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A comparison of channel morphology, marsh elevation, and biological processes in natural and dredged tidal salt marshes in Louisiana

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A COMPARISON OF CHANNEL MORPHOLOGY, MARSH ELEVATION, AND BIOLOGICAL PROCESSES IN NATURAL AND DREDGED TIDAL SALT MARSHES IN LOUISIANA

A Dissertation

Submitted to the Graduate Faculty of the
Louisiana State University and
Agricultural and Mechanical College
in partial fulfillment of the
requirements for the degree of
Doctor of Philosophy

in

The Department of Oceanography and Coastal Sciences

by

Jennifer Spicer
B. S., Coastal Carolina University, 1998

December 2007

Oh, what is abroad in the marsh and the terminal sea?

Somehow my soul seems suddenly free…

--Sidney Lanier

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ABSTRACT

Channel networks in tidal marshes serve as conduits for the exchange of water between the estuary and the marsh surface and form the spatial structure that regulates the flow of organisms, materials, and energy throughout the marsh system. Canal dredging creates channels with distinct linear shapes, and the artificial levee created from the dredged material creates 'spoil banks' along either side of the channel edge. Differences in morphology and adjacent marsh surface elevation between natural and dredged channel systems have not been well quantified, but may have important implications for the functioning of tidal channels as habitat for marsh-dependent species.

I used GIS methods commonly applied to terrestrial landscape patterns, Light Detection and Ranging (LiDAR) data, and aerial orthophotographs to examine differences in the shape complexity of natural and dredged channel networks and to compare the elevation of the marsh adjacent to natural and dredged channels and productive and non-productive oil and gas wells in a tidal salt marsh in Louisiana. I examined the distribution of *Spartina alterniflora*, *Geukensia demissa demissa*, and *Littoraria irrorata* in relation to channel morphology, soil properties and elevation of the marsh edge at natural creeks, dredged canals, and open bay edges in order to correlate landscape patterns with ecological processes.

Dredged canal networks were significantly less complex in shape. The elevation of the marsh within 30 m of the channel edge was highest along dredged channels. The marsh at distances greater than 30 m from dredged canals appears to be more broken up, but not lower in elevation than in natural systems. Densities of *S. alterniflora* and *G. demissa* were lower, and soil and leaf nitrogen were less in marsh adjacent to dredged canals. The age distribution of *G. demissa* populations provided evidence that structural differences between natural creeks and dredged canal systems influence the recruitment, growth, and mortality of mussels. This work demonstrates that landscape scale changes to tidal marshes have the potential to alter ecological processes at the microhabitat scale. Dredged canals may reduce the growth and productivity of plants and animals that are critical to maintaining coastal Louisianans marshes.

CHAPTER 1

INTRODUCTION: LANDSCAPE CHANGES IN TIDAL SALT MARSHES

Tidal salt marshes may be simply defined as low coastal grasslands that are frequently inundated by the tide. They are found on temperate shores along the edges of estuaries and on the inland side of barrier islands, where there is sufficient protection from wave energy (Chapman 1977). Although they tend to be low in species diversity, they are considered to be among the most productive ecosystems on earth (Mitsch and Gosselink 1993). Through out history, tidal salt marshes have been a somewhat inconspicuous and often overlooked ecosystem; today they are recognized for the numerous values and benefits they provide to society, including wildlife habitat, nursery grounds for many recreationally and commercially important fish, nutrient retention, and storm surge abatement. The loss of coastal wetlands both world-wide and in Louisiana is alarming, and there is a need for research to facilitate understanding, preservation, and restoration of this valuable habitat.

The Romans “reclaimed” salt marshes for agriculture, and modification of tidal marshes began as early as the seventh century, when low sea walls were constructed in Western Europe for protection from the ocean. Since then, tidal marshes have been manipulated for grazing, mining, agriculture, harvest, waste disposal, land reclamation, wildlife management, insect control, and recreation though the use of dikes, embankments, impoundments, sluice gates, ditches, and direct fill (Daiber 1986). Until the mid-1900’s, undeveloped marshland was often considered wasteland. Government programs, such as the US Swamp Lands Act of 1850 and the Canadian Maritime Marsh

Rehabilitation Act of 1948 encouraged landowners to re-claim these “un-usable” lands and convert them to a useful purpose such as agriculture.

An understanding of the functions and values of wetlands began to emerge in the mid-1900’s with the development of new quantitative analysis based on physics and biochemistry that made it possible for ecologists to calculate an ecosystem’s potential for converting solar energy into production (Siry 1984). Today tidal salt marshes in the US are regulated under the Clean Water Act, and consultatively by the Fish and Wildlife Coordination Act, the Endangered Species Act, the Marine Mammal Protection Act, and the National Environmental Policy Act.

This dissertation focuses on the specific issues of marsh modification in coastal Louisiana as an example of an extensive and important coastal wetland that has been heavily impacted by human actions. From the air, one of the most prominent features of Louisiana tidal marshes is the grid of dredged canals that cross the marsh surface. In many areas, this man-made system has replaced the natural creek network. In tidal marshes, the creek network serves as a conduit for the exchange of water between the open estuary and the marsh surface and creates a spatial structure that regulates the flow of organisms, materials, and energy throughout the system. To understand how the addition of dredged canals to the marsh system has impacted the functioning of the system, it is necessary to first devise a method of quantifying the differences in landscape structure of natural and anthropogenically affected tidal systems. Habitat shape has been shown to affect species behavior and distribution in terrestrial systems (e.g., Hovel and Lipcius 2001), and channel network shape may similarly affect nekton use of tidal channels. In Chapter 2, I use GIS methods commonly applied to terrestrial landscape patterns to quantify the differences in shape between natural and created channel networks. Perimeter-to-area ratio and measures and shape complexity of modified and natural tidal channel networks are compared using Patch Analyst, a GIS extension that is used by landscape ecologists to understand how habitat complexity affects populations of organisms. The analysis of channel networks as landscape patches from a “fish’s perspective” may provide insight into how the movement of organisms and materials is affected by changing from a system of sinuous, highly complex creeks, to uniform, linear dredged canals.

I use LiDAR elevation data to examine marsh surface elevation near dredged canals, natural channel systems, and near producing and non-producing oil and gas wells in Chapter 3. ‘Spoil banks’ are created by the deposition of dredged material along either side of newly-built channel edges. In Louisiana these spoil bank ridges, which may be 20 to 30 m wide and 2 m high when constructed, block sheet flow across the marsh surface and may form semi-impoundment marshes in areas of high density dredging (Swenson and Turner 1986). Subsurface fluid withdrawal, extracted as groundwater, oil, or gas during mineral recovery operations and the resulting geologic faulting have also been suggested mechanisms for marsh subsidence (Morton et al. 2005). These changes to marsh hydrology will affect vegetative growth, perhaps lethally, and may reduce net organic deposition behind the spoil bank ridges. I use three different methods of analysis to provided a means to distinguish the effects of dredging in the surface layer (<10 m) from the effects of deep sub-surface fluid withdrawal on both subsidence and on wetland-to-water habitat change.

The usefulness of landscape pattern metrics becomes apparent when they can be related to ecological processes. The distribution patterns of organisms across a landscape may provide an indicator of underlying variation in ecosystem processes. For example, the growth of the saltmarsh cordgrass *Spartina alterniflora* may be reduced in areas where tidal water does not drain sufficiently due to changes in oxidation-reduction potentials under flooded soil conditions (Howes et al. 1981). *Spartina* growth may therefore reflect the underlying drainage patterns of the marsh. The ecological effects of canal dredging on the tidal marsh are examined in Chapter 4 though a study of the distribution of *Spartina alterniflora*, the ribbed marsh mussel *Geukensia demissa demissa*, and the saltmarsh periwinkle *Littoraria irrorata*, sampled at the edges of dredged and natural channels in Barataria Bay. These three organisms are prevalent in the tidal marshes of Louisiana, and play important roles in the maintenance of marsh structure and the cycling of nitrogen, phosphorus, and carbon with the ecosystem. Their biomass distributions reflect gradients in environmental conditions, such as inundation and soil chemistry, between dredged and natural marshes.

The final step in this examination of the changing tidal marsh landscape is to link measurements of landscape structure to those of ecological function. The population structure of *G. demissa* is used in Chapter 5 to provide insights into how physical and structural processes affect the distribution of organisms. The structural and functional differences between natural and dredged channel systems are examined by comparing the age distribution of mussels at sites with divergent environmental conditions. This information provides a starting point for the development of a conceptual model of how dredged systems will function as mussel habitat in comparison to natural creeks.

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CHAPTER 2

LANDSCAPE EFFECTS OF CANAL DREDGING: THE QUANTIFICATION OF MORPHOLOGICAL DIFFERENCES BETWEEN NATURAL AND DREDGED CHANNEL NETWORKS

Introduction

The ecological importance of natural channel systems in tidal salt marshes has been well documented (see Mallin and Lewitus 2004 for an overview). Physical properties of tidal channels, such as edge slope and the presence of rivulets, have been shown to affect infaunal food availability and the rates of piscivorous predation, as well as nekton access to the marsh surface for foraging and refuge (McIvor and Odum 1988; Rozas et al. 1988). The presence of channel bends may affect vegetation patterns on the channel bank (Fischer et al. 2000), and vegetation type and density may in turn affect the refuge value of the marsh edge (Minello and Zimmerman 1983). The structural shape attributes of channel systems, such as the curvature of bends or the number of small branching channels, may significantly influence the overall ecological function of the channel system.

Artificial channel networks have been dredged in many tidal marshes for the purposes of drainage, marsh access, navigation, and to re-establish tidal flow to restored or created salt marshes. Dredged canal networks may function differently than natural creek systems; differences in environmental characteristics between natural and dredged systems may arise from how they were created and their resulting morphology. Natural tidal creek networks are formed by complex tidal flows over years to centuries, and have morphologies that are adjusted by hydraulic discharge and sediment erosion and deposition (Fagherazzi and Furbish 2001). Conversely, dredged canals are typically created in a few days using heavy machinery, and tend to be linear and of a uniform width and depth that reflects the type of machinery used (Davis 1973; Turner 1987). In the following text, the term “creek” refers to naturally formed systems, “canal” to artificially constructed systems, and “channel” refers to any conduit of tidal water, natural or artificial.

Dredged canals are typically straighter, deeper, and have steeper banks, higher mean dissolved oxygen concentrations, and lower mean salinities than natural creeks (Williams and Zedler 1999; Rozas and Reed 1994). These differences may have important implications for the function of channels as fish habitat because combinations of environmental variables such as dissolved oxygen, depth and salinity define the microhabitat characteristics of the marsh edge, which determine habitat use within and among species of common marsh-edge fishes (Baltz et al. 1993). Comparisons of fish use of constructed and natural channels have shown that mean species richness and mean densities are similar between channel types (Williams and Zedler 1999; Rozas 1992), but differences in abundance of individual species (Williams and Zedler 1999; Zedler et al. 1997) and population size-structure (Talley 2000) have been observed.

The first step towards determining differences in the function of natural and dredged channel systems is the development and application of suitable metrics for measuring the attributes of channel networks. Landscape metrics developed to characterize riparian stream networks, such as bifurcation ratio (R_b), sinuosity (R_s), and stream order, have been applied to tidal channels (Zeff 1999; Odum 1984). However, these metrics are difficult, if not impossible, to apply to dredged channel networks, which are composed of a series of interconnected through-flowing channels lacking a defined stem channel or distal ends, rather than the system of discrete branching networks typical of riparian streams and natural channels for which the metrics were developed. Methods applied in the field of landscape ecology for the analysis of habitat patches may be useful for describing the landscape changes associated with dredged channel networks. Habitat patches are defined by biologically significant discontinuities in environmental character states from their surroundings (Wiens 1976). Patch metrics characterize the spatial character and context of habitat patches (McGarigal et al. 2002). Patch metrics such as size, perimeter, and shape complexity have been shown to affect the distribution of organisms (e.g. Hovel and Lipcius 2001) and can be measured using aerial or satellite imagery and spatial analysis software. This method of characterizing patch attributes could be applied across channel network types regardless of the presence of a stem channel.

