Silvestri et al. (2005) concluded that salinity and elevation along with edaphic factors (resulting from tides) determined marsh plant zonation.

The goal of my research was to determine spatial and temporal patterns in a portion of the marsh macrophyte community in the Guadalupe River Estuary, TX, and identify the factors driving these patterns. Estuaries provide critical habitat for many

species, and understanding these systems is essential to predict how they will respond to both natural and anthropogenic perturbations such as hurricanes, drought, changes in freshwater inflows, eutrophication, etc.   In summers 2003 and 2004, I assessed the spatial variability in the marsh plant community with a focus on *Lycium carolinianum*, the Carolina Wolfberry. This plant produces berries each winter (Chapter III), which are a food source for a number of marsh animals (Godfrey and Wooten 1981, Chavez 1996). First, I wanted to determine if *L. carolinianum* abundance varied between sites along an estuarine gradient, and second, determine what vegetation community is associated with *L. carolinianum*. Lastly, I wanted to determine if there are any environmental parameters collected in this study that affect *L. carolinianum* distribution, or can help explain its distribution patterns.

With a rapidly growing population and an increased demand for freshwater resources, there has been a general decline in freshwater inputs into Texas estuaries (Dunton et al. 2001) and estuaries nationwide (Jassby et al. 1995). Reduced freshwater input combined with rising sea level, may significantly affect estuarine salinity gradients and inundation patterns, especially in micro-tidal, lagoonal estuaries like the Guadalupe Estuary, located on the Gulf Coast of Texas (Figure 1).

STUDY SITE

The Coastal Bend region on the Gulf Coast of Texas includes numerous bays and estuaries that are ecologically and economically important. One such estuary, the Guadalupe Estuary, is fed by freshwater inflow from the Guadalupe and San Antonio

rivers. The ANWR is located in this estuary and represents a major stretch of

undeveloped land along the southwestern edge of the estuary (Figure 6). There are

approximately 2800 hectares of salt marsh at ANWR that provide food and habitat to

many terrestrial and estuarine organisms; most well known is the endangered whooping

crane (*Grus americana)* that over-winter in Guadalupe Estuary salt marshes each year.

The gulf coast of Texas is a haven for migrating and wintering birds. One of the

most well known species, the whooping crane (*Grus americana*), migrates from Canada

to the salt marshes of the Guadalupe Estuary in the fall, arriving between late October to

mid-November (USFWS 2005). They maintain territories for the duration of the winter

meaning they select a parcel of land to occupy and defend against other whooping cranes

(USFWS 2005); the cranes have been found to return to and defend the same territory in

consecutive years (Bishop 1984, Stehn 1992). It is not completely understood how the

cranes chose their territories, but several studies (Bishop 1984, Chavez 1996, Bonds

2000) have indicated a need to understand habitat use and food availability in order to

further understand crane ecology and population success.

While the cranes are on their wintering grounds, they rely on blue crabs

(*Callinectus sapidus*) and the Carolina Wolfberry (*Lycium carolinuanum*) as significant

food sources (Chavez 1996). In fact, *L. carolinianum* produces berries that account for

21-52% of the cranes energy intake early in the wintering period (Chavez 1996). There

is little known about this plant's functional role, but Chavez (1996) noted that the berries

are most abundant when the cranes arrive each winter. Chavez's (1996) data indicate

high year-to-year variability in berry abundance in the early winter months, but he did
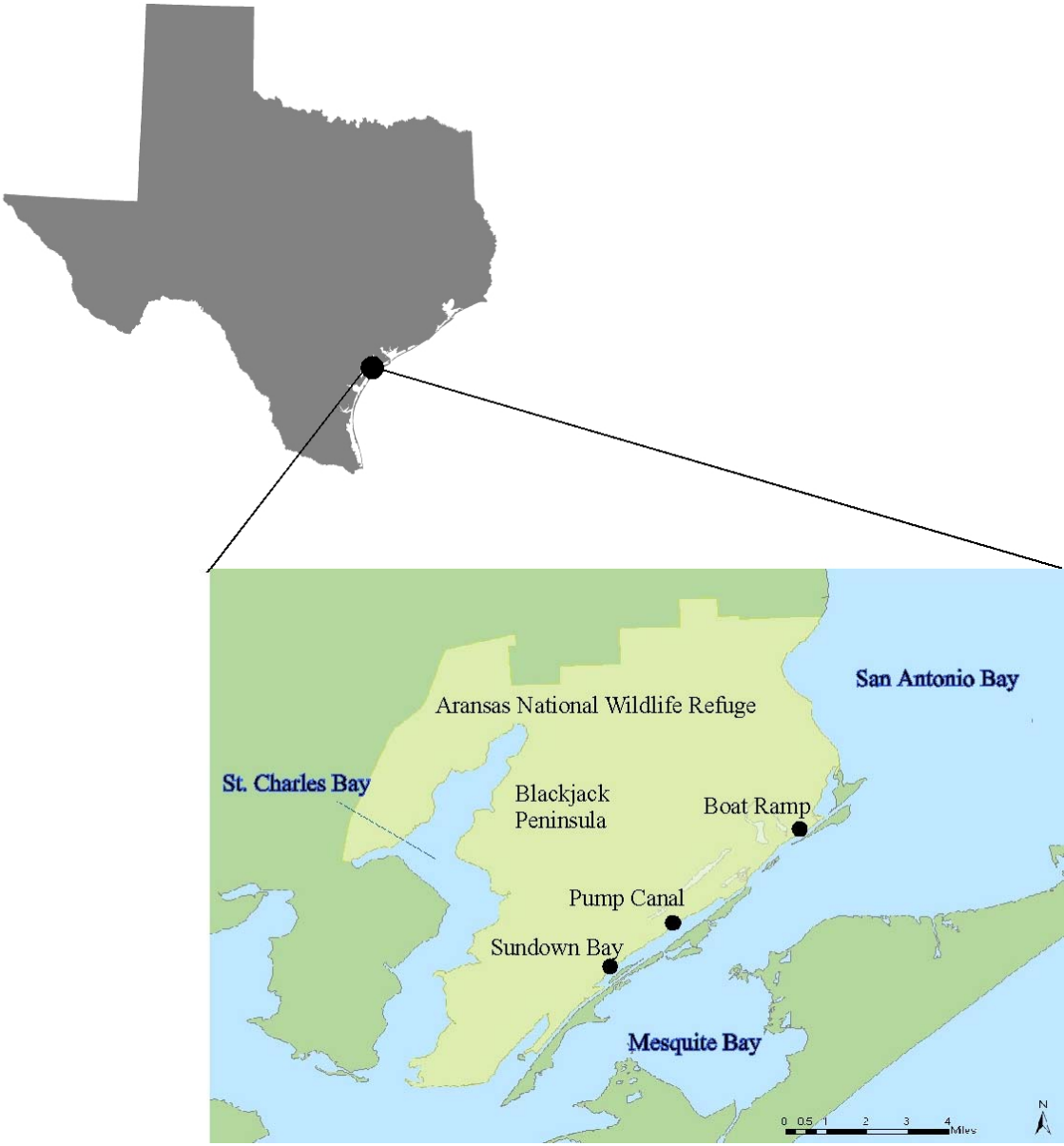
Figure 6: Location of Aransas National Wildlife Refuge along the Coastal Bend Region in Texas and approximate location of three sampling sites.

not examine the distribution of the plant throughout the marsh landscape or abundance of berries in the different territories.

METHODS

Sampling took place on a number of scales, and to monitored both spatial and temporal patterns. Due to varying sampling techniques for the various types of data collected, and conceptual diagram of sampling procedures is provided (Figure 7) to introduce terminology that will be used throughout this thesis. The black box enclosing the diagram is representative of the *site*. The site refers to the general location in which all sampling took place. I selected three sites for sampling, Boat Ramp Pump, Canal, and Sundown Bay. This diagram is representative of one site.

The yellow outline represents the *crane territory*. Whooping cranes tend to migrate as a family unit or pairs, and upon arrival at ANWR, will select an area of marsh to occupy for the winter (USFWS 2005). A territory can be defined as "defended space" (Giles 1978). While the family unit may tolerate other cranes in close proximity to their selected area, most of the time the cranes will defend their territory against intrusion from other whooping cranes (USFWS 2005). There is a behavioral change in the individual when other whooping cranes cross the "line" or boundary of its territory (Bonds 2000). The size/shape/location of these territories can be estimated from crane behavior and spatial census data gathered during weekly flyovers by ANWR researchers (as in Bonds 2000).

Figure 7: Schematic illustrating the various sampling terminology utilized during this study. This figure is not to scale, and is intended to clarify the terminology used in this thesis.

The red dashed lines indicate sampling along vegetation transects (spatial). This sampling took place on an annual basis. These transects began at the water's edge and ran perpendicular to the marsh edge until upland species were reached. The black boxes along each transect represent 0.25m^2 quadrats where vegetation was harvested; these quadrats will be referred to as points along a transect. The white boxes represent permanent vegetation plots (temporal). In these plots species composition was recorded monthly, along with extensive measurements of *L. carolinianum* to track growth patterns. The pink box designates the location of the water quality platform. This platform housed equipment that monitored surface water quality (temperature, DO, salinity, and pH) in the tidal creek, and collected water samples every 18 hours.

Environment/vegetation sampling Three sites were sampled along the length of the Blackjack Peninsula at ANWR for field monitoring of the marsh macrophyte community (Figure 6). Boat Ramp was closest to the Guadalupe River input, followed by Pump Canal, and Sundown Bay. These sites were representative of the variability in salinity, elevation, and vegetation cover types typically found throughout salt marshes along the Blackjack Peninsula. In tidal creeks nearby to each sampling site, I monitored surface water (temperature, dissolved oxygen, pH, and conductivity) every half hour for the duration of this study using calibrated Hydrolab mini-sondes.

In late summer 2003 and 2004, I sampled vegetation along duplicate transects at each of the three sites. Each transect started at the interface of the bay and ended at the upland transition zone, usually indicated by the presence of *Spartina spartinae*. The length of each transect depended on where *S. spartinae* was found, so transect length

varied. Duplicate 0.25 m^2 quadrats were randomly tossed approximately every 50 m, approximate location indicated by black boxes along the transect path, and live aboveground biomass was harvested. Plants were separated, counted, and dried at 60º C for 48 hours and weighed for biomass. Within each quadrat, I collected two, 10 cm deep soil cores. I dried a known volume of soil at 60C for at least 48 hours, weighed, and combusted at 500º C for 4 hours to determine ash-free dry mass (AFDM); AFDM was determined on these cores as a proxy for Soil Organic Content (SOC). In total, I sampled and harvested 116 quadrats in 2003 and 126 in 2004.

Data analysis Measures of biodiversity (alpha, beta, and gamma diversity) were calculated for each site/year combination. Alpha diversity, in its most basic form is the number of species per unit area (Whittaker 1972); in this case, alpha diversity was number of species per point along the transect (0.25m^2). Beta diversity represents the change in species, or simply the heterogeneity along a gradient (Whittaker 1972). Gamma diversity estimates diversity at the landscape level, and represents the total number of species across the site (Whittaker 1972). Beta diversity is calculated as gamma over alpha (McCune et al. 1997).

Frequency of occurrence of each species was calculated at each site to give an idea of the plant communities along the peninsula. Species dominance was calculated by taking each species biomass and calculating the percentage of its contribution to the overall biomass.

Analysis of variance (ANOVA) (SPSS 9.0 was statistical package used unless otherwise noted) was used to determine if live aboveground biomass values were

different at the three sites and between years. Soil Organic Content (SOC) values were analyzed using ANOVA to determine differences between years and sites.

Non-parametric analyses (Kruskal Wallis) were used to determine if *L. carolinianum* abundance and/or biomass was significantly different between the three sites. Indirect gradient analysis (Correspondence Analysis, PC-ORD) was employed to see if the plant community was responding to an environmental or combination of variables and where *L. carolinianum* was placed along that gradient. Correlation (Spearman) was run with biomass of all species to determine which species were more or less likely to occur with *L. carolinianum.*

Territory delineation Bonds (2000) estimated the boundaries of whooping crane territories using aerial survey data of banded whooping cranes from winters 1993-93 through 1996-97. Using the cranes locations from these aerial surveys, Bonds (2000) delineated a total of 29 territories in the area on and around ANWR, including 9 on Blackjack Peninsula, for the 1996-97 winter. I selected three territories (data available in Bonds 2000) from this season; the three territories are referred to as: Boat Ramp, Mustang Slough, and Middle Sundown Island (Figure 3), which coincide with my Boat Ramp, Pump Canal, and Sundown Bay sites.   In ArcView (version 9.1), I overlaid these three territories onto a 1995 USGS mosaic digital orthophoto quad (DOQ). Areas of marsh, water, and bare ground were manually digitized (Figure 8). An estimate of total area in each habitat type was summarized, yielding area estimates of marsh, water, and bare ground in each territory.
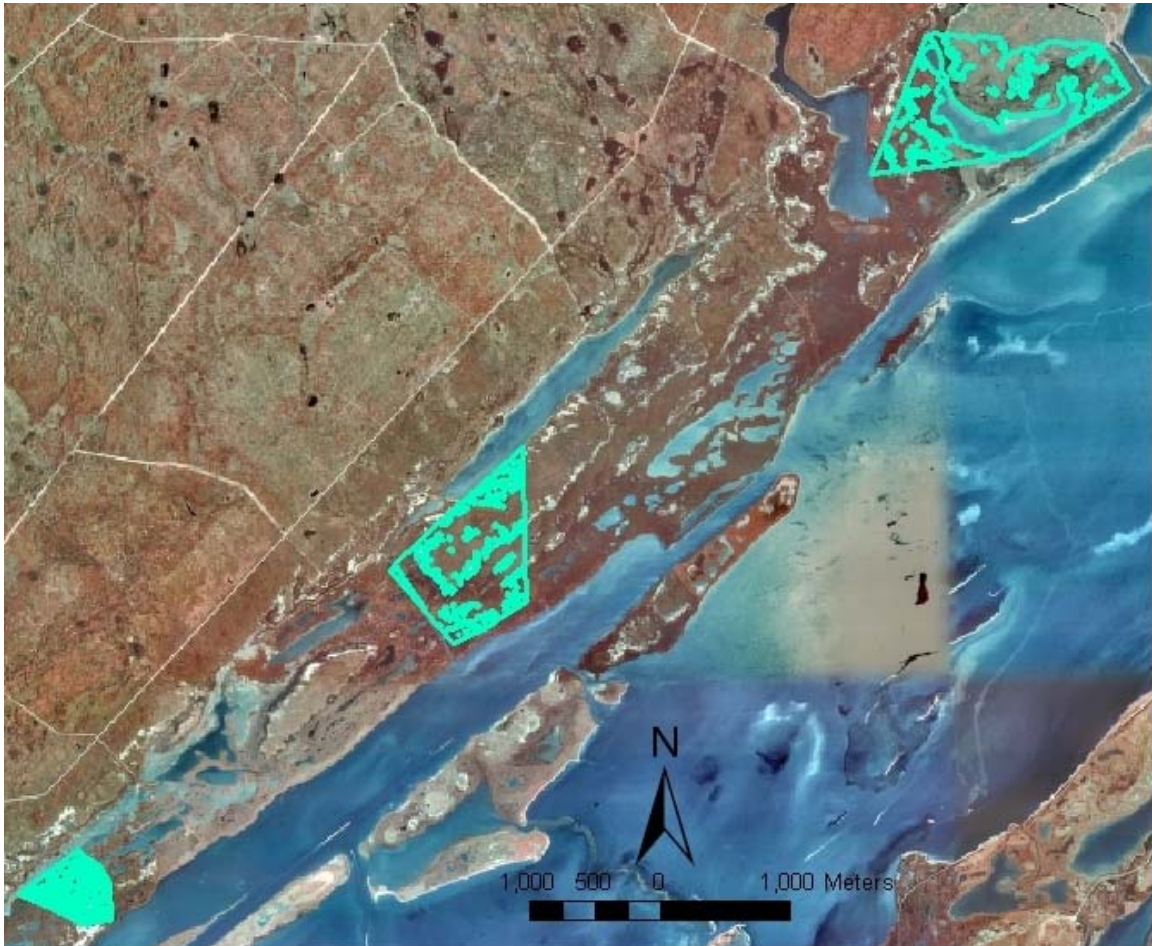
Figure 8: Outline and habitat type delineation of three territories encompassing vegetation sampling stations at Aransas National Wildlife Refuge.

The number of *L. carolinianum* (individuals • m^{-2}) from each site (from

transects) was then applied to the marsh area for an estimate of *L. carolinianum*

abundance on a marsh landscape. Data from Chapter III were applied to estimate berry

abundance on the territory scale.

$$\frac{\#\ \textit{Lycium carolinianum}\ \text{plants}}{m^2}\ \times\ \frac{10000m^2}{ha}\ \times\ \frac{\text{marsh area (ha)}}{\text{territory}}\ \times\ \frac{\#\text{berries}}{\text{plant}}\ =\ \frac{\text{berries}}{\text{territory}}$$

RESULTS

A USGS gauge station along the lower Guadalupe River (at Victoria) measured

inflows into the estuary (Figure 9). Throughout the study, there was a seasonal gradient

in surface water salinity from the Boat Ramp site to the Sundown Bay site (Figure 10).

Overall, surface water salinity was lowest and most variable at the Boat Ramp site

(Figure 10). While 2003 began as a wet year, by April, the flow was less than the 68-

year average (from 1934-2002). During 2004, inflows from the river were considerably

higher with a 4 months of average flow conditions and 8 months of above average flow.