This study explores the application of patch metrics to the measurement of natural and dredged channel networks. It is important that the criteria used to measure landscape characteristics are relevant to the organisms of interest (Wiens et al. 1993). For the purposes of this study, the organisms of interest are the transient and resident nekton using the marsh channels for foraging and refuge. It is, therefore, assumed that the marsh-water interface is a border defining the channel system as a habitat patch. Several patch metrics were tested for their ability to discriminate between dredged and natural channel systems and examined for their applicability to tidal channel networks used as fish habitat. The ability to assess the structural shape of channel networks will aid in understanding the consequences of dredging on biological functions and provide guidance to marsh management plans that include the alteration, development, and restoration of channel networks.

Methods

Study Area

I used aerial photographs and LiDAR data (U.S. Army Corps of Engineers 2003) to locate five dredged canal networks and five natural creek networks in southeastern Barataria Bay, Louisiana (Figure 2.1). The study area is primarily a salt to brackish marsh dominated by *Spartina alterniflora*, with diurnal tides that range 30 cm; marsh inundation is controlled by tides, wind, and precipitation. Petroleum exploration and drilling programs in the study area began in the late 1930’s with the advent of equipment capable of drilling in aquatic environments, resulting in a coastal plain dominated by a massive network of man-made canals (Davis 1973). The dredged networks examined in this study are estimated to have been dredged between 1933 and 1969 based on the date the oil/gas well drill bit pierced the ground (SONRIS 2006).

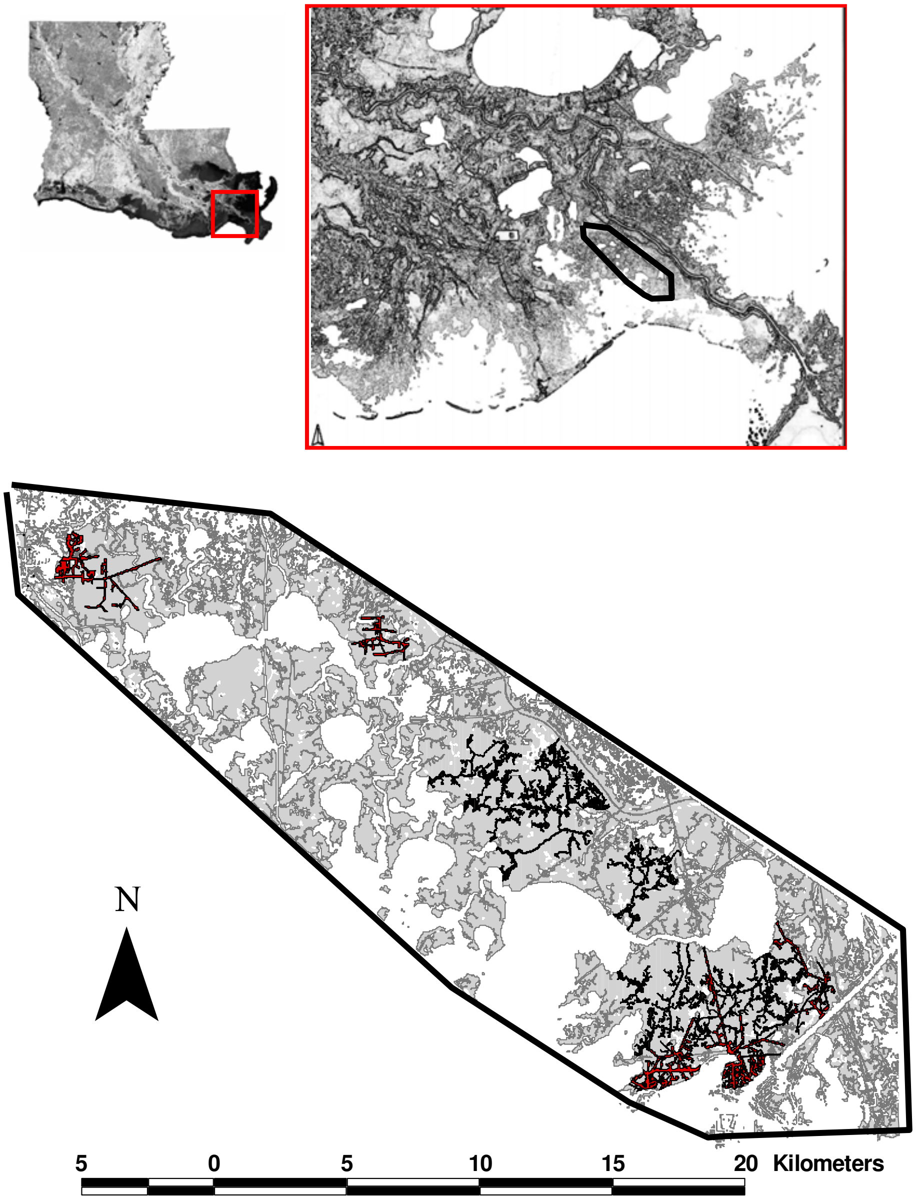


Figure 2.1. The location of the 10 channel networks analyzed for shape complexity in Barataria Bay, Louisiana.

Distinct drainage networks that had little overlap with adjacent channel systems were chosen from the study area bounded by the coordinates 90.0342^0 W, 29.5992^0 N, and 89.7386^0 W, and 29.4029^0 N. A channel was considered to terminate when the width was > 200 m or < 5 m. The selected channel networks were isolated from the landscape by re-classifying the LiDAR pixels with values equal to water within the channel using the raster recode tool in ERDAS IMAGINE®. The networks were then converted to vector shape files using the raster-to-vector utility software (Figures 2.2a and 2.2b). Three natural and 3 dredged systems were located in close proximity in a marsh area of approximately 3,900 ha (land and water). The other 4 systems were spaced over an area of approximately 12,000 ha (land and water), with a maximum of 34 km between systems.

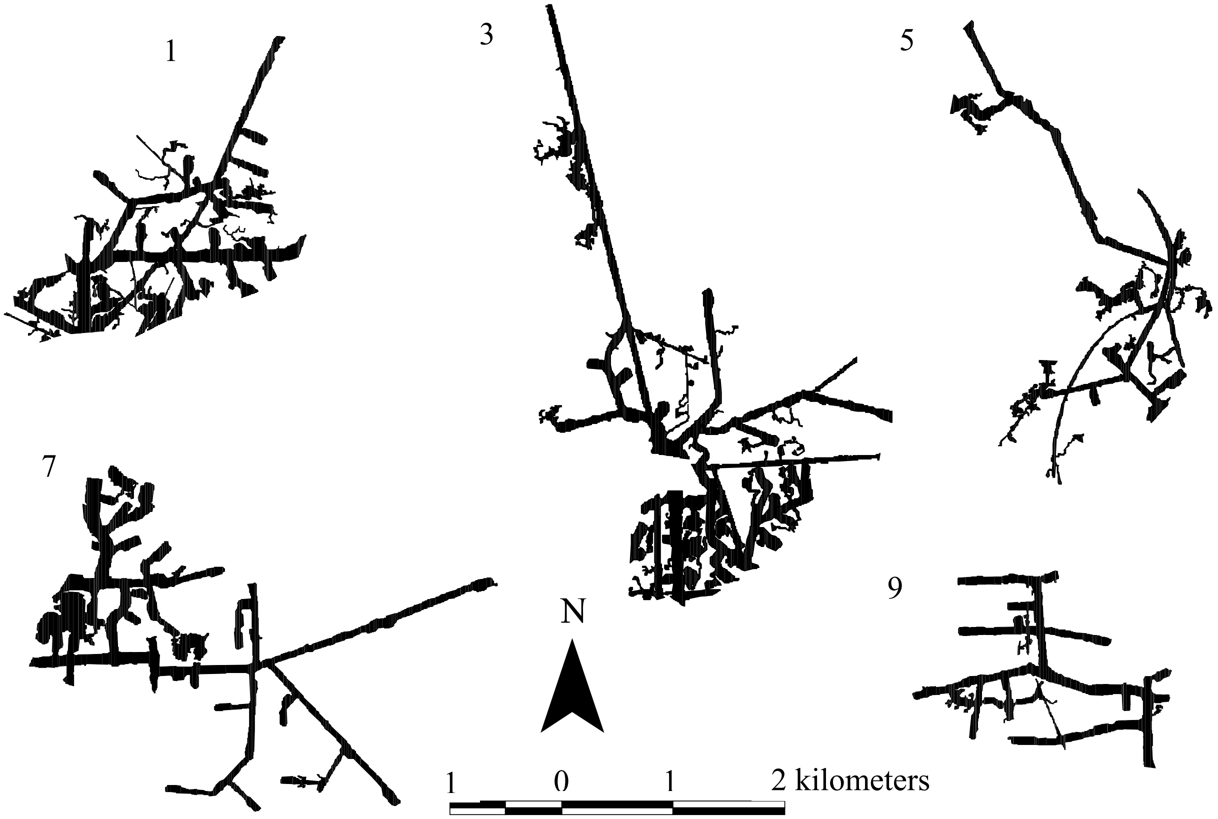


Figure 2.2a. Vector shapes developed for five dredged (D) channel systems in Barataria Bay. Systems are shown approximately to scale. Dredged systems are numbered with odd numbers 1, 3, 5, 7, 9.

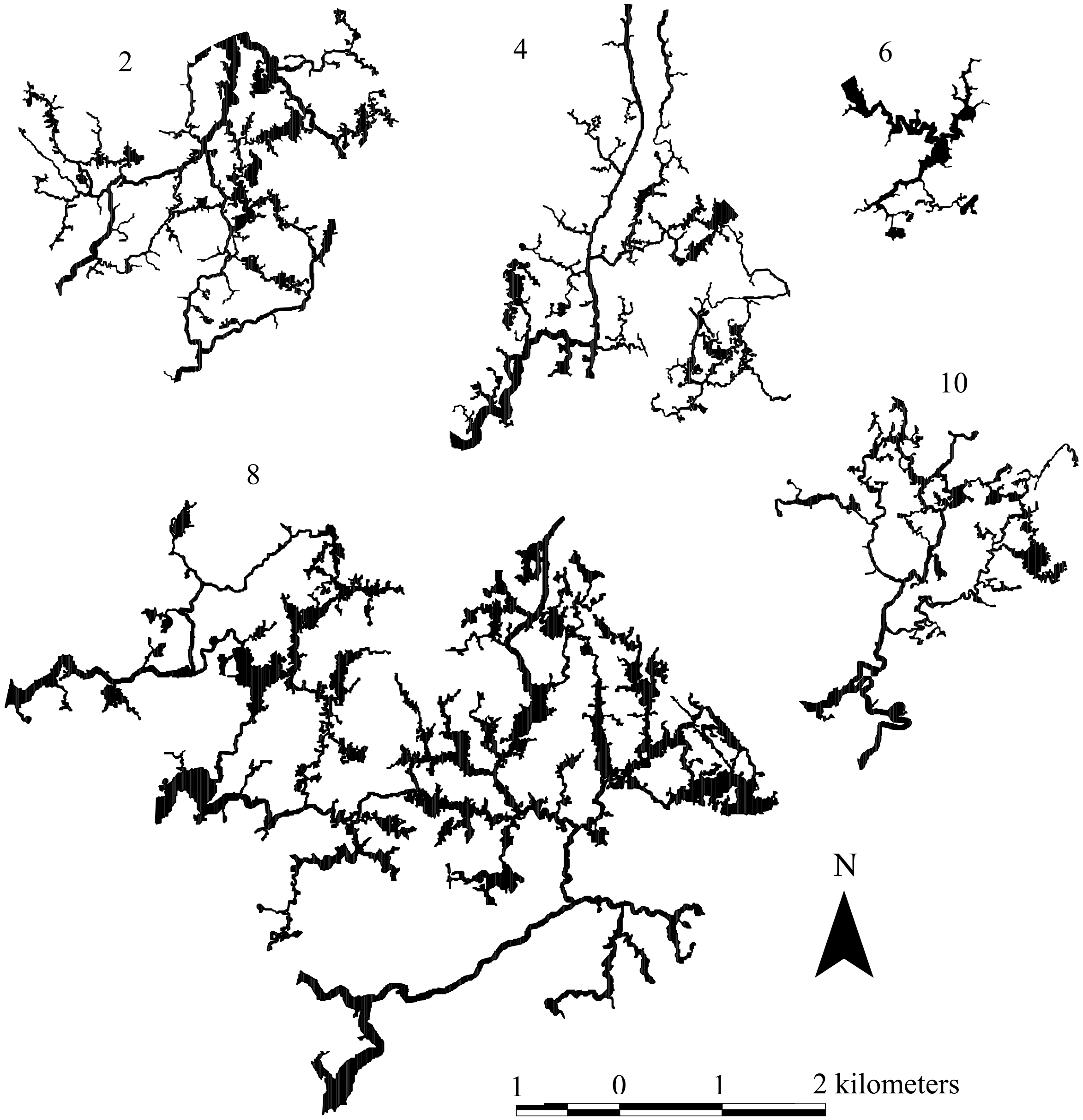


Figure 2.2b. Vector shapes developed for five natural (N) channel systems in Barataria Bay. Systems are shown approximately to scale. Natural systems are numbered with even numbers 2, 4, 6, 8, 10.