In addition, above average precipitation in the fall of 2004 resulted in record discharge in

the Guadalupe River for the month of November (Figure 9) which resulted in extended

periods of low salinity throughout the estuary.
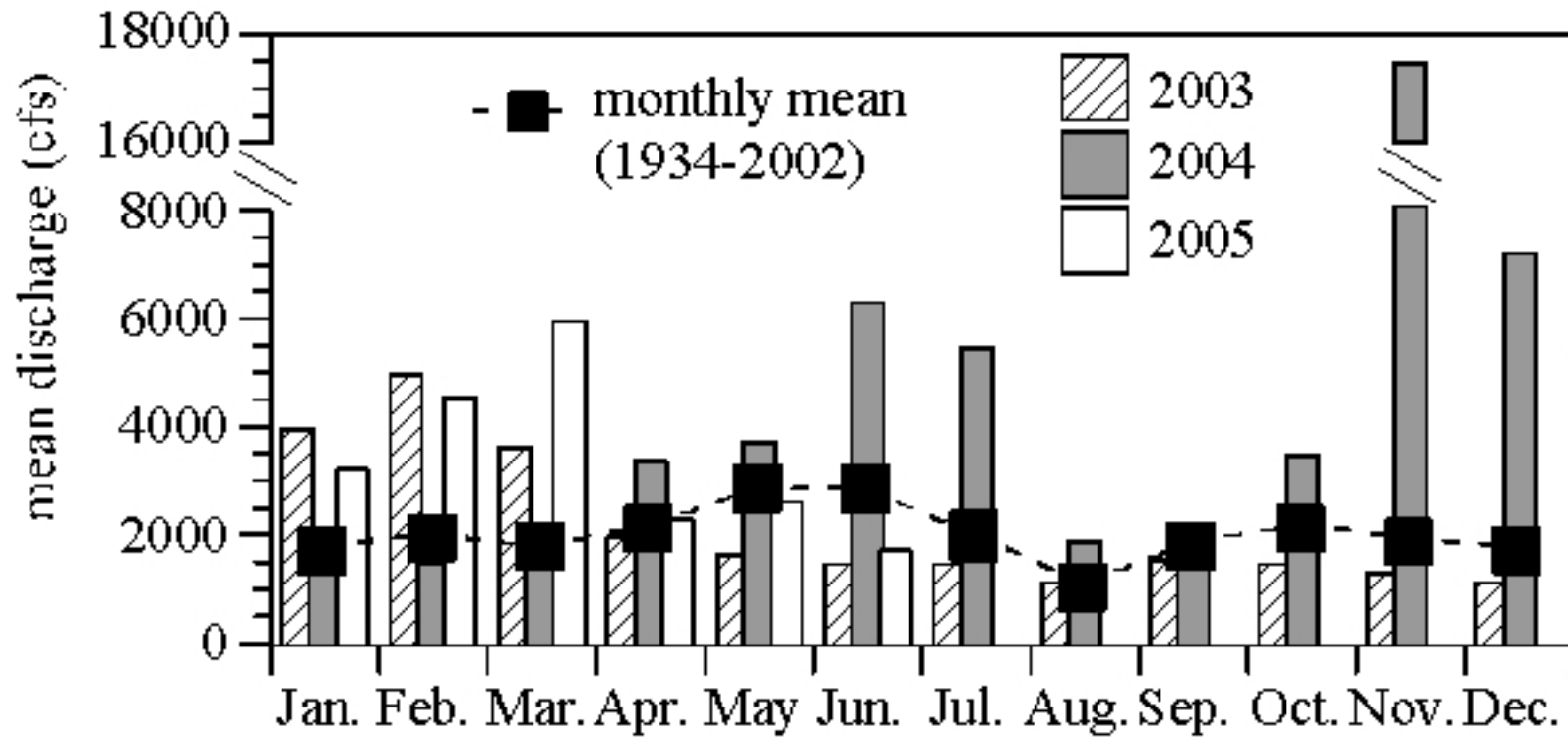
Figure 9: USGS data from the Guadalupe River gauge station at Victoria, TX, of the monthly values of flow (cfs) during the study and the 68 year mean.
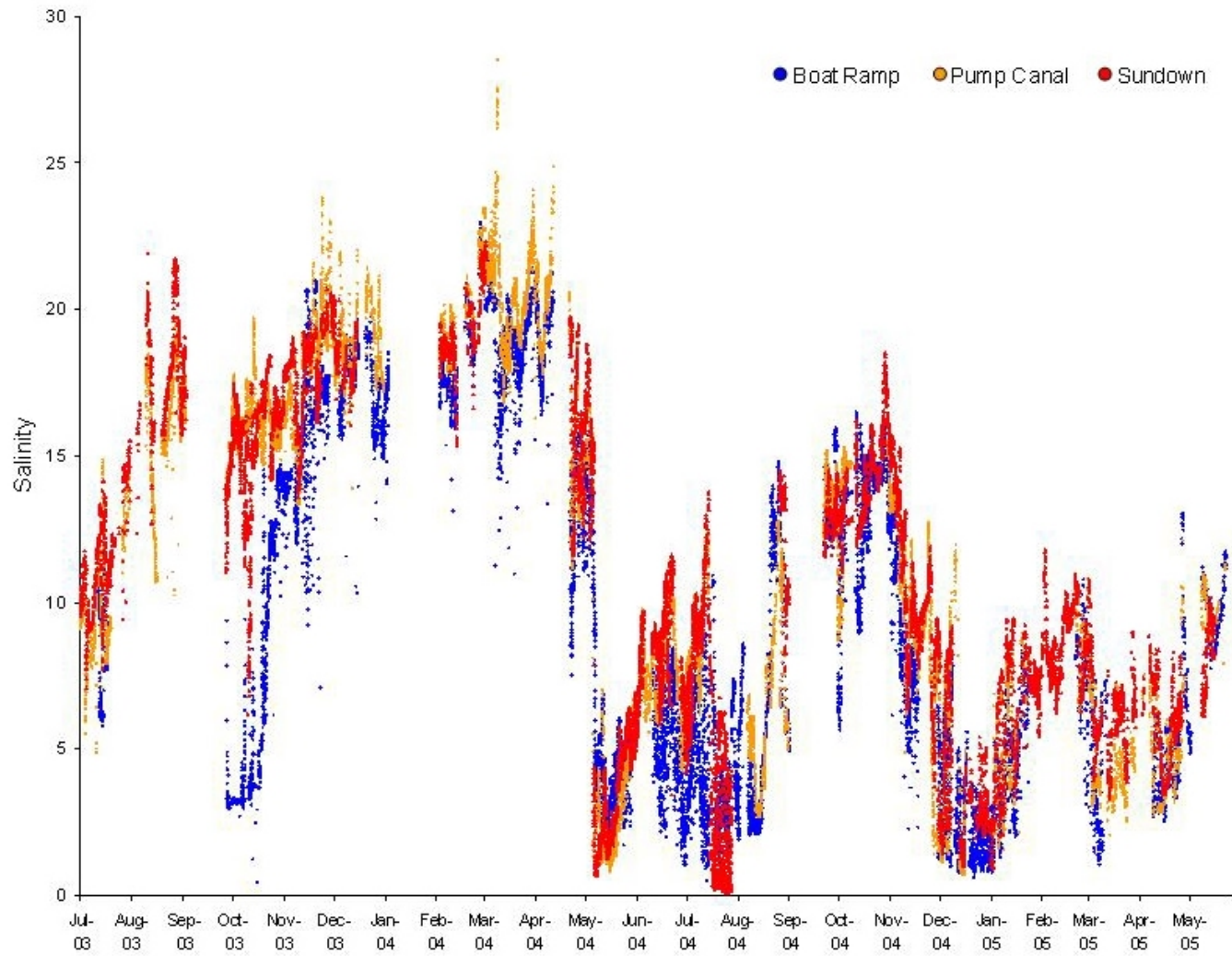
Figure 10: Salinity (ppt) data measured at half hour intervals at each water quality station using Hydrolab mini-sondes.

Vegetation sampling ANWR salt marshes are infrequently inundated—usually during spring tides, periods of high freshwater inflow, and storm events. All species names are as listed in Stutzenbaker (1999). At ANWR, there is usually a narrow fringe (a few meters in width) of *S. alterniflora* at the interface of the marsh and open water. *S. alterniflora* gives way to a patchy, mixed marsh community that includes *Aster tenuifolius, Batis maritima*, *Borrichia frutescens*, *Distichlis spicata*, *Lycium carolinianum, Monanthochloa littoralis, Salicornia bigelovii,* and *Salicornia virginica.* This mixed marsh community (approximately 0.5-1.0 km in width) grades into an upland transition zone dominated by *Spartina spartinae* and other grass and sedge species. The length to upland species at my three sites was variable, ranging from 350m to 650m (as shown in Figures 6, 7, and 9).

Mean species richness (alpha diversity) of the macrophyte community at all three sites was similar, ranging from 2.9 species per plot to 3.7 (Table 1). Gamma diversity was consistently lowest at Sundown Bay with only eight species present both years. Boat Ramp had the highest gamma diversity with 13 species in 2004, up two species from 2003. Beta diversity—a measure of heterogeneity between plots—was lowest at Sundown Bay in both 2003 and 2004.

A total of 13 species were sampled in 2003 and 15 in 2004 (Table 2). *Scirpus robustus* and *Haplopappus phyllocephalus* were only sampled in 2004. While the seven most commonly occurring species (the first 7 species in Table 2) were found at all three sites, there were some differences in species occurrences in the vegetation communities at the three sites. *M. littoralis* was the most frequently found species at Boat Ramp in

Table 1: Alpha (no./0.25m^2), Beta, and Gamma diversity in salt marsh macrophyte communities at the three ANWR sampling sites in 2003 and 2004.

	2003				2004			
	n	alpha (s.d.)	beta	gamma	n	alpha (s.d.)	beta	gamma
Boat Ramp	32	2.9 (1.21)	3.7	11	42	3.6 (1.38)	3.7	13
Pump Canal	54	3.5 (1.61)	3.2	11	52	3.4 (1.54)	3.5	12
Sundown Bay	30	3.5 (1.91)	2.3	8	32	3.7 (1.62)	2.2	8

Table 2: Frequency of each salt marsh macrophyte species found along transects sampled in Fall 2003 and 2004.

	2003			2004		
	Boat Ramp	Pump Canal	Sundown Bay	Boat Ramp	Pump Canal	Sundown Bay
	n=32	n=54	n=30	n=42	n=52	n=32
Aster tenuifolius L.	0.09	0.07	0.33	0.14	0.19	0.28
Batis maritima L.	0.53	0.76	0.67	0.57	0.73	0.75
Borrichia frutescens (L.) DC.	0.28	0.24	0.47	0.36	0.27	0.47
Distichlis spicata (L.) Greene	0.44	0.61	0.7	0.55	0.56	0.69
Lycium carolinianum Walt.	0.28	0.39	0.37	0.29	0.35	0.28
Monanthochloa littoralis Engelm.	0.63	0.39	0.07	0.52	0.37	0.09
Salicornia spp.	0.44	0.69	0.7	0.69	0.73	0.72
Spartina alterniflora Loisel.	0.06	0.22	0.23	0	0.12	0.38
Haplopappus phyllocephalus DC.	0	0	0	0.05	0.02	0
Limonium carolinianum (Walt.) Britt.	0.03	0	0	0.05	0	0
Scirpus robustus Pursh	0	0	0	0	0.02	0
Sesuvium maritimum (Walt.) B.S.P.	0	0.02	0	0.05	0	0
Suaeda linearis (Ell.) Moq.	0.06	0.06	0	0.12	0.04	0
Sunflower	0.03	0	0	0.1	0.02	0
unknown 1	0	0.02	0	0.07	0	0

2003 and the least commonly found species at Sundown Bay in 2003 and 2004. *S. alterniflora* was found in only 6, 22, and 23 % of the quadrats in 2003 at Boat Ramp, Pump Canal, and Sundown Bay, and 0%, 12%, and 38% in 2004. *B. maritima, Salicornia spp,* and *D. spicata* were the most common species found in at least 40% of quadrats at all sites in both years.

The frequency of occurrence of species did not display much inter-annual variability. The greatest difference between the two years was with *Salicornia sp.* It was found in 44% of the quadrats at Boat Ramp in 2003 opposed to 69% in 2004. Also, *S. alterniflora* was absent in the 2004 Boat Ramp transects. *Lycium carolinianum* was found in 28% (Boat Ramp 2003 and Sundown Bay 2004) to 39% (Pump Canal 2003) of quadrats.

Kruskal-Wallis independent samples test did not detect significant differences in the number of *L. carolinianum* individuals sampled (Table 3) or biomass along the transects between the three sites in either of the study years. When berry abundance values (from Chapter III) are applied to the *L. carolinianum* plants in the transects to estimate number of berries along the transect, there are significant differences in the number of berries between the three sites in March, April, June, and July of 2004 (p=0.000), and September 2004 and January 2005 (p=0.003).

Species dominance was calculated each year for the three sites individually, and also by combining Boat Ramp, Pump Canal, and Sundown Bay for an overall dominance (Figure 11). There are very few cases where one species dominated over the other

Table 3: Mean number of *L. carolinianum* plants per m^2 (+/- SD) sampled along the transects.

	2003	2004
Boat Ramp	9.5 (21.9)	8.6 (22.2)
Pump Canal	15.6 (27.7)	4.1 (7.2)
Sundown Bay	15.5 (26.9)	4.1 (9.6)

2003   2004

Boat
Ramp

Pump
Canal

Sundown
Bay

All
Sites

	A. tenuifolius
	B. maritima
	B. frutescens
	D. spicata
	L. carolinianum
	M. littoralis
	Salicornia spp.
	S.alterniflora
	S. linearis
	other
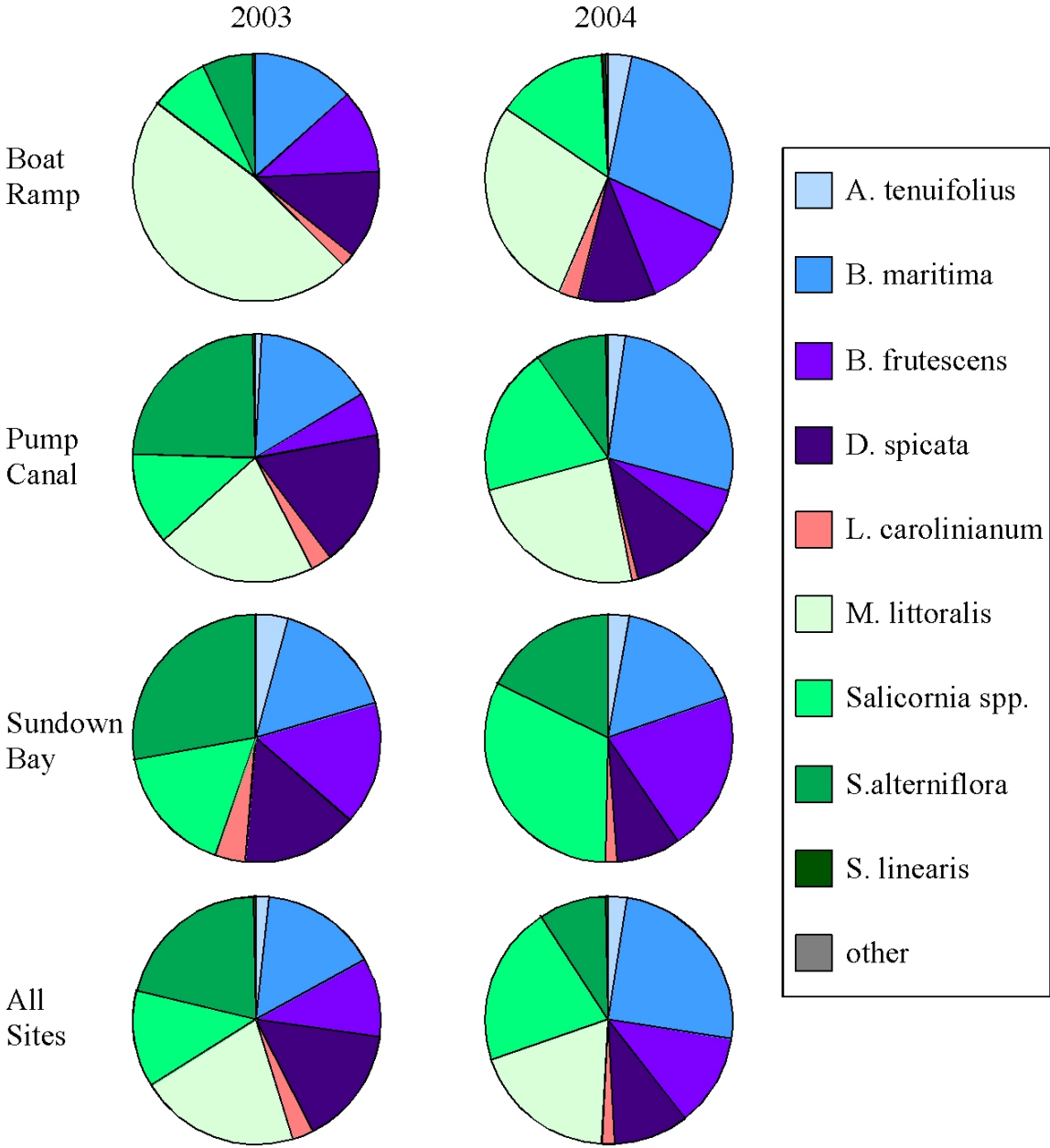
Figure 11: Species dominance, on a dry weight basis, by year and site.

species. In 2003 at Boat Ramp, *M, littoralis* was the most dominant species. At Sundown Bay, *S. alterniflora* accounted for over a quarter of the total biomass harvested, and in 2004 *Salicornia spp.* was the more dominant species. For the remainder of the sites, and overall, there are a combination of species that dominate; *B. maritima, M. littoralis, Salicornia spp.,* and *S. alterniflora* are the species that account for at least 25% of total biomass.

Live aboveground biomass values ranged from 122 g•m^{-2} to 2597.8 g•m^{-2} in 2003 and 79.44 g•m^{-2} to 1996.28 g•m^{-2} in 2004 (Figure 12). Biomass values between the three sites are not significantly different in either 2003 or 2004, or between years at the 0.05 level (ANOVA).

Percent of total biomass of each species was calculated (Figure 13) at each sampling point along a given transect. Rare (*S. robustus, H. phyllocephalus,* and *Limonium carolinianum*) and unknown species biomasses were combined as these did not contribute significantly to the overall biomass. At Boat Ramp, *M. littoralis* contributed more to the overall biomass in 2003, whereas in 2004, *B. maritima* and *Salicornia spp.* contributed more to the overall biomass. At Pump Canal, *B. maritima*'s contribution to overall biomass increased from 2003 to 2004. In Sundown Bay, *Salicornia spp.* contributed more biomass than in 2003 *S. alterniflora* dominated the shoreline at Pump Canal and Sundown Bay both years, but not at Boat Ramp. *S. alterniflora,* when present, was usually found at the marsh edge (0m). *L. carolinianum* contributes very little to the total biomass, compared to other species.
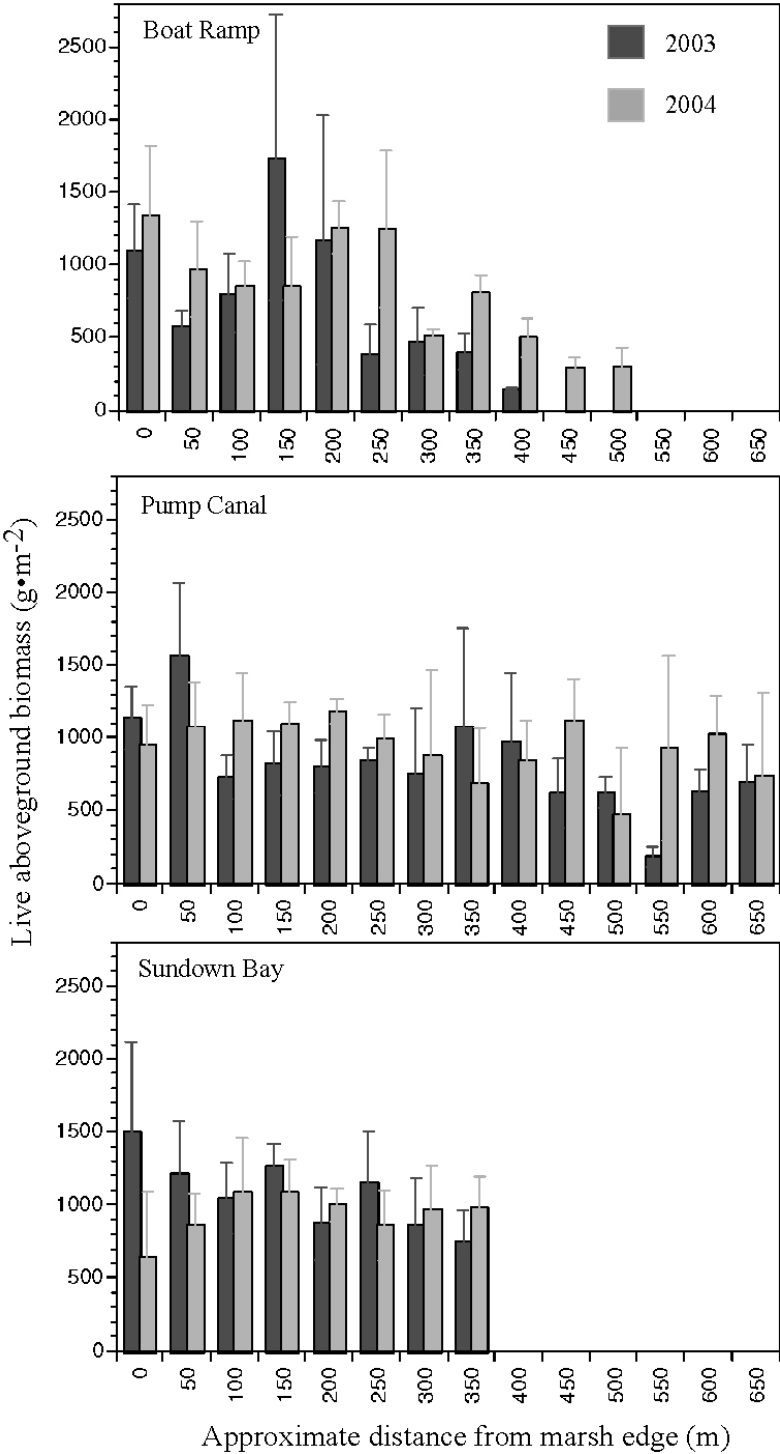
Figure 12: Mean total biomass values obtained at each site and the approximate distance from march edge where the vegetation was sampled. Data are expressed as dry weight aboveground biomass.
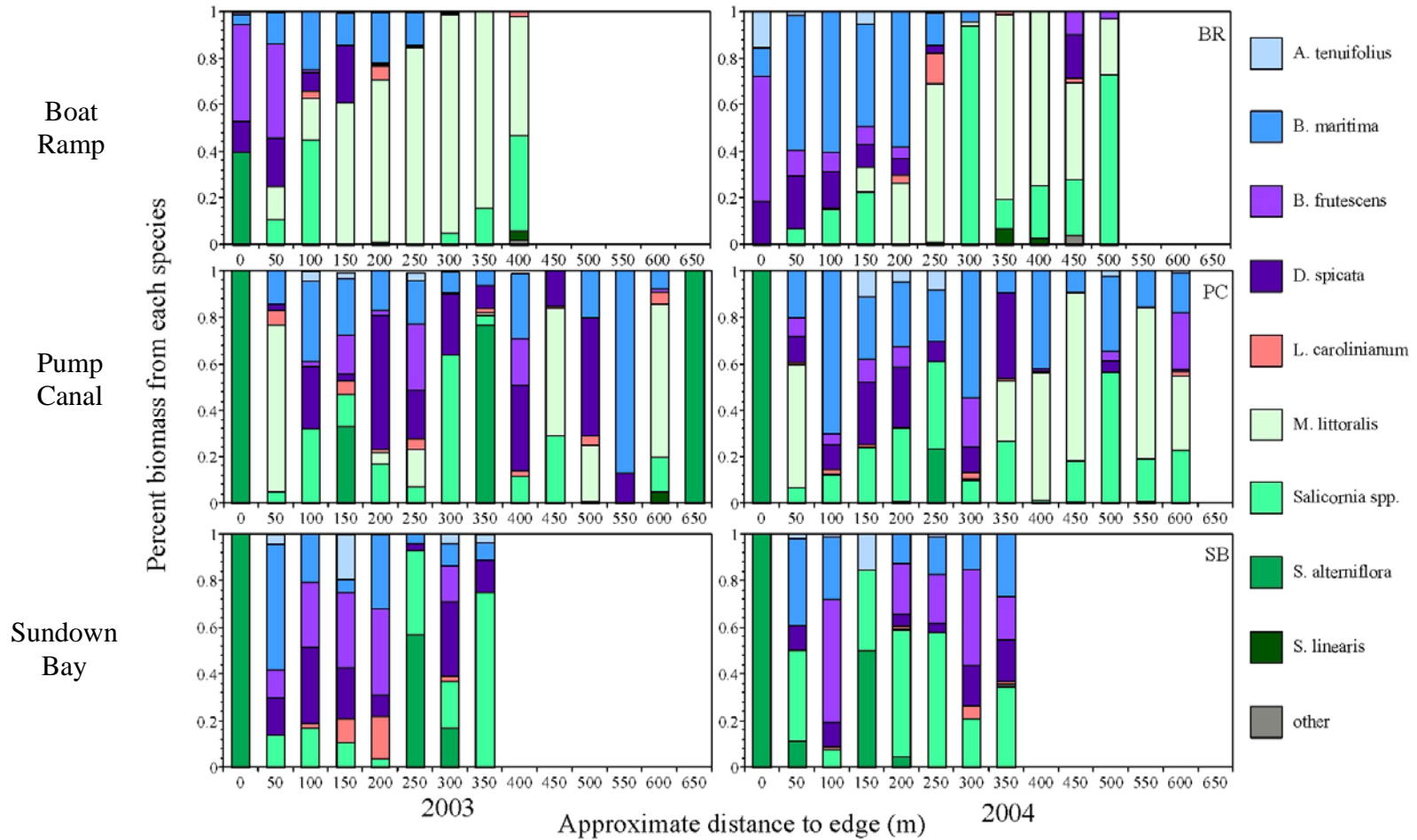
Figure 13: Percent biomass of each species present at the approximate distance from the edge. "Other" category includes *Limonium carolinianum, Happlopappus phyllocephalus, Sesuvium maritimum, Scirpus robustus,* sunflower, and the unknown species.

Correspondence analysis (CA, also known as RA – Reciprocal Averaging) was run on the species biomass to help detect the vegetation community's response to environmental gradients. CA is an indirect ordination technique that can graphically represent how species are responding to environmental gradients (Figure 14). In 2003, eigenvalues of the first and second axes were 0.846 and 0.489, and in 2004, 0.725 and 0.581. Since *S. alterniflora* dominated axis 1 so heavily in both 2003 and 2004, *S. alterniflora* biomass was removed, and CA ran again (Figure 15). In 2003, with no *S. alterniflora* biomass axis 1 was 0.491 and axis 2 was 0.222; in 2004, axes scores were 0.586 and 0.248.

Correlation analyses between biomass of all the species were used to further determine the relationship of *L. carolinianum* to other species for 2003 and 2004 separately (Table 4). In 2003, there was a positive correlation between *L. carolinianum* and *B. maritima*, *B. frutescens,* and *M. littoralis* and a negative correlation between *L. carolinianum* and *S. alterniflora*, all significant at 0.01. In 2004, again there was a positive correlation between the biomass of *L. carolinianum* and *B. maritima* and *B. frutescens* (alpha = 0.01). In 2004 a positive correlation was detected between *L. carolinianum* and *D. spicata* (alpha = 0.01), which during 2003 had an insignificant, yet negative correlation coefficient. *M. littoralis,* which had a positive and significant correlation coefficient with *L. carolinianum* in 2003 was positive, yet insignificant in 2004. *S. alterniflora* had another significant negative correlation with *L. carolinianum* in 2004, this one significant at alpha = 0.05).
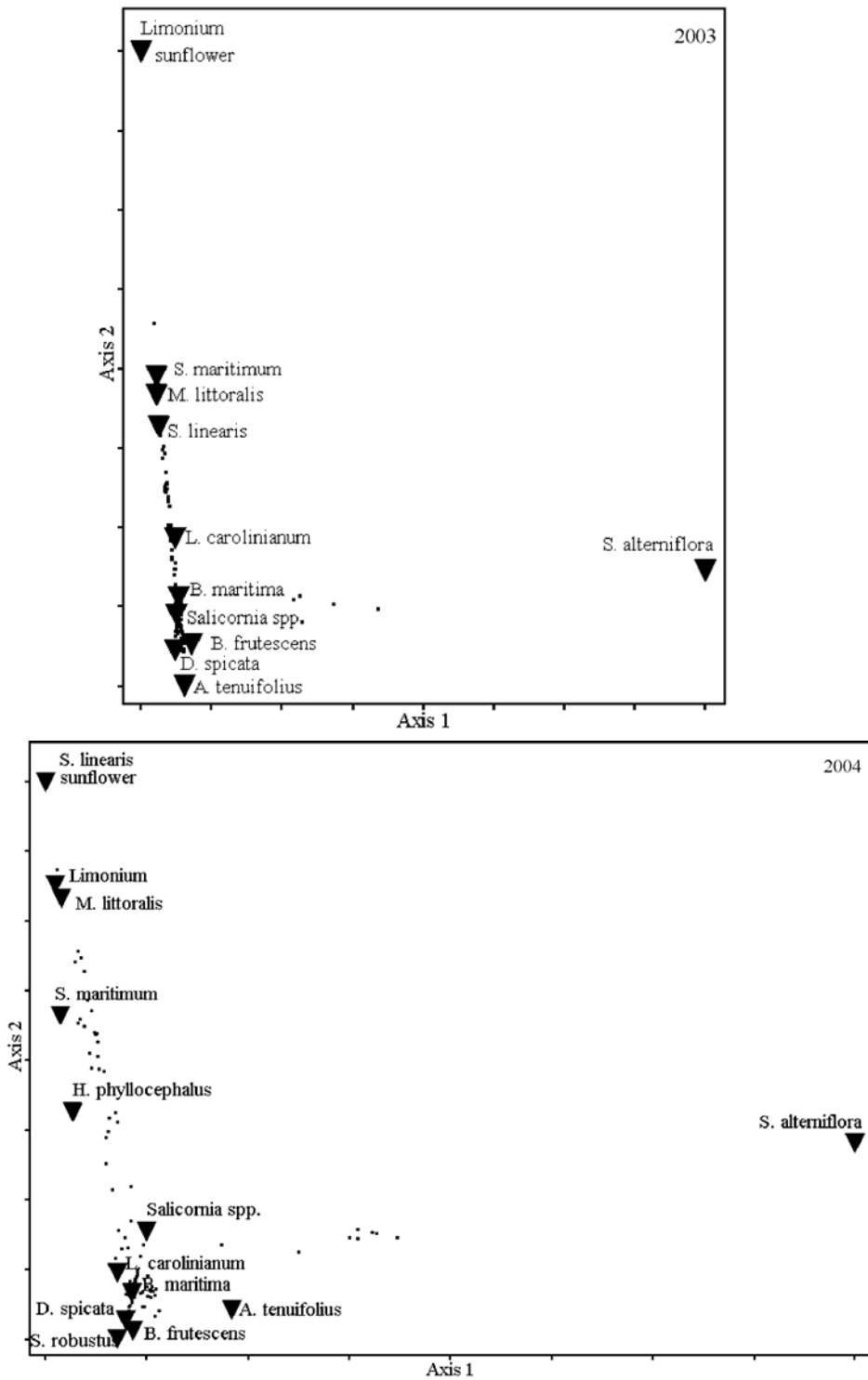
Figure 14: Correspondence Analysis (CA) of all species biomass (minus unknown) for 2003 and 2004 individually.
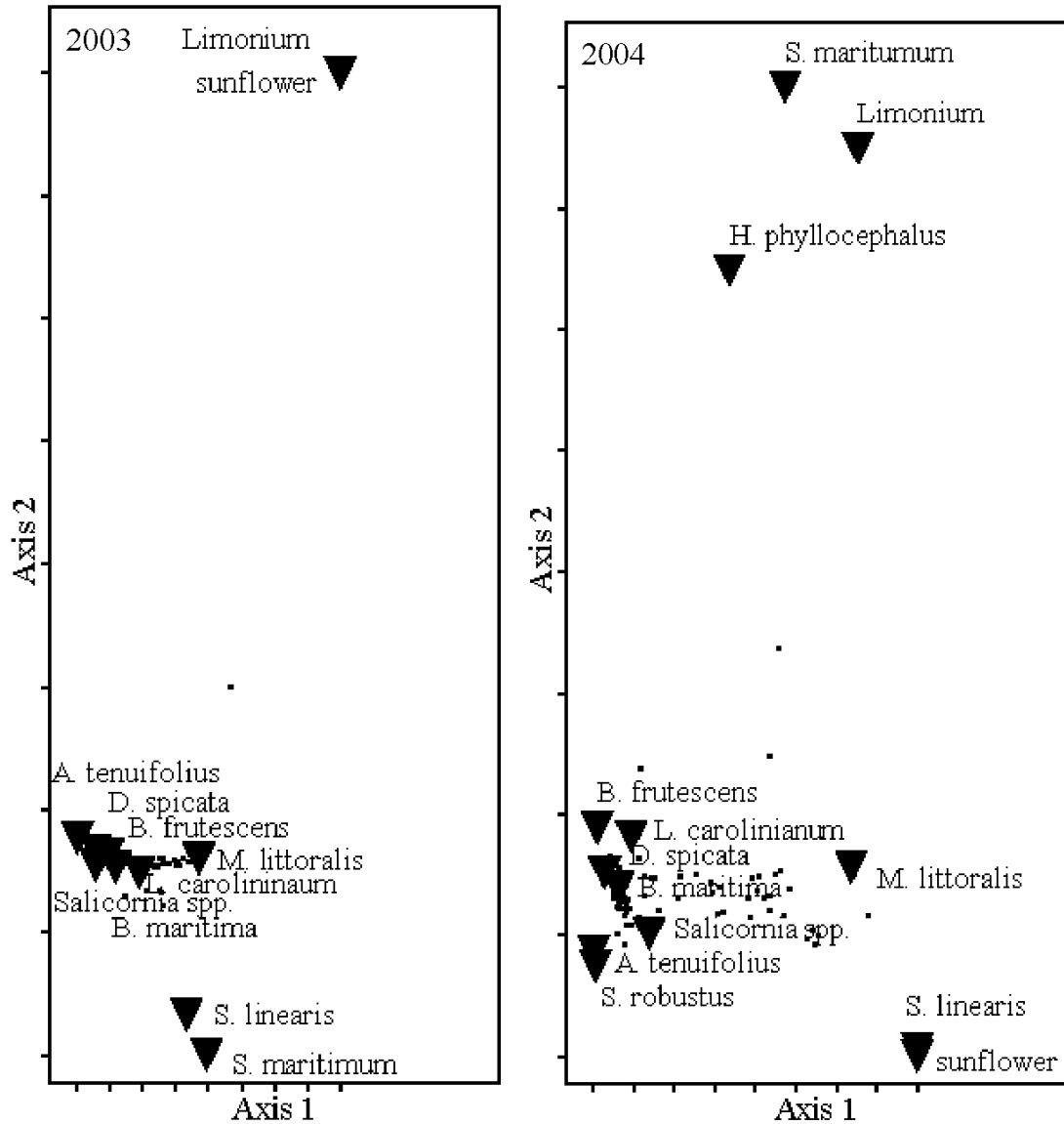
Figure 15: Correspondence Analysis (CA) of species biomass (minus unknown and *S. alterniflora*) for 2003 and 2004 individually.