Channel Metrics

I used five metrics (described in Table 2.1) to examine shape differences between natural and dredged channel systems. Total edge (TE), class area (CA), perimeter-to-area ratio (PAR), shape index (SI) and patch fractal dimension (PFD) were measured using Patch Analyst, an extension for Environmental Systems Research Institute's (ESRI) ArcView 3.0 GIS that calculates the spatial metrics of polygon files (Elkie et al. 1999). PAR, SI, and PFD are all measures of shape complexity. PAR is the simplest measure of complexity, but its use is problematic because it varies with patch size. The use of SI can correct for this problem by standardizing the perimeter-to-area ratio to a Euclidean shape, in this case a circle. The value of SI is equal to 1 when the patch is a circle, and increases without limit as the patch becomes more complex. PFD relates the perimeter 'P' of a patch to the area by $P \approx \sqrt{A}^D$, where D is the fractal dimension and A = area. For simple Euclidean shapes, $P \approx \sqrt{A}$ and D=1, the dimension of a line. The perimeter becomes increasingly plane filling as the patch shape becomes more complex, so that $P \approx A$ with $D \rightarrow 2$ (Krummel et al. 1987). PAR, SI, and PFD were all used to compare how these metrics vary among the 10 different channel networks viewed as habitat patches. Differences between dredged and natural networks were tested using standard two-sample t-tests.

An estimate of spatial heterogeneity within networks was made using Patch Analyst by creating core areas bounded at 5 m, 10 m, 20 m, 30 m and 40 m distances from the *interior* edge of each network (Figure 2.3). This analysis provides information about how much of the within-channel habitat area is less than 5 m from the edge, and how much of the within-channel habitat area is greater than each specified distance from the edge. The proportion of the total within-channel area within the 5m buffer was used as an indirect measure of the relative area of small branching channels that are <10m wide.

Table 2.1. Descriptions of the metrics used to compare the shape complexity of natural and dredged channel systems.

		description	units	range
TE	Total Edge	P_{ij}= perimeter of patch ij including any internal holes	meters	TE>0 without limit
CA	Class Area	a_{ij}= area of habitat patch ij	Square meters	CA>0 without limit
PAR	Perimeter-Area Ratio	P_{ij}/ a_{ij} =The ratio of the patch perimeter to area	none	MPAR>0 without limit
SI	Shape Index	$0.28P_{ij}/ \sqrt{a_{ij}}$=Patch shape adjusted by a constant to adjust for circle standard	none	MSI>1 without limit MSI=1 when patch is circular
PFD	Patch Fractal Dimension	$2\ln P_{ij}/\ln a_{ij}$ = Patch fractal dimension	none	$1 \leq MPFD \leq 2$

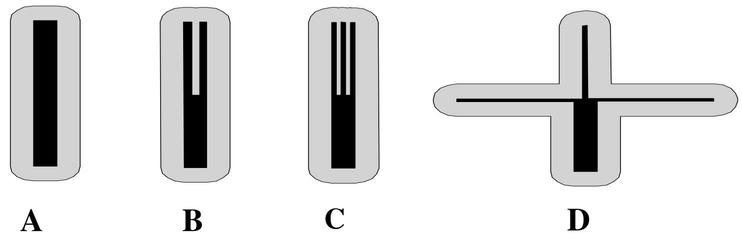
Figure 2.3. An example of core areas created using Patch Analyst. The amount of channel area within each distance category from the marsh edge was used as an indicator of spatial heterogeneity.

A drainage density index was computed for each network by determining the landscape area within a buffer distance of 100 m from the *outside* of the channel network edge and dividing by the total within-channel area of the network. Drainage density is typically measured as the length of channel per unit area of drainage basin, but in this case it was examined as unit area of channel per unit area of marsh that is within 100 m of the channel. Four model systems were created to visualize the relationship between the drainage density index and channel shape (Figure 2.4). The first system (A) was a simple rectangle of dimensions 1x10 units with a buffer distance of 1 unit. The channel complexity was increased in the second and third systems (B and C), by dividing the top half of the channel into two discrete channels separated by < 1 distance unit. The area of the landscape contained with the 1 unit buffer distance was the same in all three examples because the 1 unit buffer distance was greater than the distance between the divided channels. The perimeter and area of the system in the fourth example (D) was the same as in the third system (C); however a change in the configuration of the system increased the landscape area with the 1 unit buffer distance. The drainage density index of the Barataria channel networks was compared to that of the four model systems.

Results

The within-channel areas of the 5 dredged networks ranged from 62 ha to 169 ha, with a mean (± SE) of 115 ± 19 ha. The within-channel area of the 5 natural networks ranged from 29 ha to 345 ha, with a mean of 137 ± 54 ha. The mean within-channel area and total edge length did not differ significantly between dredged and natural networks (standard two-sample t-test; $p = 0.71$ and $p = 0.16$, respectively). Means for perimeter-to-area ratio, shape index, and patch fractal dimension were all significantly different between dredged and natural networks, although the degree of significance varied by the channel metric (t-test; PAR $p < 0.01$; SI p =

0.03, and PFD $p < 0.01$). The rankings of the ten networks based on each channel metric (Table 2.2) demonstrate the effect of metric choice on shape complexity values.



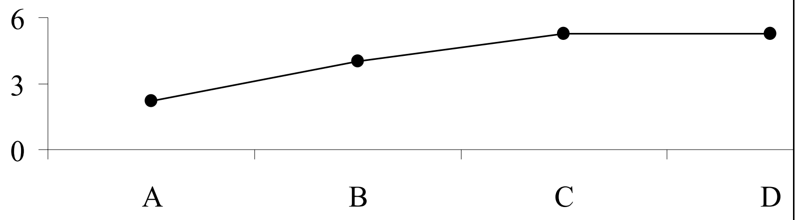
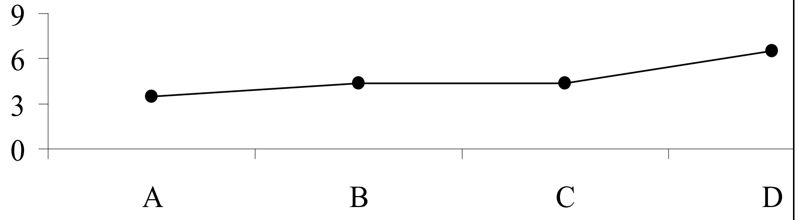
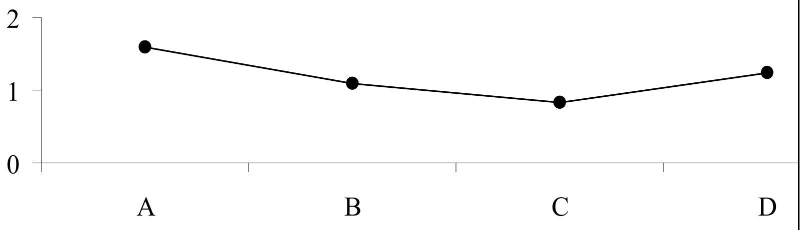
	A	**B**	**C**	**D**
Area	10	8	8	8
Perimeter	22	32	42	42
Buffer-area	35	35	35	52
Perimeter/ Area				
Buffer-area /Area				
Buffer-area/ Perimeter				

Figure2.4. A conceptual example of the relationship between system shape and perimeter, area, and buffer-area. The channel is shown in black above (A, B, C, and D), and a buffer area of a distance of 1 unit is shown in grey.

Table 2.2. The rank order of networks (D=dredged, N=natural) based on each of three measures of shape complexity (MPAR= perimeter to area ratio, MSI= mean shape index, and MPFD= patch fractal dimension). The indices vary notably in the ranking of network N6.

rank	PAR	*value*	SI	*value*	PFD	*value*
high	N4	*0.10*	N8	*35.15*	N4	*1.67*
↓	N2	*0.09*	N4	*28.73*	N2	*1.65*
	N10	*0.08*	N2	*28.28*	N8	*1.64*
	N6	*0.07*	N10	*21.19*	N10	*1.63*
	N8	*0.07*	**D3**	*17.78*	**D5**	*1.58*
	D5	*0.06*	**D5**	*14.90*	**D3**	*1.58*
	D3	*0.05*	**D1**	*13.97*	N6	*1.58*
	D1	*0.04*	**D7**	*11.90*	**D1**	*1.55*
	D9	*0.04*	N6	*10.59*	**D7**	*1.53*
low	**D7**	*0.03*	**D9**	*9.19*	**D9**	*1.52*

The spatial heterogeneity analysis (Figure 2.5) showed that natural networks had a significantly greater percentage of channel area that was < 5 m from the marsh edge than did dredged networks (standard two-sample t-test, $p < 0.01$), but had a significantly lower percent area than the dredged networks that was >10 m and > 20 m from the edge (t-test; $p < 0.01$ for both > 10 m and > 20 m measures). There was no significant difference in the percent of the channel area that was > 30 m or > 40 m from the edge between natural and dredged systems (t-test; $p = 0.08$ and $p = 0.50$, respectively). A mean (± SE) 33% ± 0.02% of the within-channel area of natural networks is < 5 m from the edge compared to 19% ± 0.01% for dredged networks. The channel area within 5 m of the channel edge increased as the size of the channel network increased for both natural and dredged networks. The ratio of area within 5 m to channel networks area, however, was greater for natural than dredged channel networks (Figure 2.6).