Table 4: Spearman correlation coefficients of these species with *L. carolinianum*. Only species with significance are presented.

	2003	2004
B. maritima	0.341**	0.253**
B. frutescens	0.339**	0.324**
D. spicata	-0.028	0.21*
M. littoralis	0.371**	0.032
S. alterniflora	-0.244**	-0.214*

** - significant at 0.01
* - significant at 0.05

Soil organic content displayed some year-to-year variability (Figure 16). At all three sites the soil organic content increased from the water's edge inland. At Boat Ramp and Pump Canal, the SOC declined after it reached a peak, while Sundown Bay increased from the marsh edge. There are some erratic organic content values (Pump Canal, 2004, 600m), which are not unexpected in highly variable systems similar to the salt marsh ecosystems. Sundown Bay was significantly different than the other 2 sites in 2003 (ANOVA, alpha = 0.03), but there were no significant differences between sites in 2004. There were no significant differences when comparing 2003 to 2004 at each of the three sites.

Territory delineation and wolfberry abundance The percentage of the total area occupied by marsh (as opposed to ponds or bare ground) was 69.05%, 86.38%, and 59.05% at Boat Ramp, Pump Canal, and Sundown Bay, respectively. Total area in marsh habitat type ranged from 93.35 ha. at the Boat Ramp to 13.92 at Sundown Bay (Table 5). With the data available from this chapter (# *L. carolinianum* plants m^{-2}, territory size, and area of marsh within each territory) and the next chapter (monthly berry production), an approximate number of berries per territory was calculated (Figure 17).
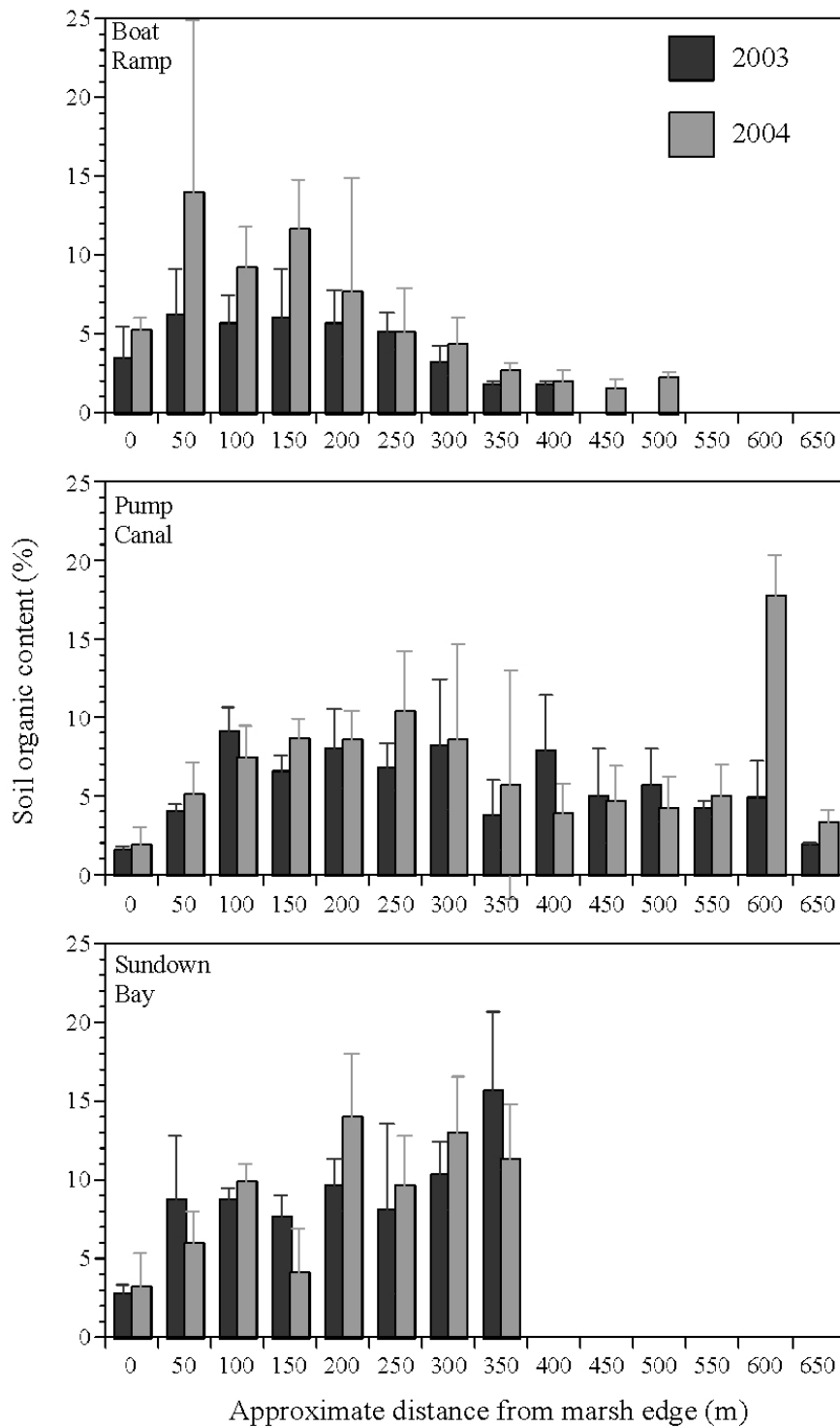
Figure 16: Mean soil organic carbon (SOC) values obtained at each site and the approximate distance from march edge where the soil was sampled. Data are expressed in percent soil organic carbon.

Table 5: Total area (ha) of each habitat type digitized in crane territory. Three territories selected (from Bonds 2000) were Boat Ramp (coinciding with my Boat Ramp site), Mustang Slough (coinciding with my Pump Canal site) and Middle Sundown Island (coinciding with my Sundown Bay site). Values in parentheses are percent area of that habitat type out of total area.

	marsh (ha.)	pond (ha.)	bare (ha.)	total(ha.)
Boat Ramp	93.35 (69.0%)	32.47 (24.0%)	9.38 (6.9%)	135.2
Pump Canal	75.8 (86.4%)	3.08 (3.5%)	8.87 (10.1%)	87.75
Sundown Bay	13.92 (59.1%)	9.26 (39.3%)	0.38 (1.6%)	23.57
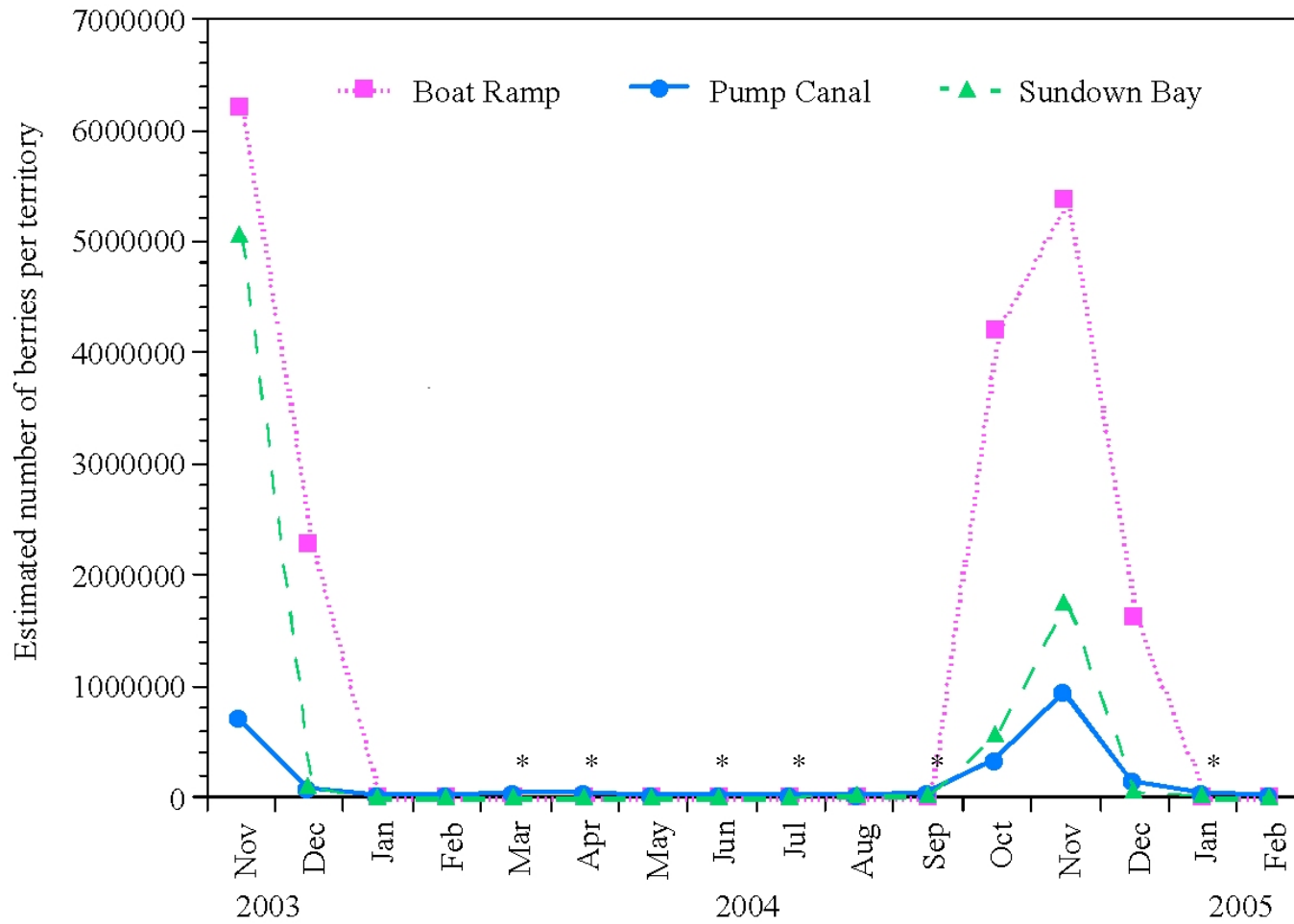
Figure 17: Estimated number of berries in the three territories.

DISCUSSION

Species composition, distribution, and abundance may shift due to hydrologic alterations. Van der Walk (1994) found that species abundance and distribution were affected in the first couple of years of increased water level, but more time (3+ years) is required for significant shifts in species composition to take place. As 2004 was a wetter year (i.e., increased precipitation, therefore increased river flow and upland runoff) than 2003, I expected to see differences in the vegetation community attributed to this increased freshwater inflow. There were minor changes in the total number of species found in the transects, with more species at Boat Ramp site in the wetter year. If this is not an artifact of sampling, it could be that the sites closest to the freshwater input are most sensitive to fluctuation in salinity. Dunton et al. (2001) studied the Nueces River Estuary, south of the Guadalupe Estuary, in a season that experienced high amounts of precipitation. While species composition and frequency of occurrence of the species along transects were relatively similar throughout the study, they found increases in cover and biomass of species less tolerant to salt, such as *B. frutescens*, and a decrease in *B. maritima* (Dunton et al. 2001). While major shifts in vegetation are not expected on a short time scale, it is quite common to see small shifts like increase in cover (Dunton et al. 2001) or peaks in biomass (Zedler 1983) when vegetation is experiences high variability causing changes in salinity.

Similarly, this study found that the species composition and frequency of occurrence were similar in 2003 and 2004. Because *S. alterniflora* is a major component of the vegetation community in many salt marshes (Mitsch and Gosselink 1993, Cronk

and Fennessey 2001), it is worth noting it was absent in the 2004 Boat Ramp transects. The salt marshes at ANWR simply supported a level of diversity more common to the high marsh zones than a *S. alterniflora* dominated zone. Also at Boat Ramp I observed a decrease in *M. littoralis,* and increases in *B. maritima* and *Salicornia spp.* contributions to the overall biomass (Figures 11 and 13). All of these species are halophytic, meaning they are successful in saline environments. *M. littoralis* does not appear to have the salt excretion ability that *S. alterniflora* possesses, nor the succulence of *B. maritima* and *Salicornia spp.* which would suggest that *M. littoralis* should have increased cover in the year with record rainfall. However, *M. littoralis* does not tolerate inundation as well as the other two species (Purer 1942), indicating that it could be a combination of salinity and inundation responsible for these small shifts in vegetation. My results were agreed with Alexander and Dunton (2002) which found increased cover in *Salicornia bigelovii* and a temporary increase in *B. maritima* with increased precipitation.

Dunton et al. (2001) found that in wet years *L. carolinianum* stems were more likely to have leaves than just bare stems. I found similar frequencies of *Lycium carolinianum* in both years (Table 2), but lower densities in the second year (Table 3); this may have occurred because I harvested only plants with leaves, and bare stems were not counted. Even though the second year was wetter, I did not detect the same trend of more stems with green tissue than bare stems as was found in the Nueces Estuary. Time of the year the sampling occurred may account for these differences - as the next chapter illustrates the distinct leafing patterns of *L. carolinianum.*

Correspondence analysis (CA), an indirect ordination technique, is often misused (McCune and Grace 2002). It can be considered useful to display species space and sample unit space; additionally, when beta diversity is low, the differences between CA and other ordination techniques are minimal (McCune and Grace 2002). The CA values for Axis 1 in 2003 and 2004 were fairly high, indicating that the first environmental gradient was rather strong (Figure 14). It seemed that *S. alterniflora* dominated this gradient which I assumed was elevation; field observations have indicated that the elevation range of *S. alterniflora* was significantly lower than that of the remaining "high marsh species". In fact, *S. alterniflora* was found on the edges of canals and ponds, and inundated longer and more frequently than the remaining species.

The placement of *L. carolinianum* on Axis 1 was similar to most other high marsh species. While the spread of Axis 1 was explained by large scale elevation change, the score for Axis 2 may be explained by the micro-topographical changes (Figure 14). The marsh at ANWR is very heterogeneous, dotted with numerous pools, ponds, and slightly elevated areas. These small changes may be very important to species like *L. carolinianum* that did not dominate entire stands, but rather survived in these transitional microhabitats.

Since *S. alterniflora* clearly dominated axis 1, it was removed from the CA to allow for a closer look at this "microtopography" effect. In the second CA (minus *S. alterniflora)*, *A. tenuifolius, B. maritima, B. frutescens, D. spicata L. carolinianum,* and *Salicornia spp.* were grouped together (Figure 15). *M. littoralis*, another common marsh species, was not in this close gathering. *M. littoralis* was one species that exhibited high

site-to-site variability; it was a major part of the community at Boat Ramp and Pump Canal in both years, but was not detected in the transects at Sundown Bay (figures 11 and 13). The lack of this plant in Sundown Bay may help account for its separation from the other common marsh species in the second CA (Figure 15). The less common species were scattered farther away from the common species – perhaps indicative in soil properties or other environmental characteristics not studied here.

Correlation demonstrated that the biomass of *B. maritima, B. frutescens*, (and depending on which year of the study for *D. spicata* and *M. littoralis*) had a positive correlation with biomass of *L. carolinianum* (Table 4). This indicated that there was a relationship between the presence of these species and *L. carolinianum,* and that when one was found there was a greater chance of the other being there than with other species harvested along the transects. However, while *B. maritime, B. frutescens* and *L. carolinianum* were found in association with each other, *B. maritima* was found more frequently than both *B. frutescens* and *L. carolinianum* (Table 2), and overall was more dominant than *B. frutescens* and *L. carolinianum (*Figure 11). These differences (one "common species" more dominant than other fairly common species) can be due to natural variability in the marsh possibly related to soil properties, minor elevational differences, and other ecological parameters that help determine plant distribution in the marsh.

Aboveground biomass values from Dunton et al. (2001) ranged from 1800 g dw m^{-2} to over 6500 g dw m^{-2}, which were much greater then values obtained from my harvesting efforts (Figure 12). An explanation for this large difference might be that

Dunton et al. (2001) only measured peak biomass for monotypic stands of vegetation. While I detected monotypic stands in the ANWR marshes, there was a mixed marsh community of around three species in most of the quadrats sampled (Table 1).

The analysis of soil organic content SOC showed significant differences between Sundown Bay and the other two sites in one year. These data have high SOC value contents at one site along the transects, and as the sample size at Sundown Bay was smaller than the other sites, it was more affected by these high values than the other sites (Figure 16). Salt marshes are highly variable ecosystems, and the vegetation communities associated with these high SOC values are representative of the marsh (i.e., these species have been found throughout the marsh) and in numbers that are also similar to other quadrats harvested. The high SOC values at Sundown Bay were located on a tidal creek near the back of the marsh, so they may have picked up an area with rich soil, perhaps due to highly variable inundations and nutrient deposition in the tidal creek.

Berry abundance data were not significantly different between the three sites when traveling along the transects (straight lines from the marsh edge to upland); however, total territory area and territory habitat composition affect food availability. Territory area (Table 5) was quite different between the sites; Boat Ramp was almost 6 times greater in area than Sundown Bay, and Pump Canal almost 4 times greater in area than Sundown Bay. Composition of the territory also was highly variable; of my three sites, Sundown Bay was the smallest of the territories, with 39% of its total area in water habitat; Pump Canal, which is the intermediate sized territory, has only 3.5% of its total area in water habitat, and Boat Ramp, the largest of the three, is 24% water.

When berry production values (from Chapter III) were applied to the number of *L. carolinianum* plants m^{-2} (estimated from transects) and then further multiplies by total marsh area in each territory, I was able to estimate berry abundances on the territory scale (Figure 17). While berry abundance data were not significantly different, there were significant differences between berry abundances on the territory scale. However, these differences were not present during the months of high berry occurrence (October, November, and December) or while the cranes were at ANWR, but in the months when berry production is typically negligible (March, April, June, July, and September 2004, and January 2005). These "significant differences" were not significant in regards to the whooping cranes; they are significant only because one site produced a few berries while the others produced none.

Also, Bonds (2000) found that the crane territories on ANWR had a mean 59.5% of overall area classified as salt marsh, which was at the lower range of my estimation (Table 5). As a result, *Lycium carolinianum* plant abundance estimates in the territories may be over estimated, as well as my estimation of berry abundance. However, this is the first step to a valuable tool in understanding crane ecology and their habitat and territory preferences. As expected, the largest of the territories contained the greatest estimated number of berries. The smallest, Sundown Bay, had the second greatest number of berries, followed by the intermediate sized territory with the least amount of berries. Taking this information into consideration, territory size could be a function of resource availability (a combination of wolf berry and blue crab), which is also tied to landscape metrics such as pond area, marsh-open water interface (i.e., edge), etc. Data

from this project combined with crab abundance data, and the cranes spatial use of the territory will help clarify how and why the cranes choose their territories, and guide management decisions that will help rather than hinder the success of the whooping crane population.

This research has increased the knowledge of the vegetation communities at ANWR. Species composition was similar at the three sites, with all sites sharing 6-7 common species. While the diversity is similar at the sites along the peninsula, those sites closer to freshwater input had more species, and may be more sensitive to salinity fluctuations. The marshes at ANWR are made up of distinct "low marsh" and "high marsh" zones; while *S. alterniflora* was only a minor part of the overall community, when found, it was dominating the "low marsh zones" in the landscape. Species composition exhibited very little variability from year one to year two of the study. Biomass values were not significantly different between sites or years. Transect sampling showed densities and biomass of *L. carolinianum* were not significantly different between sites or years either. *L. carolinianum,* while important to salt marsh ecology, accounted for only a small portion of the overall productivity (live aboveground biomass). *B. maritima, B. frutescens M. littoralis,* and in the second year, *D. spicata* were species more likely to be found with *L. carolinianum. S. alterniflora* had a negative correlation coefficient, and was less likely to be found with *L. carolinianum.* Based in these correlations, *L. carolinianum* is a found in association with some of the common species, indicating that its growth and survival requirements are typical to the salt marshes at ANWR.

This study encompassed two distinct years of freshwater input; the first year of the study was a relatively dry year with year two having record breaking river flow. However, I saw no detectable patterns in the distribution patterns of *L. carolinianum* – at least on the time scale in which I sampled. Future studies that incorporate more sites along the peninsula, and sample at a more intensive scale, may be able to discern differences in patterns along the salinity gradient.

Large shifts in the vegetation community may require events on a scale that exceeds their natural threshold (i.e., longer duration of freshwater inundation) (Zedler 1983). This study, as evidenced by similar vegetation frequencies from year to year, just illustrates how the estuarine systems can continue functioning despite high year-to-year variability. However, with pressures on estuaries nationwide, further studies looking at the effects of changes in water quality and quantity on the vegetation community will help to understand the Guadalupe Estuary better, and may be applicable in other similar systems.

CHAPTER III

GROWTH PATTERNS OF *Lycium carolinianum* Walt., THE CAROLINA

WOLFBERRY, IN THE SALT MARSHES OF

ARANSAS NATIONAL WILDLIFE REFUGE, TEXAS

SYNOPSIS

Understanding the implications of a proposed freshwater diversion from the

Guadalupe Estuary is needed because the wetlands in this system support the endangered

whooping crane (*Grus americana*). My marsh plant research and monitoring efforts

targeted the salt marshes at Aransas National Wildlife Refuge (ANWR) utilized by the

cranes each winter. Past research indicates that Carolina wolfberry (*Lycium

carolinianum)* contributes 21-52% of crane energy intake early in the wintering period

(Chavez 1996). Beginning in Fall 2003, I repeatedly sampled *L. carolinianum* in nine

$1m^2$ permanent macrophyte plots along the estuary salinity gradient. *L. carolinianum*

exhibits strong temporal patterns. Leaf production peaked in early spring and again just

prior to peak berry abundance. Flowering of *L. carolinianum* occurred in October and

November. Peak berry abundance coincided with the cranes arrival in late October and

early November. Berry production occurred in October, November, and December;

berries were virtually non-existent in the marshes for the remainder of the year.

Stepwise regression showed stem diameter alone proved to be a good estimator of

aboveground biomass, accounting for 94% of the variability ($p<0.001$). Changes in

aboveground biomass follow no distinct patterns in the year of monitoring, perhaps due

to the woody stem of the plant. Spatial patterns in *L. carolinianum* were not explained

by water quality parameters alone; I suggest that soil properties may help to account for the spatial variability.

INTRODUCTION

Tidal salt marshes are commonly found in estuaries which are formed and maintained by a number of factors, including the balance of freshwater inflow and the tidal flushing of saline water (Odum 1988, Jassby et al. 1995, Baldwin et al. 2001, Kennish 2001). Because of an increased demand in freshwater for human consumption, agricultural purposes, and other uses, there has been a decrease in freshwater inputs into Texas estuaries (Dunton et al. 2001) and estuaries nationwide (Jassby et al. 1995). Reduced freshwater input, in conjunction with rising sea level and seasonal variability, may significantly affect estuarine salinity gradients and inundation patterns, especially in micro-tidal, lagoonal estuaries like the Guadalupe Estuary, located on the Gulf Coast of Texas (Figure 3).

Salinity, temperature, and dissolved oxygen vary seasonally and annually in estuarine systems, and can significantly affect the plant, animal, and planktonic communities (Odum 1988, Jassby et al. 1995, Gough and Grace 1998, Kennish 2001). Salinity, along with elevation, drainage, nutrient limitation, and sea level rise have been suggested to influence species distribution, growth patterns, and productivity of salt marsh plants (Smart and Barko 1980, Webb 1983, Pennings and Callaway 1992, David 1996, Dunton et al. 2001). Although many plant and animal species that inhabit estuaries are adapted to a wide range of salinity, there are limits to this range, and changes in the

spatial or temporal patterns of salinity may reduce or eliminate some species (Fitch and Armstrong 1982). Plants are particularly susceptible to these types of environmental changes because they are rooted.

Studies have shown that changes in salinity can affect the estuarine macrophyte community (Gough and Grace 1998), especially in the salinity transition zones (McKee and Mendelssohn 1989). These salinity effects on vegetation communities vary depending on species, the magnitude of salinity change, and the duration of change (McKee and Mendelssohn 1989, Howard and Mendelssohn 2000). Significant and extended changes in salinity levels can result in shifts in species composition, mass plant mortality (Kennish 2001), decreased species diversity (Gough and Grace 1998, Howard and Mendelssohn 2000) and a reduction in community biomass (Gough and Grace 1998).

Oftentimes, the effects of salinity are more pronounced when paired with other variables, such as water level (McKee and Mendelssohn 1989), elevation (Pennings and Callaway 1992) or soil properties (Silvestri 2005). Webb and Mendelssohn (1996) concluded that while increased salinity had no affect on plant growth, increasing water level did. The combined effect of both increased salinity and submergence was most detrimental to plant growth (Webb and Mendelssohn 1996). In a more recent study, Silvestri et al. (2005) concluded that salinity and elevation in combination with factors including soil properties created by tidal fluctuations determined plant zonation. As roots are exposed to the salt content of the soil, rather than the standing water, soil salinity plays an important role in plant survival, growth, and productivity (DeLaune et

al. 1987). Nestler (1977) found an inverse relationship between *Spartina alterniflora* growth and porewater salinity. Smart and Barko (1978) also found an inverse relationship of growth and porewater salinity in *S.alterniflora* and *Distichlis spicata*, assuming no nutrient limitation.

While increases in salinity can alter vegetation patterns, increased freshwater input can also affect plant communities. In the Nueces Estuary, south of the Guadalupe Estuary, Dunton et al. (2001) concluded the growing season of *Lycium carolinianum* extended two months longer than normal growing seasons with increased freshwater. The same study also found significant changes in plant biomass, percent cover, and aboveground:belowground biomass in several marsh plant species with higher than normal precipitation (Dunton et al. 2001).

In ecosystems that experience high inter- and intra-annual variability in its environmental conditions the ability to track changes in aboveground biomass is important. In estuarine systems tracking aboveground biomass allows you to make broader inferences on the impacts of changing water quality. Developing a non-destructive method to estimate biomass is a beneficial tool in understanding plant productivity under variable conditions, and has been done with several salt marsh species. Morris and Haskin (1990) used a non-destructive method based on prior harvesting and plant measurements, and total stem count to track *Spartina alterniflora*. They were able to track plots of *S. alterniflora* monthly for a period of 5 years, and relate changes in Aboveground Primary Productivity to Mean-Sea-Level-Rise (MSLR) and rainfall (annual) and porewater salinity (inter-annual) (Morris and Haskin 1990). Daoust

and Childers (1988) developed non-destructive methods using only a subset of the population for nine species of freshwater marsh plants (*Cladium jamaicense Crantz, Eleocharis cellulosa Torr., Sagittaria lancifolia L., Peltandra virginica (L.) Schott* and *Endl., Pontederia cordata L., Crinum americanum L., Hymenocallis palmeri S. Wats., Panicum hemitomon Schult.*, and *Paspalidium geminatum (Forsk.) Stapf.* Thursby et al. (2002) were able to develop non-destructive biomass estimation for two more salt-marsh species, *S. alterniflora* and *Phragmites australis,* relying on only 5 shoots per quadrat.

Lycium carolinianum is a member of the Solanaceae family that is commonly found along much of the Gulf of Mexico coast. It's plant structure is more complex than the some of the species that those models were developed with, so using a subset of a larger plot maybe too difficult at this point, especially with the lack of data concerning this plant. However, the use of non-destructive methods to estimate biomass is potentially beneficial to understanding the ANWR salt marsh ecology by reducing the harvest and processing time while regularly monitoring the same plants. Applying such a tool to *L. carolinianum* at ANWR would be efficient way to track changes in Net Aboveground Primary Productivity (NAPP) of this plant, and gain a better understanding of the growth patterns of this plant, an important food source of the endangered whooping crane (*Grus americana).*

The three objectives to this study were to 1. develop a non-destructive method to estimate and track changes in aboveground biomass of *L. carolinianum*; 2. determine growth patterns (including berry production) of *L. carolinianum* using permanent vegetation quadrats; and 3. relate growth patterns to environmental factors such as

salinity and water levels. The results of this study will significantly increase the general knowledge of this plant, and provide a better understanding of the growth patterns of this plant.

STUDY SITE

The Coastal Bend region of Texas includes numerous bays and estuaries that are ecologically and economically important. One such estuary, the Guadalupe Estuary, is fed by freshwater inflow from the Guadalupe and San Antonio Rivers (Figure 6). Aransas National Wildlife Refuge (ANWR) is located in this estuary and represents a major stretch of undeveloped land along the southwestern edge of the estuary. There are approximately 2,800 acres of salt marsh at ANWR that provide food and habitat to many terrestrial and estuarine organisms; most well known is the endangered whooping crane (*Grus americana)* that winter in the salt marshes of this part of the Texas coast.

The salt marshes at ANWR are irregularly inundated -- only in extreme high tides and wind driven events (McAlister and McAlister 1993). The marsh vegetation community is more diverse compared to *Spartina alterniflora* dominated salt marshes that typify much of the Gulf and SE Atlantic coastlines.   At ANWR, there is usually a narrow fringe (a 1-2 meters in width) of *S. alterniflora* at the interface of the marsh and open water. *S. alterniflora* gives way to a patchy, mixed marsh community that includes *Aster tenuifolius, Batis maritima*, *Borrichia frutescens*, *Distichlis spicata*, *Lycium carolinianum, Monanthochloa littoralis, Salicornia bigelovii, Salicornia virginica*, and *Sueada linearis* (Chapter II). This mixed marsh community (approximately 1 km in

width) grades into an upland transition zone dominated by *Spartina spartinae* and other

grass and sedge species.

While the cranes are on their wintering grounds, they rely on blue crabs

(*Callinectus sapidus*) and *Lycium carolinuanum* as significant food sources (Chavez

1996). In fact, *L. carolinianum* produces berries that account for 21-52% of the cranes

energy intake early in the wintering period (Chavez 1996). Chavez's (1996) data

indicate that the year-to-year availability of berries and blue crabs is variable. However,

there is a paucity of information on this plant in the primary literature, as there have been

only taxonomic descriptions and brief mentions in larger studies. The only insight to

berry production of *L. carolinianum* is that it is "tardily deciduous, and denuded very

quickly in early winter" by birds eating the ripe fruit (Godfrey and Wooten 1981).

METHODS

Three sites were sampled along the length of the Blackjack Peninsula at ANWR

for field monitoring of the marsh macrophyte community (Figure 6). Boat Ramp was

closest to riverine input, followed by Pump Canal, and Sundown Bay. These marsh sites

represented the range in salinity, elevation, and vegetation cover types that were found

along the Peninsula

Growth patterns of L. carolinianum Nine permanent vegetation plots (three per

site) were established to track temporal dynamics (i.e., growth patterns) of *L.*

carolinianum (Figure 7). Plots were $1m^2$ quadrats made of 1/2" ID PVC. The locations

of theses plots were selected based on the presence of *L. carolinianum* (ie – at least one

L. carolinianum was present) and their proximity to water quality stations. Plots were revisited monthly from November 2003 to February 2005. All plant species were identified, counted, and recorded monthly. Each *L. carolinianum* in the plot was tagged with a unique number, and morphologic characteristics (stem diameter at base of the plant, plant height, number of leaves, branches, open buds, closed buds, live flowers, dead flowers, mature berries, and premature berries) were measured monthly (Figure 18). The number of branches included both branching at the base of the plant and branching of the aerial portions of the branch. Woody growths off the main stem were counted as branches if they supported any leaves, flowers, or berries.

During each sampling, new *L. carolinianum* individuals were tagged and entered into the monthly monitoring. In February 2004 water quality measurements were added to the sampling protocol. When surface water was present, surface water salinity and water depth measurements were recorded. PVC wells were installed in each plot mid-way through the study (June 2004) to allow for monthly measurement of soil porewater salinity. These measurements, which would then be used to explain differences in the growth patterns along the estuarine salinity gradient, may help understand the implications of changes in water quality and levels.

Non-destructive estimation of biomass – model development A non-destructive method to estimate biomass of *L. carolinianum* was developed between April 2004 and February 2005. Between 10 and 15 *L. carolinianum* individuals were harvested from each site 4 times throughout one growing year. A total of 118

Figure 18: Tagged *L. carolinianum* in permanent macrophyte plot.   I measured stem diameter with calipers as part of the monthly sampling protocol.

L. carolinianum plants were sampled to generate this model. Ten morphologic

measurements were taken from each plant (same as those from permanent plots - stem

diameter, plant height, number of leaves, branches, open buds, closed buds, live flowers,

dead flowers, mature berries, and premature berries), and each individual was dried at

60C to obtain aboveground biomass. These phenometric measurements were entered

into a stepwise regression as independent variables to determine which variables best

explained patterns of aboveground biomass, the dependent variable.   Once this equation

was determined, ideally a smaller set of variables could be used to accurately estimate

the aboveground biomass of *L. carolinianum* individuals in each plot for each sampling.

Also, this equation could then be applied to the measurement from permanent

macrophyte plots to estimate aboveground biomass of *L. carolininaum* in each

permanent macrophyte plot.

RESULTS

This study was initiated this study in late 2003, following a period of relatively

low riverine inflow into the Guadalupe Estuary (Figure 9). However, most of 2004 and

early 2005 had above average inflows into the estuary. A few of these months saw

historic record (November 2004) or near-record (December 2004) inflows for this

estuarine system (Figure 9).