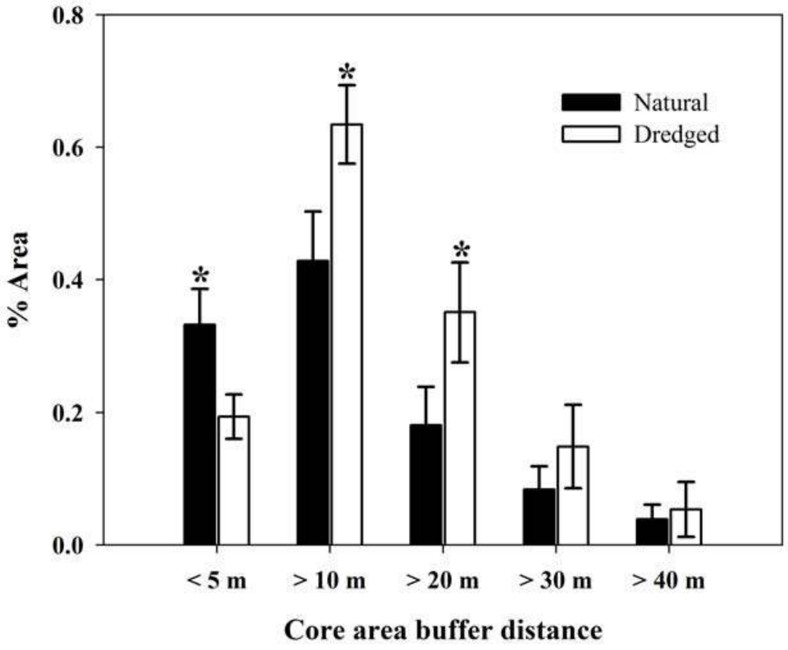


Figure 2.5. Results of the spatial heterogeneity analysis using measures of core area. Stars indicate significant differences between dredged and natural channel values, bars are ± 1 standard error.

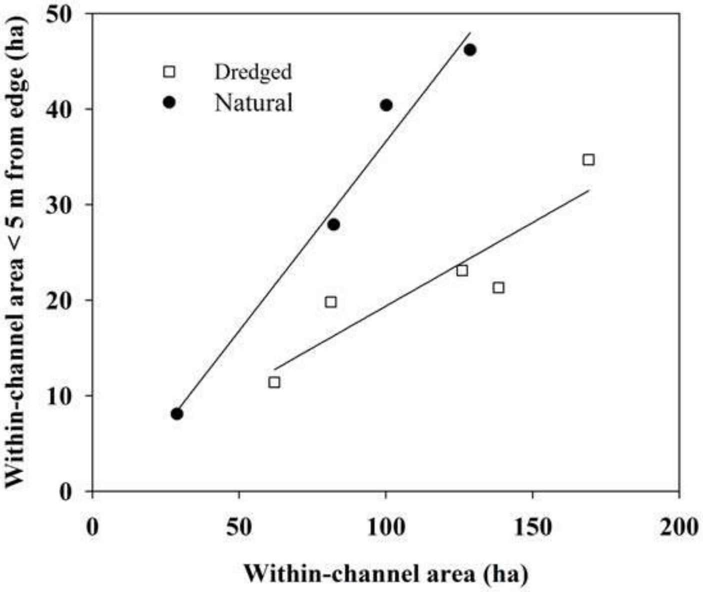


Figure 2.6. A comparison of the total within-channel area to the within-channel area that is less than 5 m from the edge. Data for natural channel system number 8 has been removed due to its larger size than the other systems. Linear regressions were fit to the data; natural channel networks (filled circles) were fit by the equation $y=0.3962x-3.0188$, $r^2=0.99$, dredged channels were fit by the equation $y=0.1749x+1.8839$, $r^2=0.91$.

Natural networks had a significantly higher mean (± SE) drainage density index than did dredged networks (5.13 ± 0.66 natural, 3.45 ± 0.45 dredged, $p < 0.01$). The values of the drainage density index for natural networks fell between the values for model systems D, B and C, while the values for dredged channels were more similar to the model system A (Figure 2.7).

The perimeter-to-area ratio was correlated with total edge for natural but not for dredged networks (natural $r^2 = 0.73$). The shape index was strongly linearly correlated with total channel network edge for both natural and dredged networks ($r^2 = 0.98$ for natural and $r^2 = 0.86$ for dredged) and the patch fractal dimension was strongly correlated with the total channel network edge for natural systems ($r^2 = 0.92$) and less strongly for dredged systems ($r^2 = 0.45$) (Figure 2.8). The perimeter-to-area ratio, shape index, and patch fractal dimension were strongly correlated with natural channel network area ($r^2 = 0.57$, $r^2 = 0.91$, and $r^2 = 0.83$ respectively). Only shape index was correlated with channel area in dredged networks ($r^2 = 0.46$). The relationship of channel metric to network area was strongest for the perimeter-to-area metric ($r^2 = 0.97$ for natural and $r^2 = 0.81$ for dredged), intermediate for the shape index metric ($r^2 = 0.91$ for natural and $r^2 = 0.46$ for dredged), and weakest for the patch fractal dimension metric ($r^2 = 0.83$ for natural and $r^2 < 1$ for dredged) (Figure 2.9).

Discussion

The mean area and total edge of the 5 dredged networks examined did not differ significantly from that of the 5 natural networks, but natural networks were significantly more complex in shape, contained more channel habitat < 5 m from the marsh edge, and had higher drainage area indices than dredged networks.

Shape complexity may influence species distributions by affecting habitat selection, the behavior of individual organisms within the habitat, population age structure, and the movement of organisms through the habitat. Ground-dwelling beetles, for example, are more likely to move into rectangular than square patches (Collinge and Palmer 2002), and grizzly bears show a preference for clearcuts with higher perimeter-to-edge ratios (Nielsen et al. 2004). The effects of patch shape on behaviors such as home range and foraging may be highly species dependent. The home range size of ruffed grouse increases as the irregularity of habitat patches increases (Fearer and Stauffer 2003), while the home range of mule deer is inversely correlated with the mean shape index (Kie et al. 2002). The age structure of a population may be affectedby habitat patch shape if the edge-to-interior ratio is positively related to adult or juvenile mortality. Major et al. (1999) hypothesize that a correlation between habitat size and/or shape and changes in breeding productivity and adult mortality of red-capped robins explains the significantly higher

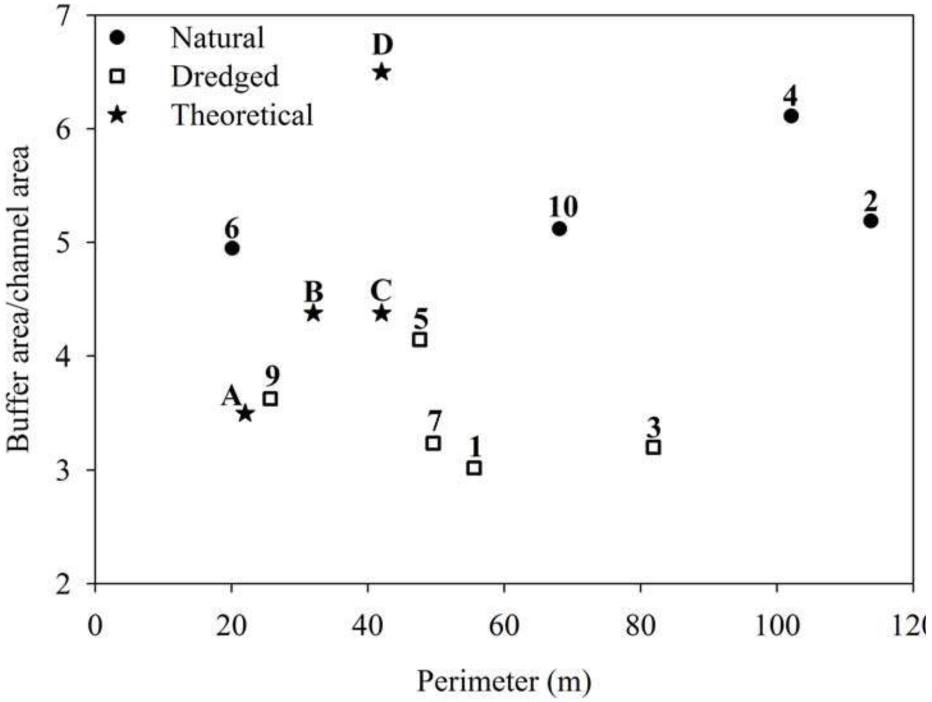


Figure 2.7. The ratio of buffer-area/channel area for the four model channels (A,B,C,D) and the five dredged and four natural channel systems as function of perimeter. Natural channel 8 was excluded due to its much larger size than the other 9 systems.

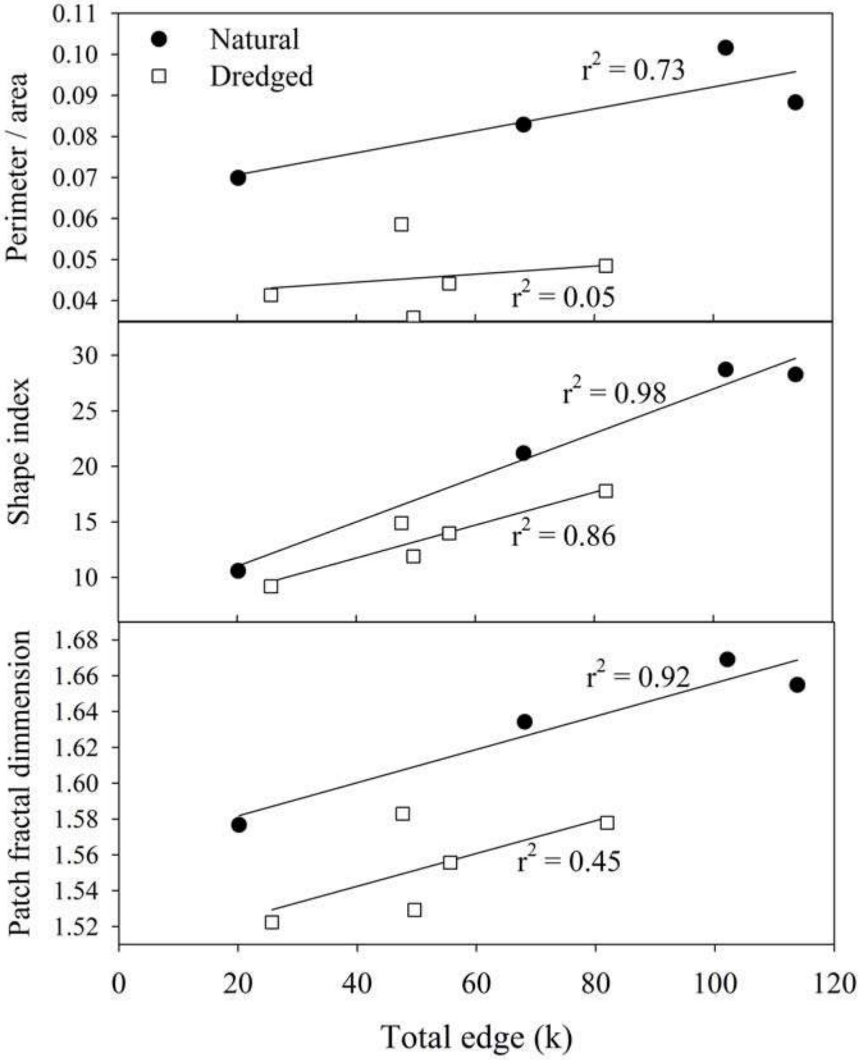


Figure 2.8. The relationships between total edge and perimeter/area (top), shape index (middle) and patch fractal dimension (bottom) are overall linear, although they differs between natural and dredged channels.