Water depth, salinity, and porewater salinity varied both spatially and temporally

(Figure 19). Water depth was consistently greater at Sundown Bay, with the greatest

mean water level 16.8cm. The greatest mean water level at Pump Canal was 10.5cm,
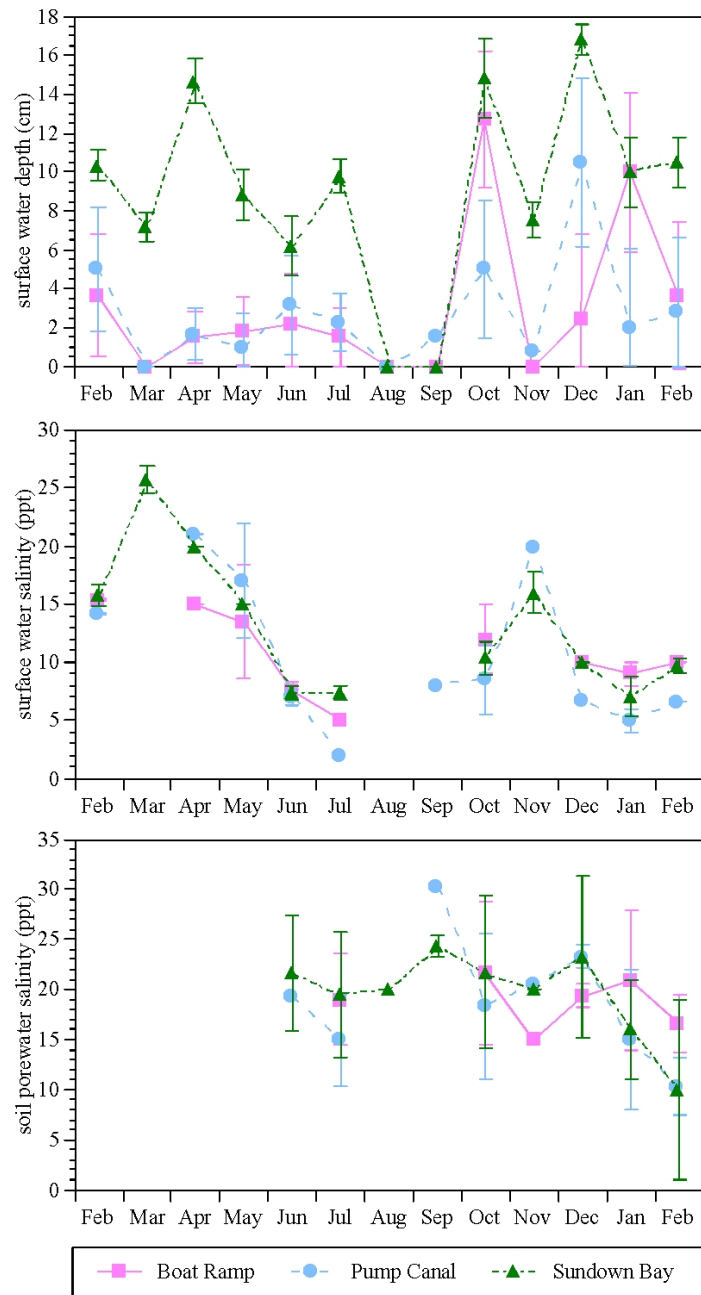
Figure 19: Water depth and salinity values taken during each monthly sampling from February 2004 to February 2005. Porewater sampling was initiated midway through the study, beginning June 2004. All values are mean +/- SD. When standing water was not present, water depth = 0, and there is no value for salinity. If porewater was not extracted, there is no value.

and Boat Ramp was 12.7cm. Pump Canal and Boat Ramp water levels were more similar to each other than Sundown Bay except in October, December, and January. All three sites experienced no standing water in all plots at least once during the sampling period. There were 4 samplings without standing water at the Boat Ramp site and only 2 samplings at both the Pump Canal and Sundown Bay sites.

Surface water salinity in the plots was also highly variable over time (Figure 19), with ranges following a gradient along Blackjack Peninsula. Monthly mean salinity ranged from 5-15.4 ‰ at Boat Ramp, 2-21 ‰ at Pump Canal, and 7-25.6 ‰ at Sundown Bay. Porewater salinity was only measured for part of the study, and was not always extractable. During the period of measurement in August and September, when the marsh surface was dry, I was unable to extract porewater from any of the plots at Boat Ramp, (August and September) and Pump Canal (August). Mean porewater values range from 15-21.7 ‰ at Boat Ramp, 10.3-30.3 ‰ at Pump Canal, and 10-24.3 ‰ at Sundown Bay (Figure 13).

The number of tagged *L. carolinianum* plants varied across plots (Table 6). This variability was mostly due to *L. carolinianum* plants dying and new plants germinating in the plots. However, there was also the inability to find every plant during each sampling. This could be due to deep water, leafless plants, and lost tags from either stem breakages, or zip-ties breaking. Boat Ramp 1 contained the greatest number of individuals (139 in February 2005), and Boat Ramp 2 and Sundown Bay 1 the least (13 in February 2005). Pump Canal and Sundown Bay plots had lower variation in *L. carolinianum* individuals compared to Boat Ramp. The number of tagged plants

Table 6: Number of *L. carolinianum* plants measured each month in the permanent macrophyte plots. Number listed is mean from the three plots (+/- SD).

		Boat Ramp (mean +/- SD)	Pump Canal (mean +/- SD)	Sundown Bay (mean +/- SD)
2003	Nov	30.67 +/- 19.60	26.67 +/- 3.21	22.00 +/- 12.63
	Dec	32.67 +/- 21.03	36.33 +/- 17.21	20.67 +/- 12.06
	Jan	34.00 +/- 22.52	41.33 +/- 15.95	23.00 +/- 14.53
	Feb	34.33 +/- 21.55	43.00 +/- 13.23	23.33 +/- 14.01
	Mar	50.00 +/- 37.51	55.00 +/- 21.93	30.33 +/- 20.01
	Apr	62.67 +/- 52.32	55.67 +/- 17.21	30.00 +/- 21.07
	May	71.67 +/- 64.30	56.33 +/- 17.62	28.33 +/- 19.22
2004	Jun	71.67 +/- 64.30	54.33 +/- 19.73	29.67 +/- 19.55
	Jul	72.67 +/- 65.29	53.67 +/- 17.01	27.67 +/- 18.01
	Aug	71.67 +/- 64.93	56.33 +/- 17.90	31.00 +/- 19.52
	Sep	71.33 +/- 65.38	56.67 +/- 22.94	31.33 +/- 21.20
	Oct	68.67 +/- 64.30	53.00 +/- 18.52	26.00 +/- 16.09
	Nov	74.67 +/- 70.50	56.67 +/- 17.47	31.00 +/- 20.30
	Dec	71.00 +/- 65.20	54.00 +/- 16.52	27.33 +/- 18.23
2005	Jan	68.67 +/- 61.08	55.67 +/- 18.72	29.67 +/- 17.79
	Feb	69.67 +/- 63.95	57.33 +/- 21.57	29.00 +/- 19.00

increased considerably early on in the study (March, April, and May 2004) as a result of a leafing event that made the plants easier to identify into the spring. However, once all the plants were tagged, I saw little recruitment of plants into each plot, as the number remained fairly constant (Table 5).

Morphologic Patterns of L. carolinianum in ANWR Marshes Data presented in this section were taken from the November 2004 sampling of the permanent macrophyte plots because it represented the largest number of plants measured during this period of study. These data were summarized and is intended to provide a brief characterization of *L. carolinianum* in the ANWR saltmarsh landscape. The number of plants measured during the November 2004 sampling at Boat Ramp was 224, 144 at Pump Canal, and 93 at Sundown Bay.

Boat Ramp had the shortest plants, while Pump Canal had the tallest plants. Mean height at Boat Ramp was 37.4 cm, 55.0 cm at Pump Canal, and 44.4cm at Sundown Bay (Figure 20). Maximum height in the November 2004 sampling was 97.0 cm, found at the Boat Ramp site. The tallest plant at Pump Canal was 95.5 cm, and 84.0 cm at Sundown Bay. Plant height at all three sites were significantly different from each other (Kruskal Wallis and Dunnett's post hoc, alpha= 0.05).

The branching of this plant had the potential to be quite extensive (Figure 21). The maximum number of branches measured was 16 at Sundown Bay. The greatest number of branches at Pump Canal was 15, and 7 at Boat Ramp. However, the normal branching patterns were not this high; the mean number of branches at Boat Ramp was
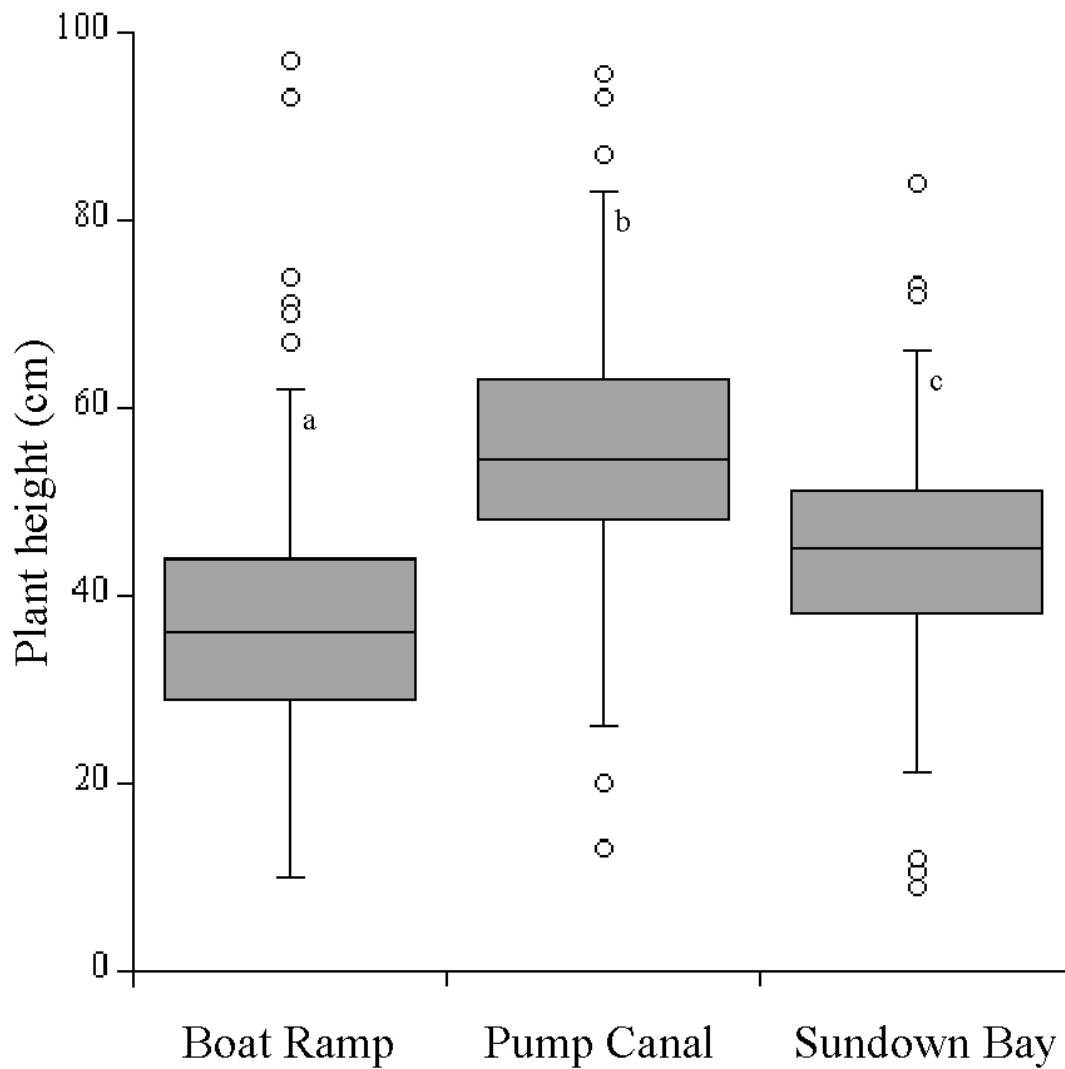
Figure 20: Box plot of *L carolinianum* plant heights measured in permanent macrophyte plots in November 2004. Different letter indicate significant differences (alpha = 0.05)
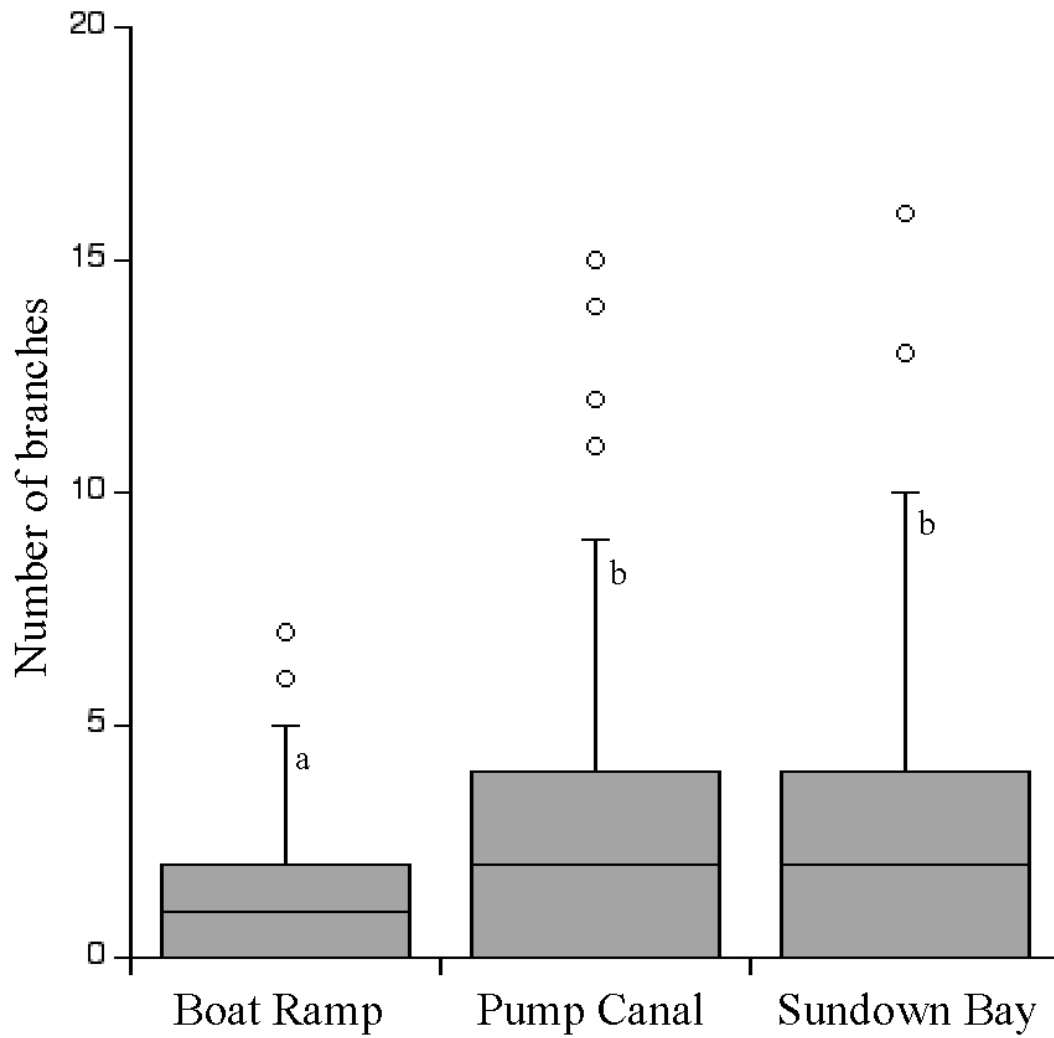
Figure 21: Box plot of number of branches on *L. carolinianum* plants measured in the permanent macrophyte plots in November 2004. Different letters indicate significant differences.

1.2 branches, 2.7 at Pump Canal, and 2.5 at Sundown Bay. Number of branches at Boat

Ramp was significantly different than the number of branches at Pump Canal and

Sundown Bay (Kruskal Wallis and Dunnett's post hoc, alpha=0.05)

Growth Patterns of L. carolinianum in ANWR Marshes *Lycium carolinianum*

exhibited seasonality in leaf, flower, and fruit production. Peak leaf production occurred

in February 2004, and as the season progressed, the plants shed most of their leaves

(Figure 22). I saw a secondary, smaller peak in leaf production early fall, just prior to

peak berry abundances (Figures 22 and 23). *L. carolinianum* flowering and subsequent

fruit (i.e., berry) production occurred just after the September peak in leaf production

(Figure 22).

Flowers peaked in October and berries were most abundant in November, but

were frequently observed as late as December (Figure 23). Sundown Bay sites produced

more berries per plant than the other two sites, though the means were not statistically

different. In October 2004, Sundown Bay sites produced significantly more flowers per

plant ($p < 0.01$) than the other sites. This higher number of flowers corresponded with

noticeably higher numbers of berries per plant at this site, but the differences were not

statistically significant (Figure 23).

Non-Destructive Estimation of Biomass Stepwise linear regression indicated that

for the 118 *L. carolinianum* plants collected throughout the year, diameter at the base of

each *L. carolinianum* plant best explained the variability in aboveground biomass.

Further, a polynomial model provided the "best fit" for stem diameter and aboveground

biomass with an R^2 value of 0.94 and p-value < 0.0001 (Figure 24). Since the permanent
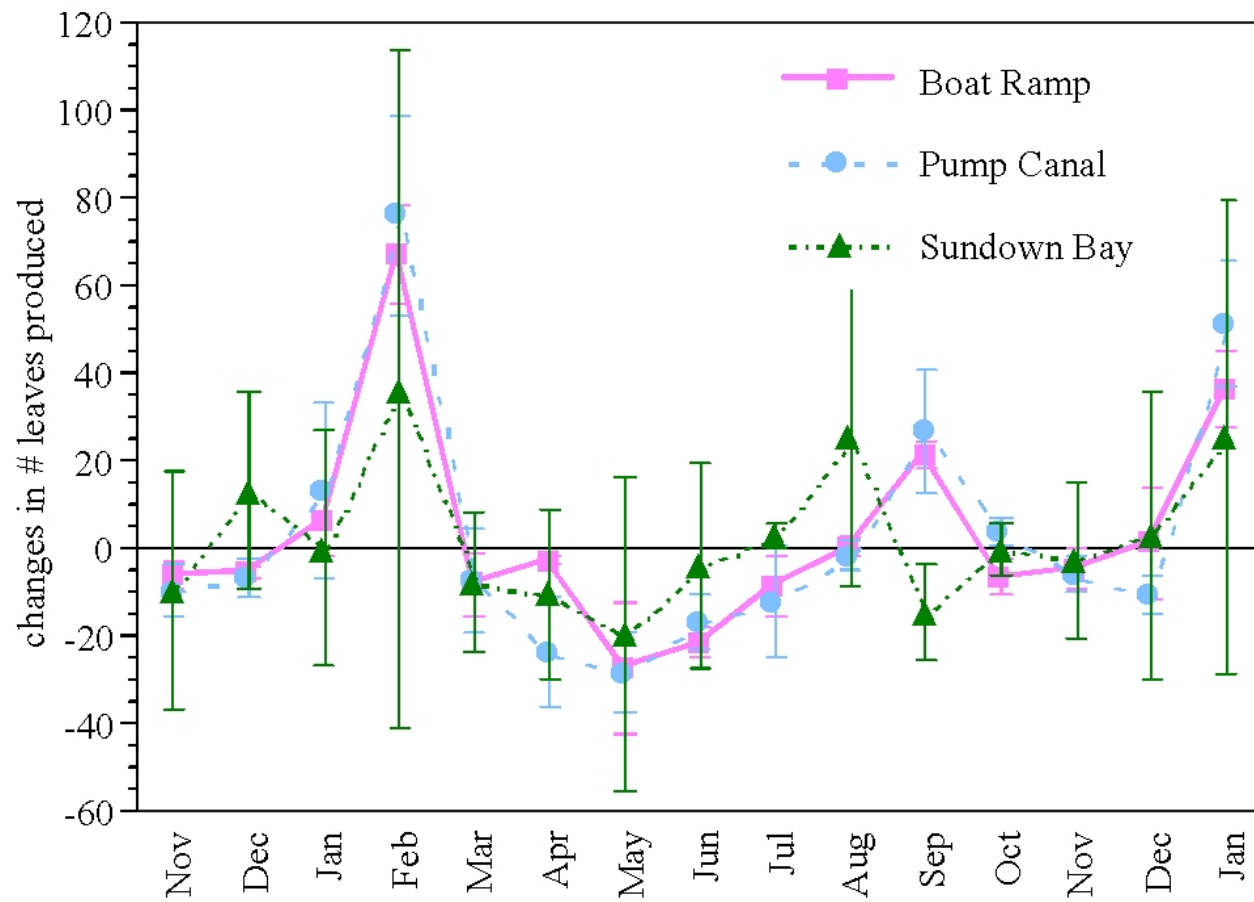
Figure 22: Leaf production per plant per site · m^{-2}· month. Measurements were taken from November 2003 to February 2005 at the beginning of each month; the value for November reflects change in leaf production from the November to December sampling.
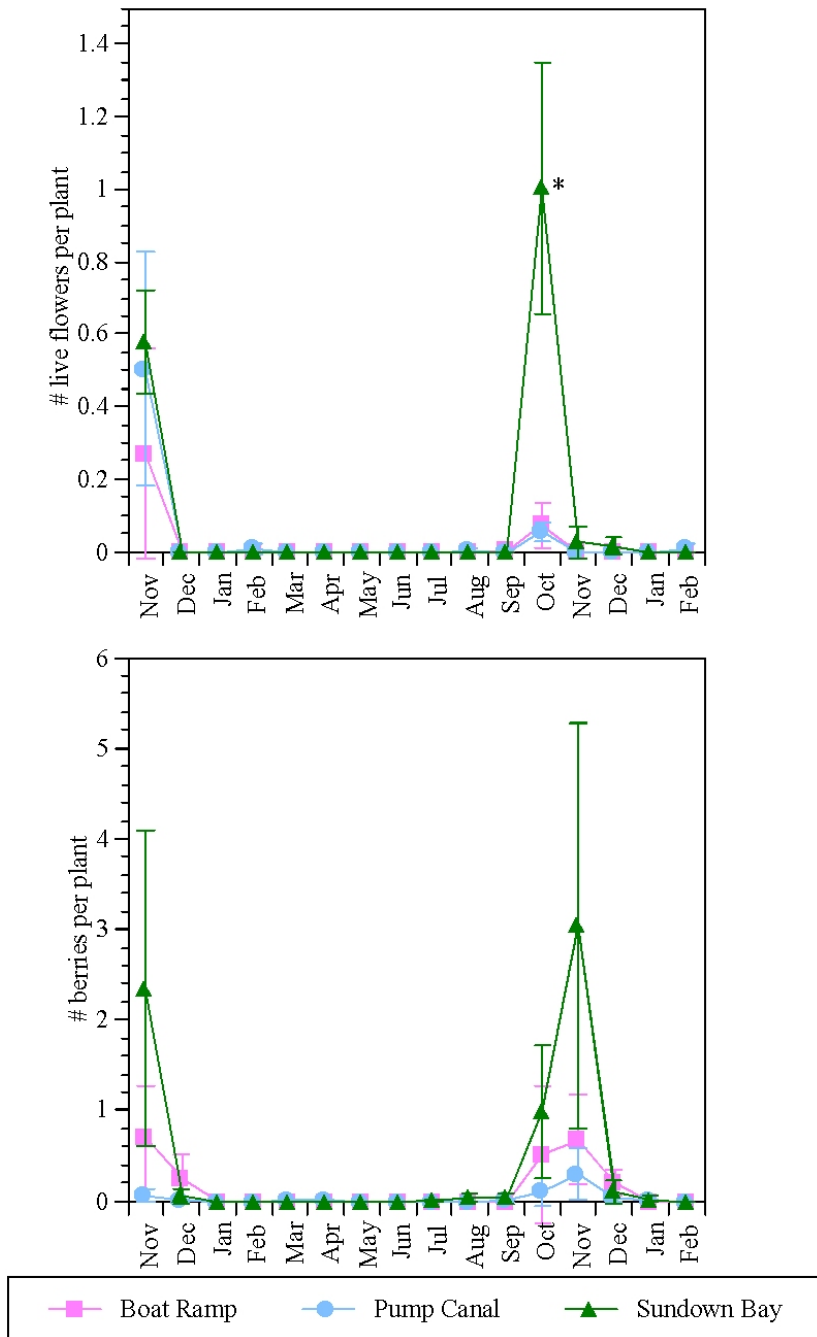
Figure 23: Mean number (+/- SD) of flowers and berries per plant from permanent macrophyte plots. Number of flowers at Sundown Bay during the October 2004 sampling was higher than the other two sites (p<0.01). Though site to site trends in number of berries appear, berry abundances between sites was not significantly different at any site (p<0.05).
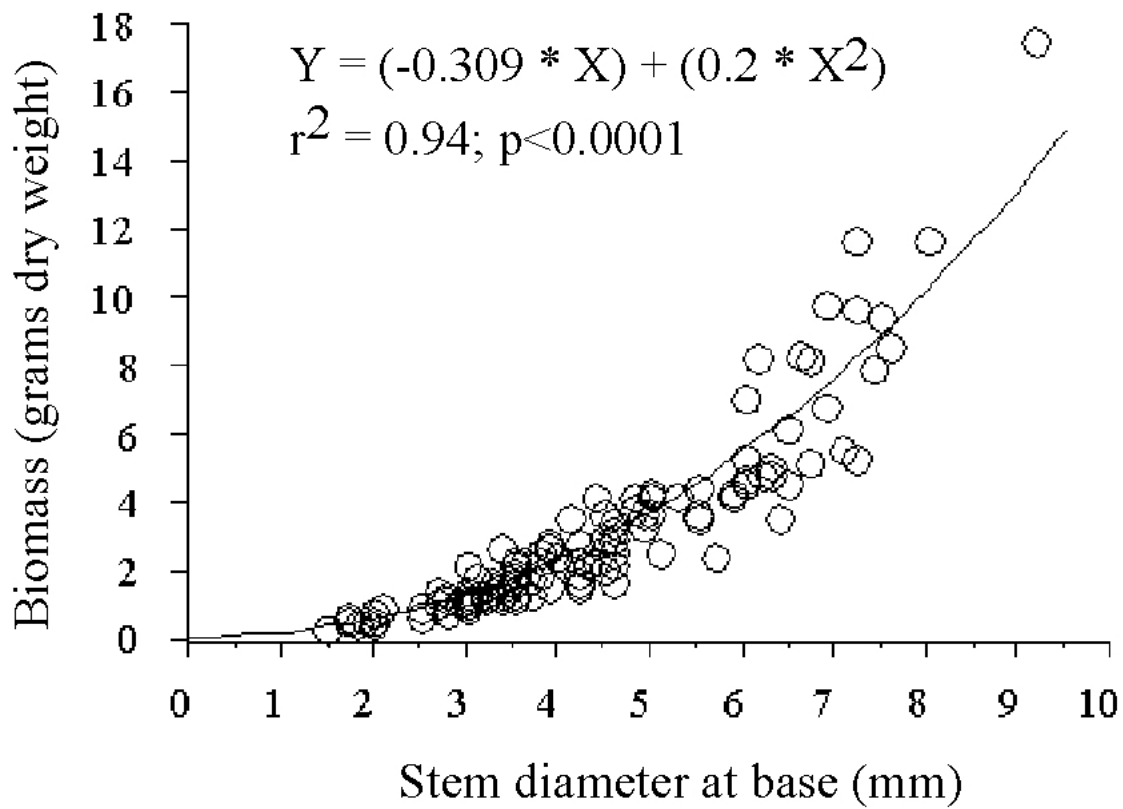
Figure 24: Polynomial regression derived from phenometric measurements of 118 *L. carolinianum* plants relating these measurements to biomass (g). Stem diameter alone accounted for 94% of the variability in aboveground biomass (p<.001).

macrophyte plots had been measured monthly, *L. carolinianum* stem diameter data were available for the 16 months of the study. The regression equation (Figure 24) was applied to stem diameter data collected during each monthly sampling and used to estimate changes in live aboveground biomass of *L. carolinianum* inside each permanent macrophyte plot. I also calculated changes in aboveground biomass from one month to the next and assumed this to be an indication of net aboveground primary productivity (Figure 25). These data show that aboveground biomass of *L. carolinianum* in my plots fluctuated about zero throughout the study, but did not follow any distinct seasonal patterns. Boat Ramp exhibited noticeably greater fluctuation than the other sites during the months of September and November (Figure 25).

DISCUSSION

The wintering grounds of the cranes, the Guadalupe Estuary, like many estuaries in Texas, is facing increased demands in freshwater due to population growth in the watershed. Understanding basic plant ecology of *L. carolinianum* is imperative before we can even speculate the impacts of changing environmental conditions**.**

The number of *L. carolinianum* plants in each of the m^2 macrophyte plots was highly variable, ranging from 13-139. This variability (Table 6) was representative of the variability in *L. carolinianum* distribution in the marsh. In certain vegetation stands, *L. carolininanum* was absent or only a small component of the community; in others it was very abundant (as high as 139 per m^2). However, from the field observations at
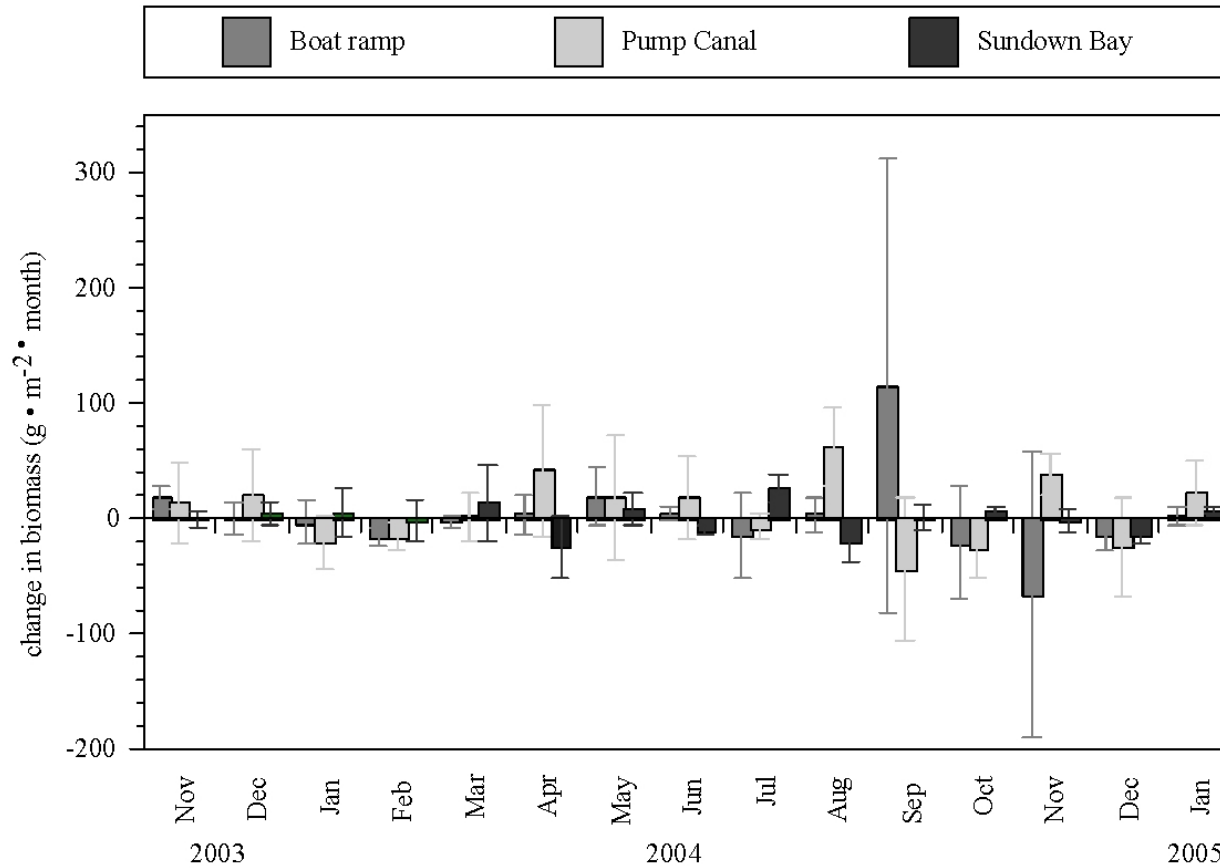
Figure 25: Non-destructive estimation of change in biomass in g· m^{-2}· month. These values were obtained by applying the polynomial regression equation to the diameter measurements taken from the permanent macrophyte plots. Measurements were taken from November 2003 to February 2005 at the beginning of each month; the value for November reflects change in biomass from the November to December sampling.

ANWR, *L. carolinianum* does not colonize areas completely in monotypic stands like *S. alterniflora* or *D. spicata*.

     Lycium carolinianum exhibited spatial and temporal variability. The plant exhibited high plasticity in terms of its growth forms, ranging from short solitary stems to numerously branched thick woody stems, to creeping vine-like stems. There was variability in plant height, though a majority of the plants in the marshes at ANWR were significantly shorter (Figure 20) than the projected 0.9-3.0 m height of this species (Stutzenbaker 1999, Godfrey and Wooten 1981). In most cases, *L. carolinianum* possessed a woody stem, unless the plant experienced rapid shoot growth, in which case the stem is green. While the branching patterns had the potential to be quite extensive (Figure 21), *L. carolinianum,* in the marshes at ANWR, was not nearly as bushy as the projected 3-6 foot spread of this plant (Gilman 1999). This species in the marshes at ANWR appear to be shorter and less busy and branched those the same species found in other ecosystems.

     The complexity of the basic structure was further complicated by the plant's leafing patterns (Figure 22). During several months (early spring and late fall) of the sampling period, leaf production was high. But during other times of the year it was common for plants to shed their leaves and possibly remain leafless throughout numerous sampling months.

     Berry production in early winter was similar to patterns previously described (Godfrey and Wooten 1981, Chavez 1996). I found that the fruiting patterns of *L. carolinianum* were very distinct and predictable. Berry production and related processes

(budding, flowering, etc) were virtually non-existent throughout the year until October and November (Figure 23). Berry abundance peaked in late October and early November, which coincided with the arrival of the whooping cranes each year. In fact, 94% and 98% of the cranes had arrived at ANWR by November of 2003 and 2004 (Table 7).   After the greatest peak in November, December had a small number of berries, and, in January, the number of berries present approached zero. While the project did demonstrate the timing of peak berry abundances corresponding with the cranes arrival, it did not address berry availability. In order to really understand salt marsh ecology with regards to environmental conditions and the whooping crane population, it is essential to understand berry availability.

The highest numbers of berries, though differences were insignificant, were found at Sundown Bay, which had the highest intra-annual range of surface water salinity of the three sites (Figures 19 and 23). However, salinity did not appear to be a strong determinant of berry production, as Boat Ramp had similar numbers in berries as the more saline Pump Canal (Figure 23). Water depth was consistently greater at Sundown Bay; project observations have indicated that Sundown Bay is lower in elevation than the other two sites. Therefore, the plots are inundated more frequently or with deeper water than the other two sites while water depths at Pump Canal and Boat Ramp were more similar to each other. While Webb and Mendelssohn (1996) concluded that the combined effect of both increased salinity and submergence was most detrimental to plant growth, in contrast, I found the combined effect was not necessarily negative, but may have translated to higher production. Therefore, it may be that factors

Table 7: Percentage of whooping cranes arrived on ANWR by the end of each month.
Percentages are based on total population of 194 (2003-2004) and 217 (2004-2005).
These numbers include juvenile and adult whooping cranes. Data were supplied by Tom
Stehn's 2003-04 and 2004-05 annual reports (Stehn 2004, Stehn 2005)

	2003-04	2004-05
Oct	25.26	8.29
Nov	93.3	98.16
Dec	100	99.54

other than salinity and water level contribute to the fruiting patterns I saw. Studies have indicated that the effects water level (McKee and Mendelssohn 1989), elevation (Pennings and Callaway 1992) or soil properties (Silvestri 2005) can cause the effects of salinity to be greater than the effects of salinity alone. Perhaps in the marshes at ANWR, soil properties are acting in conjunction with salinity and water level to produce these patterns.