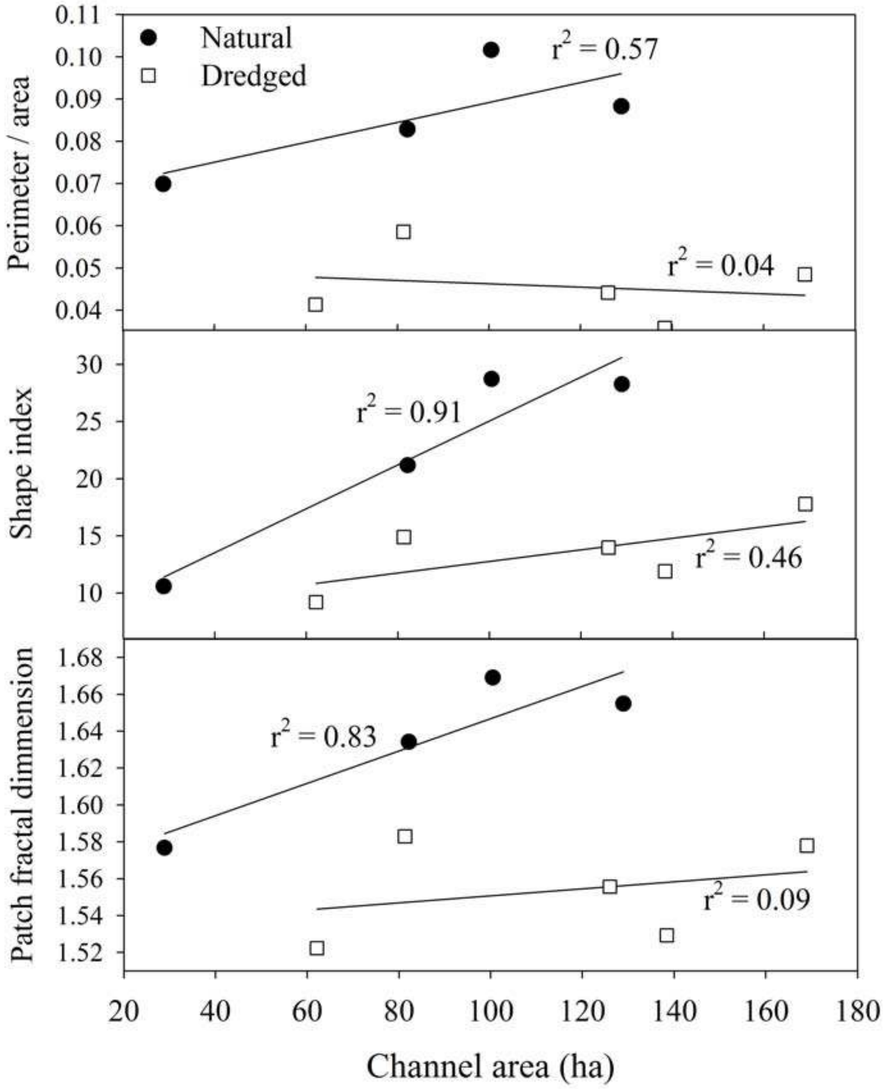


Figure 2.9. The relationships between each of the three shape metrics and channel area. The perimeter to area relationship is more sensitive to patch size than the shape index or patch fractal dimension.

proportions of yearling males found in linear remnant woodlands compared to those found in large non-linear remnants. Finally, the movement of organisms through a habitat patch may be affected by the connectivity and complexity of the patch. In a model of species invasion, for example, Cumming (2002) showed that habitats with more complex shapes allow organisms to invade them at a faster rate than in simple linear shaped systems.

Although habitat patch metrics have been widely applied to a variety of terrestrial systems, they have not been examined for tidal marsh channels in a similar manner. There is, however, evidence to support the hypothesis that nekton population dynamics are correlated with channel shape. Marsh edge, for example, has received much attention for its importance as fish habitat, specifically as a nursery area for juveniles that provides both food and refuge (Boesch and Turner 1984), and is positively correlated with the abundance and distribution of white shrimp (Webb and Kneib 2002). Because the amount of marsh edge increases linearly with shape complexity (Figure 2.8), more complex systems would be expected to provide more edge habitat. More complex channel systems may, therefore, support a greater abundance of edge dependent species. Channel shape may also affect environmental conditions within the water column that determine habitat suitability, such as current velocity, suspended sediment, and dissolved oxygen.

Natural networks have a greater degree of complexity than dredged networks relative to total channel network edge (Figure 2.8) and channel network area (Figure 2.9). This result suggests that dredged channels change the configuration of the edge. Although the relationship between edge complexity and habitat value for nekton has not been well studied, it may be an important consideration when linking nekton abundance to amount of marsh edge. The physical properties of the marsh edge, such as erosional versus depositional structure, slope, and vegetative structure, may affect food availability and predation (Lewis and Eby 2002; Cicchetti and Diaz 2000). Dredged canals lack meanders and tend to have a uniform edge type and slope. Natural channel networks, in contrast, typically have many meanders with both erosional and depositional edge types and steep and shallow banks. Nekton populations may be affected by channel shape on a microhabitat level if channels with less complex edges provide less variability in edge structure.

The results of the spatial heterogeneity analysis support the idea that channel shape affects microhabitat distribution. Compared to dredged networks, the natural networks had a significantly greater percent channel area that was < 5 m from the channel edge. This result can be attributed to the presence of small side channels that are < 10 m wide in natural channels that are rarely present in the dredged channels examined in this study. These smaller low-order channels provide corridors to the marsh surface during the flood tide (Rozas et al. 1988). Channel width and bank slope are also important determinates of species-habitat associations (Williams and Zedler 1999), and low-order side channels also may have steeper slopes (Williams and Zedler 1999), than higher order main channels. Natural channel systems may support more diverse species assemblages because of the presence of more varied channel habitat conditions.

Compared to the four test model systems (A, B, C and D; Figure 2.4), the drainage density index of natural networks fell between those of the forked systems B and C, and the spread out system D, while the values of dredged networks fell between B and C and the un-forked system A (Figure 2.7). The drainage density index is not correlated with the perimeter of the system, but is closely related to channel shape. Natural networks tend to have widely spaced branching channels more similar in geometry to system D than system C. This branching shape and the higher drainage density index suggests that a greater area of marsh may be accessible to nekton using the marsh surface during flood tide in natural channel systems, compared to dredged channel systems. This pattern has implications for how sediment accumulates on the marsh surface and the export of marsh materials to the open estuary.

Perimeter-to-area ratio, shape index, and patch fractal dimension all were useful for separating natural from dredged channels on the basis of shape complexity. The perimeter-area ratio is the easiest metric of shape complexity to measure and to conceptualize and gave statistically-meaningful results. The perimeter-to-area ratio, however, varies with size (for example, increasing the area of a circle decreases the perimeter-area ratio), and larger patches tend to be more complex than smaller ones. The shape index and patch fractal dimension metrics are less influenced by patch area (Figure 2.9) and may be more useful in systems where there is a large amount of variability in channel network size.

I assumed in this study that the tidal channel network is a habitat patch for the transient and resident nekton using the marsh channels for foraging and refuge, and that the marsh-water interface is a border defining the channel system as a habitat patch. These assumptions allows for the application of patch metrics to tidal networks. As shown in this study, this type of analysis provides a useful starting point for examining the ecological changes in marsh function that occur with channel network modification. The marsh-water interface in real tidal channels is not, however, a static border, and the qualities of the marsh edge are not homogenous. Further studies should examine the tidal channel network, not as a single habitat patch, but as a landscape with a mosaic of habitat types, such as marsh edge, deep channel, and pools. The application of patch metrics to tidal channel networks viewed as landscapes of habitat patches combined with biological sampling of species distributions within the network would provide useful information about the habitat values of natural tidal creeks versus dredged canals, and may also provide guidance to tidal marsh restoration and creation projects.

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CHAPTER 3

ELEVATION AND OPEN WATER IN TIDAL SALT MARSHES ADJACENT TO NATURAL CREEKS, AND DREDGED CANALS WITH AND WITHOUT FLUID WITHDRAWAL

Introduction

The movement of water through tidal marshes is largely regulated by channel bank elevation. Inorganic accretion rates are usually higher at the stream side than in the back marsh (Craft et al. 1993), forming a natural levee at the channel edge. Fewer tides overflow the marsh surface and less entrained sediment is deposited as natural levee elevation increases (French and Stoddart 1992), because the accretion of inorganic sediments on the marsh surface is, in general, a function of the relationship between the depth and period of tidal inundation and marsh surface elevation (Pethick 1981). Increasing or decreasing the elevation of the marsh edge interrupts this natural feedback mechanism mediated by the hydrology of marshes.

Tidal marsh hydrology can be altered by direct changes to the morphology of the channel network and the surface profile of the marsh edge, and possibly indirectly by subsurface fluid withdrawal. The process of constructing canals creates ‘spoil banks’ of dredged material along either side of the newly-built channel edge. In Louisiana these artificial levees, which may be 20 to 30 m wide and 2 m high when constructed, block sheet flow across the marsh surface. They may form semi-impoundment marshes in areas of intense dredging. Partially-impounded marshes have been shown to be flooded for more hours per month, have fewer but longer flooding and drying events, and reduced water exchange both above- and below ground (Swenson and Turner 1986). These changes to marsh hydrology will affect vegetative growth, perhaps lethally, and may reduce net organic deposition behind the spoil bank ridges.

Morton et al. (2005) postulate a way for the marsh surface to subside during oil and gas recovery operations and to result in wetland habitat conversion to open water. They suggest that some of Louisiana's dramatically high coastal wetland loss (111.2 km^2 yr^{-1} from 1955 to 1978; Baumann and Turner 1990) was the result of subsurface fluid withdrawal (as ground water, oil, or gas during mineral extraction operations). They propose that a fluid withdrawal > 1 to 2 km belowground leads to a minimum 1- to 20- cm subsidence in the surface layer, which then causes plant stress, and then open water formation. They also suggest that this fluid withdrawal induces geologic faulting which would also cause additional subsidence in the surface layer. The proposed consequences to marsh-to-water conversions from the induced subsidence would have the same result as from hydrologic changes created by impeding water flow in the surface layer by dredged channels. There are, therefore, two different mechanisms explaining the spatial coincidence of coastal landloss rates and canal density in subsections of the Louisiana coastal zone (Turner 1987; Turner and Rao 1990). One directly affects the surface hydrology to cause land loss, and the other arises from subsidence in deep layers (< 1000 m), leads to a soil subsidence in the surface layers, and plant stress.

In the following text, the term "creek" refers to naturally formed systems, "canal" to artificially constructed systems, and "channel" refers to any conduit of tidal water, natural or artificial. Here I test whether there is: 1) a difference in marsh elevation adjacent to dredged canals and natural creeks, 2) a difference in the amount of open water in marshes adjacent to dredged canals compared to in marshes with only natural creeks, 3) a difference in elevation in marsh areas with producing wells or with non-producing wells, compared to natural systems, and, 4) if there is more or less open water formed in marsh areas with producing wells or with non-producing wells, compared to natural marsh systems. The results from these tests provided a means to distinguish the effects of dredging in the surface layer (< 10 m) from the effects of deep sub-surface fluid withdrawal on both subsidence and on wetland-to-water habitat change. This analysis was greatly facilitated by using Light Detection and Ranging (LiDAR) elevation data in combination with GIS analysis tools in a novel and widely-applicable way to quantify differences in marsh elevation in different habitats.

Methods

Three different sampling methods were employed to examine the effects of dredged canals on the surface structure of the surrounding salt marsh. Estimates of marsh open water and elevation were made by: 1) sampling along 100 m transects perpendicular to dredged channels and natural channels, 2) sampling within 0.79 ha to 28.3 ha circular plots placed adjacent to producing wells, non-producing wells and natural channels, and, 3) examining marshes within a 100 m buffer area surrounding dredged and natural channels.