In ecosystems that experience extremes in environmental conditions such as salinity and inundation, the ability to track changes in aboveground biomass is important. Developing a non-destructive method to estimate biomass is a beneficial tool in understanding plant productivity under variable conditions, and has been done with several freshwater and salt marsh species (Morris and Haskin 1990, Daoust and Childers 1998, Thursby et al. 2002). I found that stem diameter alone proved to be the best predictor of live aboveground biomass, accounting for 94% of the variability in aboveground biomass (p<.001). When the monitoring efforts began in November 2003, many plants were leafless - appearing dead and were not included in the sampling. However, when the plants experienced the spring productivity, many of these plants that had appeared dead produced leaves, and then they were added to the database. Therefore plot biomass maybe under-estimated until March 2004 (Figure 25), and as evidenced by the large jump in *L. carolinianum* numbers in Table 6.

Unlike other studies on grass-like and herbaceous marsh species like *S. alterniflora, S. patens,* and *D. spicata,* (Hopkinson et al. 1978, Morris and Haskin 1990, Pezeshki and DeLaune 1991) that have distinct annual or seasonal peaks, *L.*

carolinianum displayed no distinct temporal patterns in aboveground biomass production (Figure 25); I found both positive and negative spikes in aboveground biomass over the study period, resulting in very little net growth (Figure 25). However, there was no observable link between this growth and seasonal or site-specific conditions. This type of growth has been shown in other coastal plant communities and may be a response to environmental forcings such as nutrient availability, tropical storms, frontal passages, or high inflow events (Gallagher 1975, Gratton and Denno 2003). The growth I observed for *L. carolinianum* may also be an artifact of my biomass model that uses stem diameter as a measure of biomass. This species has a woody stem that accounts for most of the living aboveground biomass and shows little net change in size over time. This is just the opposite of *L. carolinianum*'s leaves that are relatively insignificant in terms of their contribution to total biomass, but showed the greatest seasonal fluctuation. As a result, stem diameter is an important means of non-destructively tracking biomass over large temporal and spatial scales, but leaf counts will likely better reflect this plant's metabolic activity and response to environmental stoachasticity.

In conclusion, *L. carolinianum* exhibited distinct fruiting and leafing patterns. The berry production coincided with the arrival of the whooping cranes in late October and early November, providing an energy source to this endangered species. Regarding the importance of freshwater inflows, this study encompassed both lows and extreme highs in Guadalupe River inflows to the estuary. The project commenced in late 2003 during a period of relatively low inflow, but inflows were "average" or well above

average for all of 2004 and into early 2005. As a result, surface water and pore water salinity at these three sites did not exceed 35 ‰ during the period of this study. Given the lack of hypersaline conditions, I was unable to offer much insight into the effects of salinity on the growth of *L. carolinianum* and production of fruits in ANWR marshes.

However, *L. carolinianum* also exhibited site-to-site variability, which may be a result of some combination of inundation patterns and other physico-chemical or environmental variables such as soil properties that were not included in my monitoring. Additionally, it would be very important to further address berry availability. The *L. carolinianum* plants at Pump Canal and Sundown Bay had more complex branching patterns than Boat Ramp, perhaps affecting the availability of the berries – i.e., the branches are more spread out, making it easier for animals to pick the berries. Also, since *L. carolinianum* in the salt marshes at ANWR seemed to have a different geometry (short and less branched) than the same species in other ecosystems, further investigation into the plant geometry would help understand *L. carolinainum's* importance in the ANWR saltmarsh ecosystem.

Monitoring populations in the field can be difficult as you have little control over weather and tidal patterns, and time is a constraint. Continual monitoring of *L. carolinianum* population may be beneficial to further understanding the marsh ecology at ANWR, A longer data set may include a spectrum of environmental conditions that may clarify the effects of changing conditions on patterns in the aboveground biomass data and berry production.

CHAPTER IV

SUMMARY AND CONCLUSIONS

Lycium carolinianum is an important component of the salt marsh ecosystem on the Gulf Coast of Texas because it is one of the known food sources for the endangered whooping crane as well as numerous other birds and small mammals that utilize these marshes. The wintering grounds of the cranes, the Guadalupe Estuary, like many estuaries in Texas, is facing increased demands in freshwater due to population growth in the watershed. The Lower Guadalupe Water Supply Project (LGWSP) was conceived as a means of providing additional water to meet the needs of South Texas, while maintaining critical estuarine and spring inflows (WWW.LGWSP.ORG). This project proposes a withdrawal of freshwater from just below the confluence of the San Antonio and Guadalupe Rivers to meet projected municipal demands in the rapidly expanding San Antonio metropolitan area. In order to understand the potential impacts of this diversion on the crane's wintering grounds—as required under the Endangered Species Act—the agencies that make up LGWSP have funded an ecosystem project, the San Antonio Guadalupe Estuary System (SAGES) project, to study the influence of freshwater inflows on the salt marsh ecosystem at ANWR.

Three sites were sampled along the length of the Blackjack Peninsula at ANWR that were representative of the variability in salinity, elevation, and vegetation cover types typically found throughout salt marshes along the Blackjack Peninsula. While the diversity is similar at the sites along the peninsula, those sites closer to freshwater input have more species, and may be more sensitive to salinity fluctuations. Species

composition is similar at the three sites, with all sites sharing 6-7 common species. Species composition exhibited very little variability from year one to year two of the study. Densities and biomass of *L. carolinianum* were not significantly different between sites or years either. Overall, the three sites were similar in regards to their vegetation community. *L. carolinianum,* while important to salt marsh ecology, accounts for only a small portion of the overall productivity (live aboveground biomass).

Through correlation analyses, I was able to determine which species are more likely to be found with *L. carolinianum* (*B. maritima, B. frutescens M. littoralis,* and in the second year, *D. spicata*), and which were not (*S. alterniflora*). Based in these correlations, *L. carolinianum* is found in association with some of the common species, and the Correspondence Analysis indicated that its growth and survival requirements are typical to the salt marshes at ANWR.

Lycium carolinianum exhibited distinct fruiting and leafing patterns. The berry production coincided with the arrival of the Whooping Cranes in late October and early November. *L. carolinianum* also exhibited site-to-site variability, which may be a result of some combination of inundation patterns and other physico-chemical or environmental variables such as soil properties that were not included in my monitoring. The plant geometry of *L. carolinianum* in the salt marshes at ANWR was unique to, in that the plants measured were shorter and less bushy (as a result of being less branched) than this species is described (Carlton 1975, Godfrey and Wooten 1981, McAlister and McAlister 1995, Stutzenbaker 1999).

Continual monitoring of *L. carolinianum* population may be beneficial to further understanding the marsh ecology at ANWR. The fact that *L. carolinianum* in the salt marshes ANWR seemed to diverge from the common description, and that there may be site-to-site differences in its morphology necessitates further study. Additionally, a longer data set may include a spectrum of environmental conditions that may clarify the effects of changing conditions on patterns in the aboveground biomass data and berry production. Future studies that incorporate more sites along the peninsula, and sample at a more intensive scale, may be able to discern any differences in the patterns along the salinity gradient.

Large shifts in the vegetation community may require events on a scale that exceeds their natural threshold (i.e., longer duration of freshwater inundation) (Zedler 1983). This study, as evidenced by similar vegetation frequencies from year to year, just illustrates how the estuary handles the natural year-to-year variability. However, with pressures on estuaries nationwide, further studies looking at the effects of changes in water quality and quantity on the vegetation community will help to understand this system better, and may be applicable in other similar systems.

LITERATURE CITED

Alexander, H.D, and K.H. Dunton. 2002. Freshwater inundation effects on emergent vegetation of a hypersaline salt marsh. Estuaries 25:1426-1435.

Baldwin, A. H., M. S. Egnotovich, and E. Clark. 2001. Hydrologic change and vegetation of tidal freshwater marshes: field, greenhouse, and seed-bank experiments. Wetlands 21(4): 519-531.

Bianchi, T.S., J.R. Pennock, and R.R. Twilley. 1999. Biogeochemistry of Gulf of Mexico estuaries: implications for management. In T.S. Bianchi, J.R. Pennock, and R.R. Twilley (eds.)   Biogeochemistry of Gulf of Mexico estuaries. John Wiley and Sons. New York, NY, USA.

Bishop, M. A. 1984. The dynamics of subadult flocks of whooping cranes wintering in Texas, 1978-79 through 1982-1983. M.S. Thesis, Texas A&M University, College Station, TX, USA.

Bonds, C. J. 2000. Characterization of banded whooping crane winter territories from 1992-93 to 1996-97 using GIS and remote sensing. M.S. Thesis, Texas A&M University, College Station, TX, USA.

Carlton, J.M. 1975. A guide to common Florida salt marsh and mangrove vegetation. Florida Marine Research Publications 6: 30.

Chavez-Ramirez, F. 1996. Food availability, foraging ecology, and energetics of whooping cranes wintering in Texas. Ph.D. Dissertation, Texas A&M University, College Station, TX, USA.

Cronk, J.K, and M.S. Fennessey. 2001. Wetland plants: biology and ecology. Lewis Publishers. Boca Raton, FL, USA.

Daoust, R.J., and D.L. Childers. 1998. Quantifying aboveground biomass and estimating net aboveground primary production for wetland macrophytes using a non-destructive phenometric technique. Aquatic Botany 62:115-133.

David, Peter. 1996. Changes in plant communities relative to hydrologic conditions in the Florida Everglades. Wetlands 16:15-23.

DeLuane, R.D., S.R. Pezeshki, and W.H. Patrick, Jr. 1987. Response of coastal plants to increase in submergence and salinity. Journal of Coastal Research 3:535-546.

Dunton, K.H., B. Hardegree, and T.E. Whitledge. 2001. Response of estuarine marsh vegetation to interannual variations in precipitation. Estuaries 24:851-861.

Fitch, J.A., and N.E. Armstrong. 1982. Technical Report CRWR-200. Prediction of salinities in the Matagorda Bay area. Center for Research in Water Resources. University of Texas at Austin, Austin, TX, USA.

Gallagher, J. L. 1975. Effect of an ammonium nitrate pulse on the growth and elemental composition if natural stands of *Spartina alterniflora* and *Juncus roemerianus*. American Journal of Botany 62:644-648.

Giles, R.H. 1978. Wildlife management. W.H. Freeman and Company. San Francisco, CA, USA.

Gilman, E.F. 1999. Fact sheet FPS-356. University of Florida Cooperate Extension Service, Institute of Food and Agricultural Sciences, Gainesville, FL, USA.

Godfrey, R.K., and J.W. Wooten. 1981. Aquatic and wetland plants of the southeastern United States. University of Georgia Press. Athens, GA, USA.

Gough, L. and J. B. Grace. 1998. Effects of flooding, salinity, and herbivory on coastal plant communities, Louisiana, United States. Oecologia 117: 527-535.

Gratton, C. and R. F. Denno. 2003. Inter-year carryover effects of a nutrient pulse on *Spartina* plants, herbivores, and natural enemies. Ecology 84:2692-2707.

Hopkinson, C.S, J.G. Goeeslink, and R.T. Parrondo. 1978. Aboveground production of seven marsh plant species in coastal Louisiana. Ecology 59:760-769.

Howard, R. J. and I. A. Mendelssohn 2000. Structure and composition of oligohaline marsh plant communities exposed to salinity pulses. Aquatic Botany 68: 143-164.

Jassby, A.D., W.J. Kimmerer, S.G. Monismith, C. Armor, J.E. Cloern, T.M. Powell, J.R. Schubel, T.J. Vendilinski. 1995. Isohaline position as a habitat indicator for estuarine populations. Ecological Applications 5:272-289.

Johns, N.D., M. Hess, S. Kaderka, L. McCormick, and J. McMahon. 2004. Bays in peril: A forecast for freshwater flows to Texas estuaries. National Wildlife Federation, Reston, VA, USA.

Kennish, M. J. 2001. Coastal salt marsh systems in the U.S.: a review of anthropogenic impacts. Journal of Coastal Research 17: 731-748.

McAlister W. H. and M.K. McAlister. 1995. A naturalist's guide: Aransas. University of Texas Press. Austin, TX, USA.

McCune, B., J. P. Dey, J. E. Peck, D. Cassell, K. Heiman, S. Will-Wolf, and P. N. Neitlich. 1997. Repeatability of community data: species richness versus gradient scores in large-scale lichen studies. The Bryologist 100:40–46.

McCune, B. and J. B. Grace, with a contribution from D. L. Urban. 2002. Analysis of ecological communities. Mjm Software Design, Gleneden Beach, OR, USA.

McKee, K. L. and I. A. Mendelssohn 1989. Response of a freshwater marsh plant community to increased salinity and increased water level. Aquatic Botany 34: 301-316.

Mitsch, W.J., and J.G. Gosselink. 1993. Wetlands, second edition. John Wiley and Sons, Inc. New York, NY, USA.

Morris, J.T. and B. Haskin. 1990. A 5-yr record of annual primary production and stand characteristics of *Spartina alterniflora*. Ecology 71: 2209-2217.

Nestler, J. 1977. Interstitial salinity as a cause of ecophenic variation in *Spartina alterniflora*. Estuarine and Coastal Marine Science 5:707-704.

Odum, W. E. 1988. Comparative ecology of tidal freshwater and salt marshes. Annual Review of Ecology and Systematics 19:147-176.

Orlando, S.P., Jr., L.P. Rozas, G.H. Ward, and C.J. Klein. 1993. Salinity characteristics of Gulf of Mexico estuaries. National Oceanic and Atmospheric Administration, Office of Ocean Resources Conservation and Assessment, Silver Spring, MD, USA.

Pennings, S.C. and R.M. Callaway. 1992. Salt marsh plant zonation: the relative importance of competition and physical factors. Ecology 73:681-690.

Pezeshki, S.R and R.D. DeLaune. 1991. A comparative study of aboveground productivity of dominant U.S. Gulf Coast marsh species. Journal of Vegetation Science 2:331-338.

Purer, E.A. 1942. Plant ecology of the coastal salt marshlands of San Diego County, California. Ecological Monographs 12:81-111.

Silvestri, S., A. Defina, and M. Marani. 2005. Tidal regime, salinity, and salt marsh plant zonation. Estuarine, Coastal, and Shelf Science 62:119-130.

Smart, M.R., and J.W. Barko. 1978. Influence of sediment salinity and nutrients on the physiological ecology of selected marsh plants. Estuarine and Coastal Marine Science 7:457-495.

Smart, M.R., and J.W. Barko. 1980. Nitrogen nutrition and salinity tolerance of *Distichlis spicata* and *Spartina alterniflora*. Ecology 6:630-638.

Stehn, T. V. 1992. Unusual movements and behaviors of color banded whooping cranes during winter. Proceedings of the North American Crane Workshop 6:95-101.

Stehn, T. 2004. Whooping cranes during the 2003-2004 winter. U.S Fish & Wildlife Service, Aransas National Wildlife Refuge, Austwell, TX, USA.

Stehn, T. 2005. Whooping cranes during the 2004-2005 winter. U.S. Fish & Wildlife Service, Aransas National Wildlife Refuge, Austwell, TX, USA.

Stutzenbaker, C. D. 1999. Aquatic and wetland plants of the western Gulf Coast. Texas Parks and Wildlife Press, Austin, TX, USA.

Tanyeri-Abur, Aysen, L. Jones, and H. Jiang. 1998. Guadalupe Estuary: Economic impacts of recreational activities and commercial fishing. Prepared for Texas Water Development Board by Department of Agricultural Economics, Texas A&M University, College Station, TX USA.

Thursby, G.B., M.M. Chintala, D. Stetson, C. Wigand, and D.M. Champlin. 2002. A rapid, non-destructive method for estimating aboveground biomass of salt marsh grasses. Wetlands 22:626-630.

United States Fish and Wildlife Service. 2005. DRAFT whooping crane recovery plan. U.S Fish & Wildlife Service, Aransas National Wildlife Refuge, Austwell, TX, USA.

Van der Valk, A.G., L. Squires, and C.H. Welling. 1994. Assessing the impacts of an increase in water level on wetland vegetation. Ecological Applications 4:525-534.

Webb, E.C., and I.A. Mendelssohn. 1996. Factors affecting vegetation dieback of an oligohaline marsh in coastal Louisiana: field manipulation of salinity and submergence. American Journal of Botany 83:1429-1434.

Webb, J.W. 1983. Soil water salinity variations and their affects on *Spartina alterniflora*. Contributions in Marine Science 26:1-13.

Whittaker, R. H. 1972. Evolution and measurement of species diversity. Taxon 21:213-251.

Zedler, J. B. 1983. Freshwater impacts in normally hypersaline marshes. Estuaries 6:346-355.

VITA

Name: Rachel Elizabeth Butzler

Address: 201 Old Herman Heep, Department of Wildlife and Fisheries Sciences,
               Texas A&M University, College Station, TX 77843-2258

Email Address: rbutzler@tamu.edu

Rachel received her B.S. in Environmental Resource Management and a Marine Science

minor from The Pennsylvania State University, University Park, Pennsylvania. After

graduation, she took several seasonal temporary positions, one in Lake Placid, Florida,

another in Panama, and then back to Central Pennsylvania for another. It was while

working as a technician at the USDA-ARS in Beltsville, MD, that she realized she

wanted to go to graduate school studying wetlands.