I used LiDAR data (U.S. Army Corps of Engineers 2003) and aerial photographs for southeastern Barataria Bay, Louisiana, to select and analyze elevation transects of marsh adjacent to dredged and natural channels (Figure 3.1). The flights for the LiDAR took place in 1999, and the aerial photographs were taken in 1998. This area has diurnal tides that range up to 30 cm and is primarily a salt to brackish marsh dominated by *Spartina alterniflora*. For the transect method of analysis, 30 natural sites and 30 sites with dredged channels were identified by using a random number generator to select numbered 10 ha grid cells placed over all channel systems within the study area. The locations of the transects were randomly chosen without a priori knowledge of the locations of active wells in the area. Four of the 30 well sites occurred within 1 km of an active well. Two 100 m transects were drawn perpendicular to the channel bank, one on either side of the channel, at each site. The transects started at the first pixel that

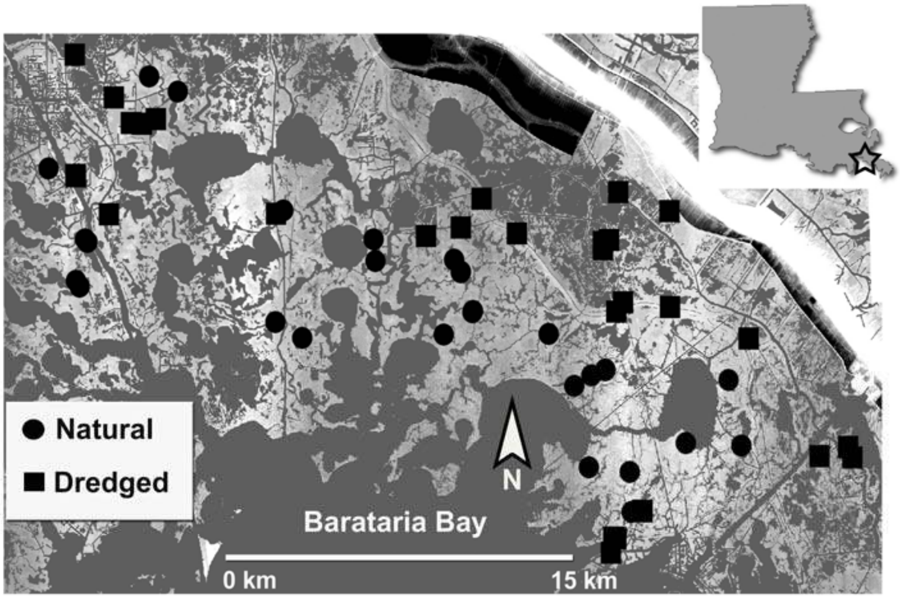
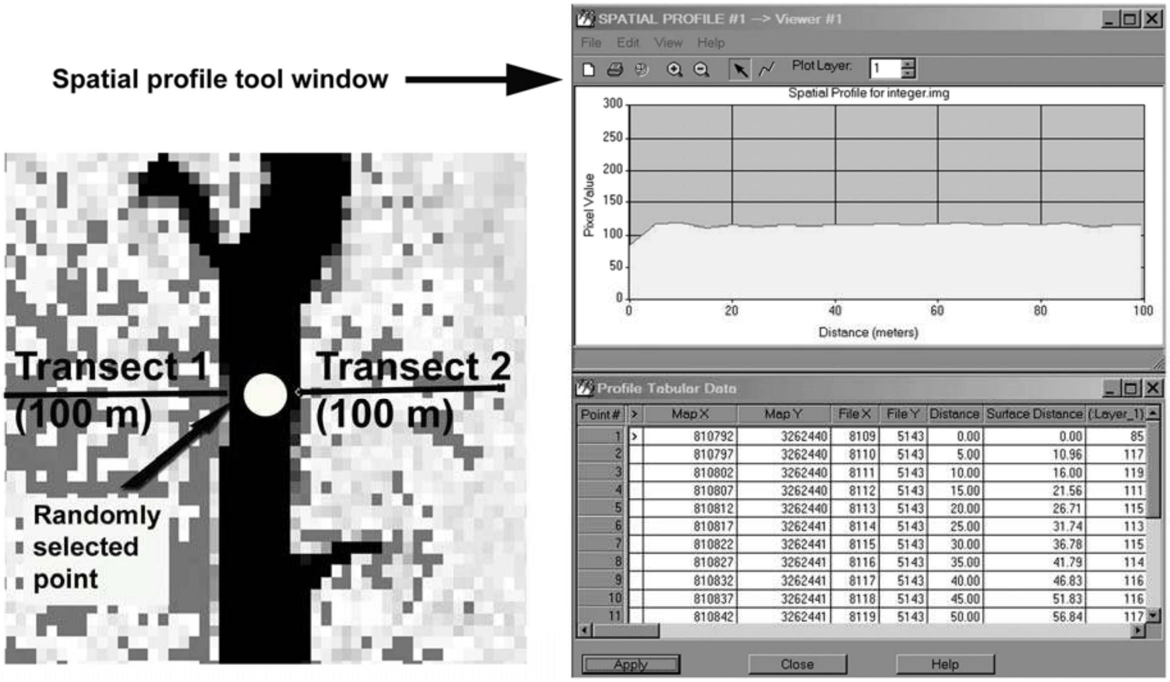


Figure 3.1. The location in Barataria Bay, Louisiana, of the 60 sites used for the transect analysis of marsh elevation.

did not have a value equal to that of the channel water, and extended 100 m in a straight line into the marsh at a 90° angle to the channel edge. The spatial profiler tool in ERDAS Imagine® was used to obtain the pixel values along each transect (Figure 3.2). The pixel resolution (horizontal) for the LiDAR data was 5 m, so the marsh elevation was effectively sampled every 5 m over the 100 m transect, with a vertical resolution of ± 0.07 m. The cumulative study area was approximately 57,000 ha, and the cumulative area of the pixels included in this part of the analysis was 6 ha. Mean elevations were compared between natural and dredged channels for each 5 m distance from the channel edge by Welch modified two-sample t-test. Twenty-two of the 120 transects crossed pixels with values equal to that of pixels of water because of the density of channels and frequency of open water "pond" areas within the marsh. Aerial photos of these 22 transects were examined and it was determined that, in all cases, these values represented areas where open water occurred. Water pixel values were removed from transects for the analysis of marsh elevation. The probability of water at each distance for natural and dredged marshes was calculated as the number of pixels that had water values divided by the number of replicates of each distance measure (60).



Point #	>	Map X	Map Y	File X	File Y	Distance	Surface Distance	(:Layer_1)
1	>	810792	3262440	8109	5143	0.00	0.00	85
2		810797	3262440	8110	5143	5.00	10.96	117
3		810802	3262440	8111	5143	10.00	16.00	119
4		810807	3262440	8112	5143	15.00	21.56	111
5		810812	3262440	8113	5143	20.00	26.71	115
6		810817	3262441	8114	5143	25.00	31.74	113
7		810822	3262441	8115	5143	30.00	36.78	115
8		810827	3262441	8116	5143	35.00	41.79	114
9		810832	3262441	8117	5143	40.00	46.83	116
10		810837	3262441	8118	5143	45.00	51.83	116
11		810842	3262441	8119	5143	50.00	56.84	117

Figure 3.2. A demonstration of how the spatial profile tool was used in ERDAS Imagine® to obtain the values of LiDAR data points along 100 m transects perpendicular to the channel edge.

The second method tested for the influence of fluid withdrawal on marsh elevation within 45 circular sampling sites (Figure 3.3). Thirty sites were located using the mineral recovery well data from the SONRIS database (Louisiana Department of Natural Resources). The 30 wells ranged in age from 0 to 54 years, had a modal drill date of 1957, and included 15 actively or formerly productive wells and 15 wells that were drilled but were never productive. All drilling sites were accessed through dredged channels. The average measured depth of the wells was 1.5 km. In addition, 15 sites from natural channel areas were randomly selected. At each of these 45

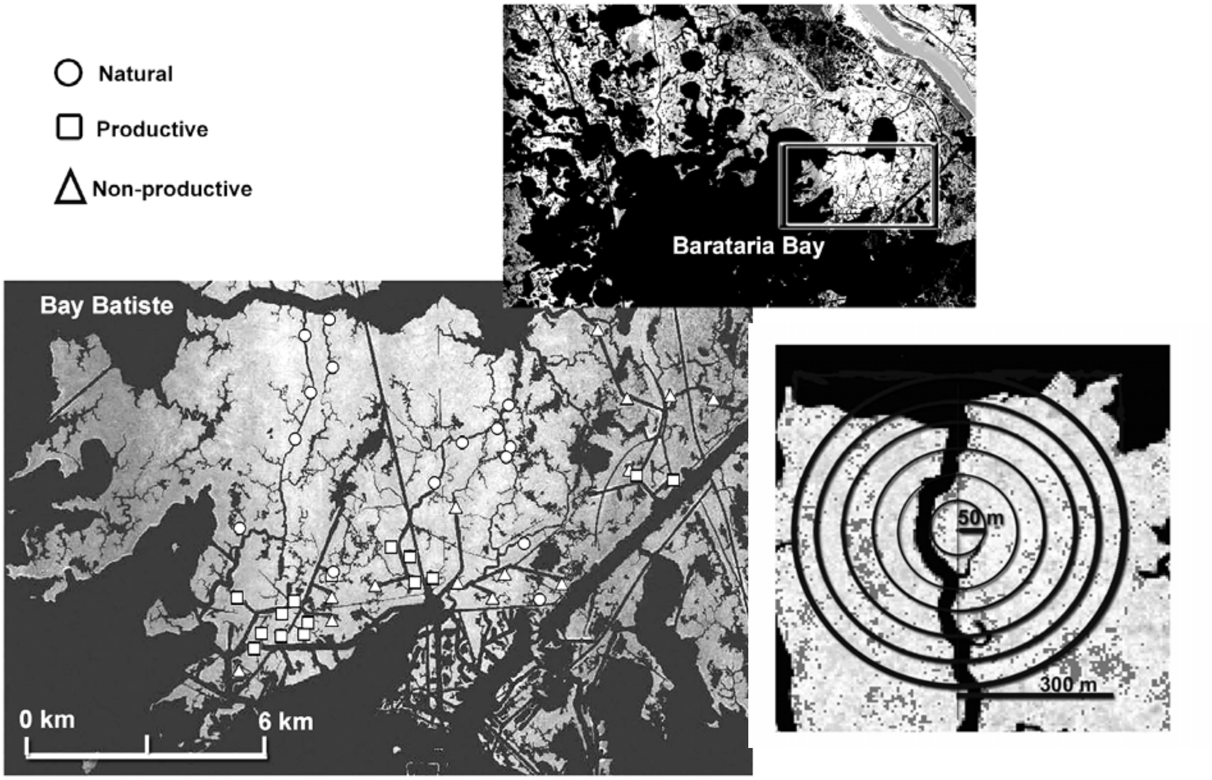


Figure 3.3. The location of the 15 natural sites, 15 productive well sites, and 15 non-productive well sites near Bay Batiste, in Barataria Bay, Louisiana. An example of the circular sample method showing the placement of the circles in relation to the channel is shown at the right.

sites, pixel values were obtained from the LiDAR data set for land elevation within a circle of 50 m radius adjacent to the channel edge by using ERDAS Imagine® to draw and subset the data within a circular area of interest (AOI) of 7,850 m^2. The mean elevation and percent of pixels with water values were calculated from the histograms for each circular subset of the LiDAR data. The effect of sample area size on the results was examined by increasing the radius of the sample circle to 100, 150, 200, 250, and 300 m and repeating the procedure for each of the natural creek, productive, and non-productive well sites.

The third method measured the elevation and percent water of the total area within 100 m of each of five dredged canal systems and of five natural creek systems. Buffers of 100 m from the channel edge were created around the channel networks and used to subset the LiDAR data (Figure 3.4). The mean elevations and percent of pixels with water values were calculated from the histograms of the subset data, and the dredged and natural marsh values were compared using standard t-tests. Two of the dredged canal buffer areas also contained active wells.

Note that the LiDAR measured elevations are in relation to the NAVD88 vertical datum, and tide height data is measured from mean low low water (MLLW) which is approximately 16.5 cm above the NAVD88 datum.

Results

Transects

Compared to natural marshes, the dredged marshes had a greater probability of water at all distances beyond 10 m from the channel edge (Figure 3.5a). The probability of water occurring in dredged marshes increased between 10 and 45 m from the marsh edge, where the probability of water in dredged marshes (16 %;) was eight times higher than the probability of water in natural marshes (2 %). When all distances were pooled, the overall probability of water in dredged marshes (12.3 %) was significantly higher than that of natural marshes (1.2 %; t-test, $p < 0.01$).

The overall mean elevations for natural and dredged marsh transects were 82 cm and 76 cm, respectively. Marsh elevation was significantly higher in dredged marshes than in natural marshes from 10 to 30 m from the channel edge, but there was no significant difference at distances greater than 30 m (Figure 3.5b). The results from the transect analyses indicate that there was more open water in marshes that were dredged than in natural marshes, that the marshes > 30 m from natural creeks had the same elevation as the marshes that were not dredged, and that a spoil bank significantly increases the height of the marsh within 30 m of the channel edge.

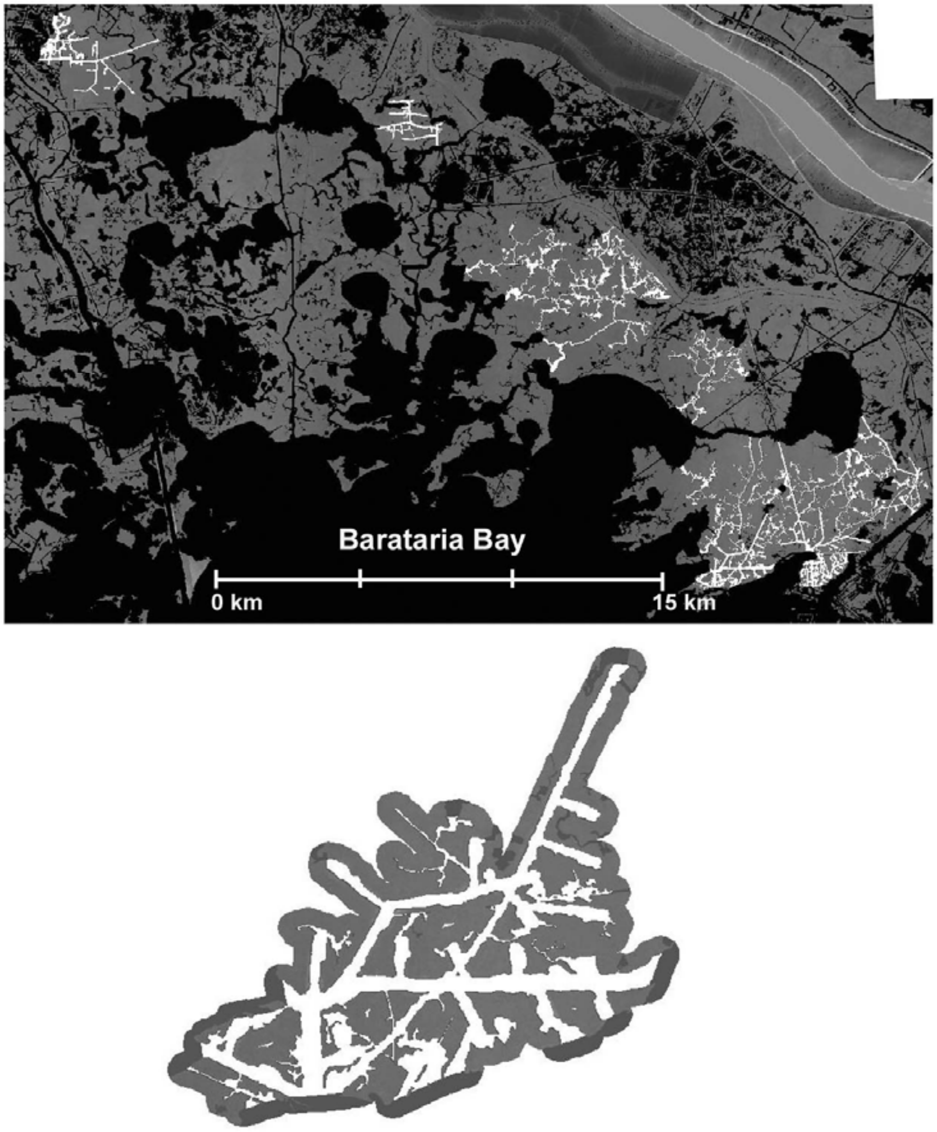


Figure 3.4. The location of the 5 natural and 5 dredged channel networks used for the buffer analysis of elevation. An example showing a 100 m buffer around a dredged system is shown below.

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Circles

The radial distance of the circle used to sample the LiDAR data affected the results for both the mean elevation and the percent water. The mean elevation of the 50 m radius circle samples of the marsh adjacent to natural creeks was significantly higher than that found in marsh adjacent to non-producing wells, but was not significantly different from producing wells (ANOVA, $F_{2,41} = 5.09$, $p = 0.01$, natural $\bar{x} = 90$ cm, non-producing wells $\bar{x} = 58$ cm, producing wells $\bar{x} = 70$ cm). The difference in elevation between natural marsh and non-productive wells sites was no longer significant at radial distances greater than 50 m. There were no significant differences among site types in the mean percent of water found in the 50 m radius circle samples. The difference in percent water between natural channels and productive wells and between natural channels and non-productive wells became significant at radial distances greater than 150 m (ANOVA 150 m: $F_{2,41} = 6.49$, $p < 0.01$; 200 m: $F_{2,41} = 9.46$, $p < 0.01$; 250 m: $F_{2,41} = 11.99$, $p < 0.01$; 300 m: $F_{2,41} = 12.28$, $p < 0.01$) (Figure 3.6). The percent water in natural sites increased from a mean of 1.5 % over the 50 m radius sample to 19 % over the 300 m radius sample, from 0.2 % to 33 % for non-productive wells sites, and from 1.3 % to 38 % for productive well sites.

The disparity in results obtained using a 50 m radius circle sample versus using a 300 m radius to sample the marsh is illustrated in Figure 3.7. The results of the circular sample method did not show a 'spoil bank effect' for marsh area adjacent to channels dredged for well access unlike results from the transect method already described. The results of the circular sample method also demonstrated that as the sample area increased to include a greater amount of marsh area beyond the channel edge, there was a greater proportion of open water.

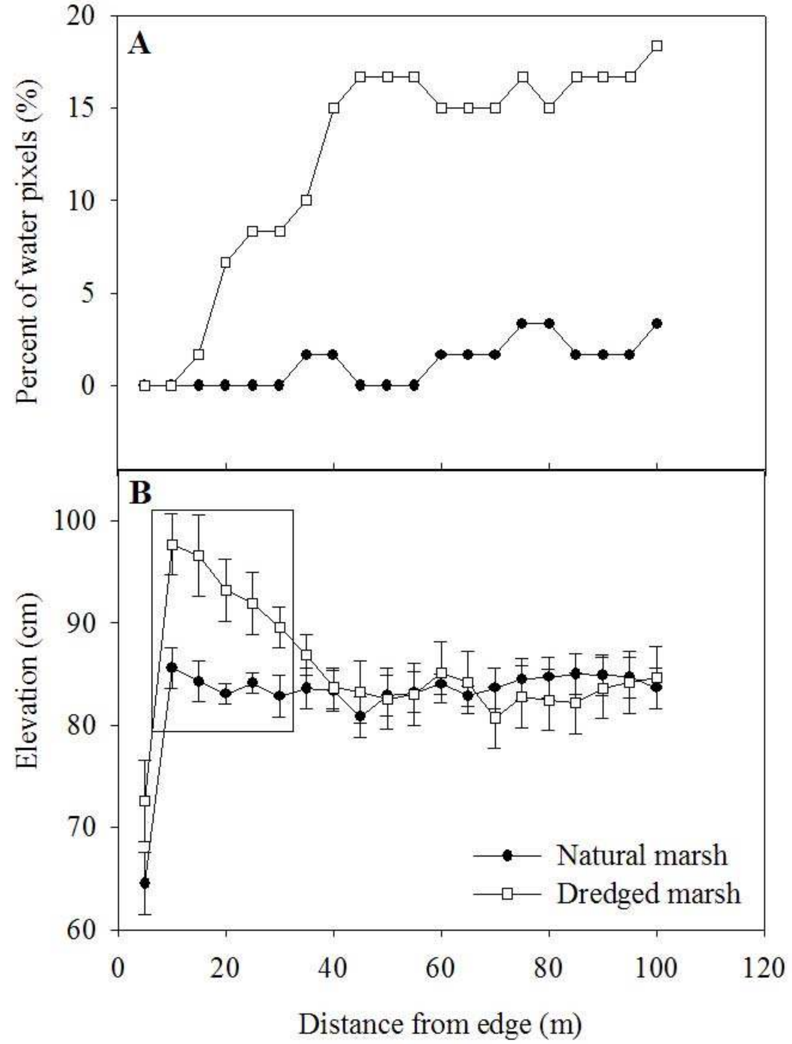

Figure 3.5. A. The percent of pixels that had water values over distance from the channel edge. Natural and dredged marsh both had 0 % water at 5 and 10 m. B. The elevation of dredged marshes over distance from the channel edge. Values in the box are significantly different by t-test. See table 3.1 for p-values. The bars represent ± 1standard error.

Buffers

There was no significant difference in marsh elevation between marshes within 100 m of dredged channel networks and marshes within 100 m of natural creek networks (natural $\bar{x}$ = 84 cm, dredged $\bar{x}$ = 85 cm). Dredged marsh sites, however, had nearly twice as many water pixels as natural marsh sites (t-test, $p = 0.01$, natural $\bar{x}$ = 9 %, dredged $\bar{x}$ = 17 %).

The results of the three sample methods (100 m transect, 100 m buffer area, and circular samples) all demonstrate that there was significantly more open water in areas of marsh that have been disturbed by channel dredging, but that marsh elevation did not appear to be affected, except where spoil bank occured (at the marsh edge; Table 3.1). The marsh elevation and open water at producing well sites did not differ from that at non-producing well sites, or from transect and buffer results for dredged channels.

Discussion

The land elevation at the land-water interface of dredged canals showed an increase for up to 30 m from the marsh-water interface, but the elevation of interior marsh did not differ from that found in natural marshes. Literature values for spoil bank dimensions range from 20 to 30.5 m wide and about 0.5 to 1 m high (Monte 1978; Swenson and Turner 1987; Abernethy and Gosselink 1988). The elevated area of marsh adjacent to dredged canals examined by the transect method extended approximately 30 m from the edge and the mean highest elevation of 0.98 m occurred at 10 m from the edge of dredged canals. The natural channel levees in this study had an approximate width of 10 m and maximum height of 0.85 m. A difference in height of 13 cm between dredged and natural marsh edge is noteworthy in a microtidal setting where the tidal amplitude is 30 cm, and sea level rise is 0.2 cm/year.

No significant difference was found in marsh elevation between natural and dredged

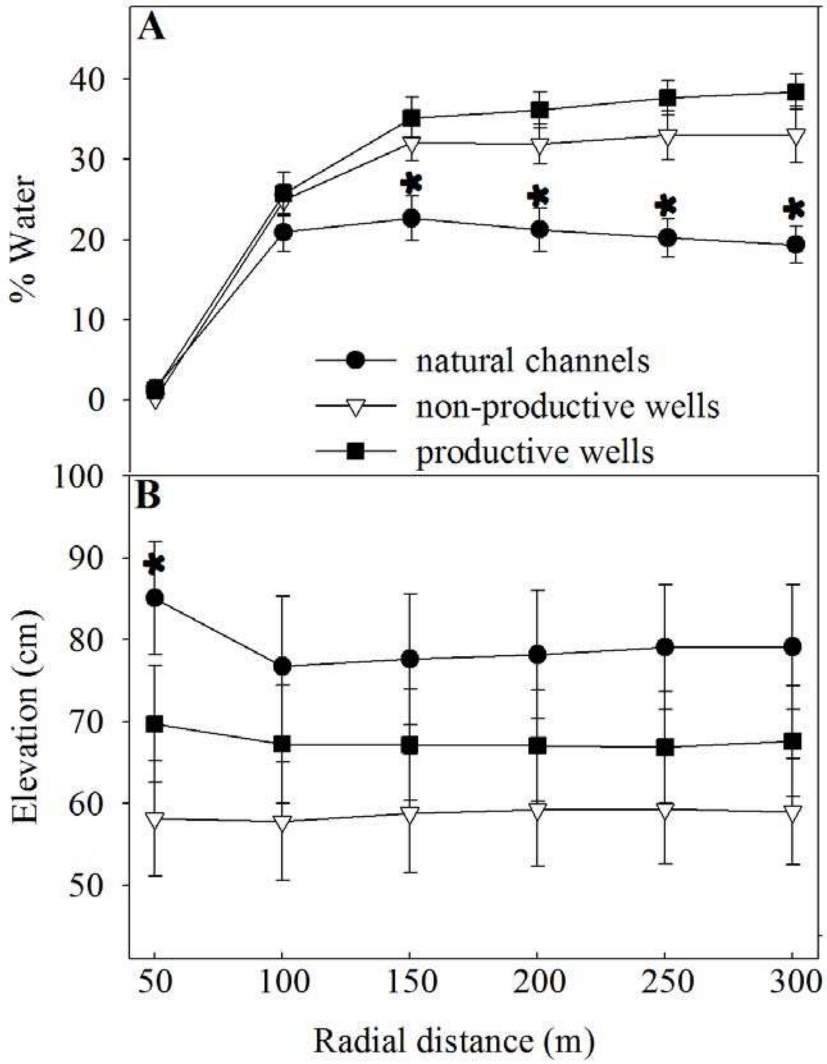

Figure 3.6. The results from the circular sampling scheme. A. The percentage of open water with increasing circle sample size. Stars indicate results that were significantly different from the other results at that radial distance. B. The mean elevation of marsh sample with increasing circle sample size. A significant difference was only found for natural channels at a radial distance of 50 m, indicated by a star. The bars are ± 1 standard error.

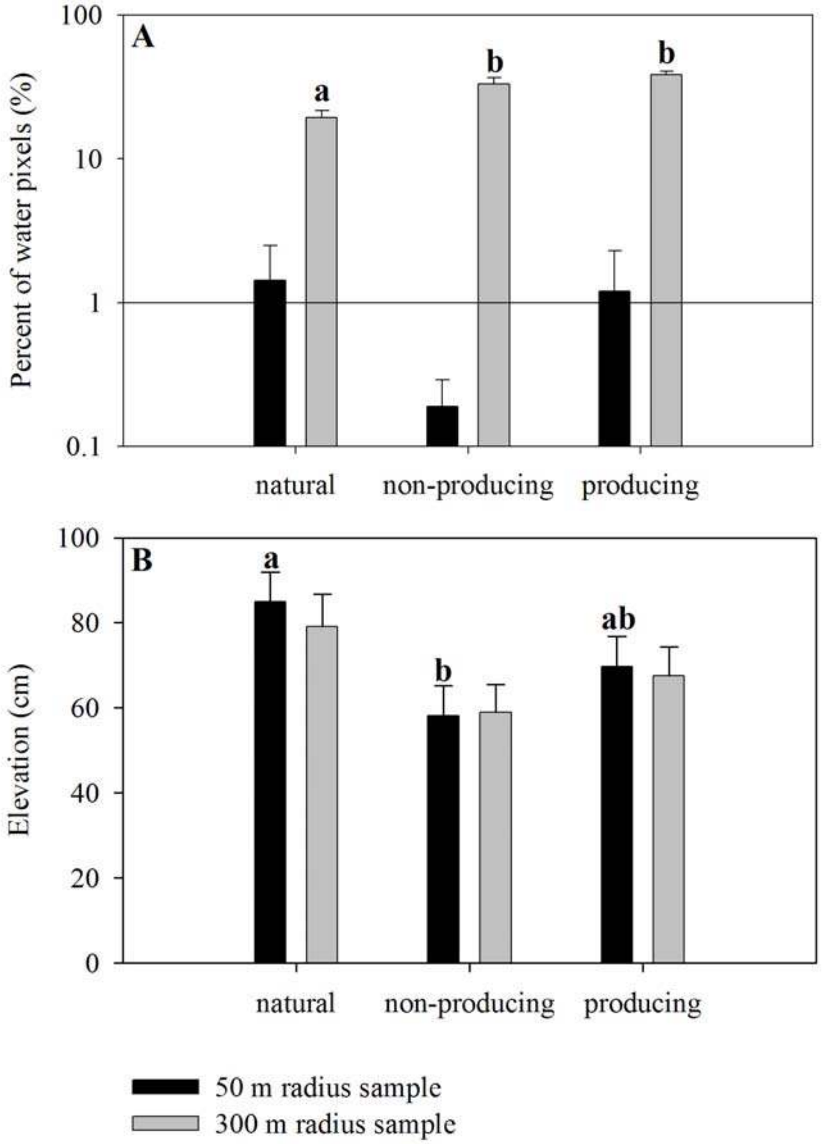

Figure 3.7. A. The results for percent of open water pixels obtained using a sample circle with a 50 m radius versus a 300 m radius. Note that the y- axis is a log scale. B. The results for mean elevation obtained using a circle with a 50 m radius versus a 300 m radius. Letters represent significant differences between site types and bars are ± 1 standard error.

Table 3.1. A summary of the marsh elevation and the percentage of water data obtained by three different sampling strategies: one hundred and twenty 100 m transects at 60 sites, ten buffered channel networks, and forty-five 100 m diameter circles.

	100 m Transects	**100 m diameter Buffered Networks**
A. Mean marsh elevation		
A.1 Natural marshes	83 cm	84 cm
A.2 Dredged marshes	76 cm	85 cm
Statistically significantly difference?	Yes, @ 5 to 30 m No @ > 30 m	No
B. Percentage open water		
B. 1. Natural marshes	1 %	9 %
B. 2. Dredged marshes	12 %	17 %
Statistically significantly difference?	Yes	No

	Circular Samples	
	50 m radius	**100 m radius**
C. Mean marsh elevation		
C.1 Natural marshes	90 cm	79 cm
C.2. Non-producing well sites	53 cm	59 cm
C.3 Producing well sites	70 cm	68 cm
Statistically significantly difference?	Natural > non-Producing; Producing Not different from non-producing	No
D. Percentage open water		
D.1. Natural marshes	1.5 %	19 %
D.2. Non-producing well sites	0.2 %	33 %
D.2. Producing well sites	1.3 %	38 %
Statistically significantly difference?	No	Natural < Non-producing and Producing; Producing Not different from non-producing

channel systems within a 100 m buffer of the channel system. The elevated spoil bank area found by the transect method extended 30 m from the edge, so the buffer analysis may have included enough non-spoil bank area to minimize the effect of spoil bank on the overall elevation measurement. It is also important to note that the transect method measured elevation at points on individual channels at locations widely distributed across the marsh, whereas the buffer method sampled the marsh around entire channel systems. The spoil bank may have eroded from these particular systems; their approximate age, estimated from the drilling dates, was 38 to 74 years. Additionally, the dredged material tends to spread out over the marsh after deposition, and has been observed to cause elevation changes over an area five to six times as wide as the canal itself (Monte 1978). A combination of spreading near the spoil bank and marsh subsidence could cause the elevation of the dredged marsh end up at an elevation similar to that of an undisturbed marsh.

Oilfields producing from shallow horizons may contribute to subsidence of the marsh (Walker et al. 1987; Morton et al. 2006). Because oil wells are usually sited within dredged canals, it is necessary to distinguish between the effects of fluid withdrawal at wells from the effects caused by the canals and spoil banks. The circular sample method was used to test for evidence of decreased surface elevation due to subsurface fluid withdrawal over distances from 50 m to 300 m from the well site. Only natural and non-producing well sites were significantly different in mean elevation at a radius of 50 m (Figure 3.6), but no differences were found at radial distances greater than 50 m. It is, therefore, unlikely that fluid withdrawal changed marsh elevation at these sites. It is interesting that the dredged marshes with non-producing wells had the lowest mean elevation, compared to dredged marshes with producing wells and natural marshes. One explanation is that the spoil bank eroded after these dredged sites were abandoned, while actively producing well sites are periodically re-dredged and the newly deposited materials 'leaked' into the surrounding marshes.

Both the transect and buffer methods demonstrated that there was significantly more open water in dredged than in natural marshes. The percentage of open water 35 m into the marsh was 15.7 % on transects in dredged marshes and 1.7 % on transects in natural marshes. The difference, 14 %, compares well with the 18.3 % of the total wetland area that became open water from 1955/6 to 1978 when land loss was at its peak (Baumann and Turner 1990; Turner et al. 2007), and the estimated 22 % of the land lost from 1956 to 2005 (Barras 2006). These comparisons suggest that the results from this analysis of the 57,000 ha study area are broadly applicable to the entire coast.

The marsh around dredged canals appears to be more broken up, but not lower in elevation. Turner and Rao (1990) found that the area and number of new ponds formed in Louisiana tidal marshes was positively related to canal area, and that if new small ponds formed, then it was more likely that larger ponds would also form, so that the marsh broke up internally, rather than at the edge. The results of my study are consistent with the pattern of marsh loss which would result from the internal break up of the marsh into open water. Two proposed mechanisms explaining the marsh-to-water conversion of interior marshes are 1) the marsh surface behind spoil banks subsides because of a reduced sedimentation on flooding tides (Cahoon and Turner 1987), and, 2) through the loss of vegetation that occurs when changes in marsh hydrology cause extended flooding and drying events that severely stress marsh plants (Turner 1987). Spoil banks have been shown to affect tidal inundation patterns by decreasing the number of flood events, increasing the duration of flooding and drying events, and reducing water exchange above- and below ground (Swenson and Turner 1987). Extended flooding can inhibit the growth and increase the mortality of marsh plants. Extended drying periods increase soil salinity which may inhibit plant growth and cause catastrophic marsh 'die-back' (Howes et al. 1981, Mendelssohn et al. 1981). More than 90 % of the soil volume is composed of organic matter and water in healthy salt marshes (Turner et al. 2001), so that once vegetative production is decreased or lost, the marsh surface may quickly subside and convert to open water. The elevation of the marsh adjacent to spoil banks in this study was not significantly lower than that of the marsh adjacent to natural levees, and there was no correlation between marsh elevation and the probability that a pixel was open water. If marsh elevation is maintained through the deposition of over-bank sedimentation, then marshes with spoil banks blocking over-marsh flow should show a decrease in elevation as subsidence continues and sediment input is reduced. On the other hand, if marsh elevation is maintained by organic matter input from vegetative productivity, then the conversion of marsh surface to open water may occur rapidly as vegetation dies off due to the increased hydrologic stress, and occur without an intermediate phase of decreased elevation.

The marshes of natural, non-producing well, and producing well sites all showed an increase of about 20 % in the percent of water pixels found when the circular sample size was increased from a radial distance of 50 m to 100 m. The 50 m samples were proportionally more filled with natural levee or spoil bank area than the 100 m samples; the smaller radius size was not effective at measuring the amount of open water beyond the spoil bank. The position of the circle at 100 m often covered spoil bank area on both sides of the channel (Figure 3.3) so even at 100 m, there still appears to be an effect of the presence of spoil bank on the results. Increasing the radial distance to 150 m further increased the percent water found at non-productive and productive well sites, but did not at natural sites. At 150 m, the difference in percent water between natural sites and non-producing and producing well sites becomes statistically significant, and the results of the circular sample method parallel the findings of the transect and buffer sampling methods that there is more open water associated with dredged marsh sites than with natural marshes.

The results of this study indicate that: 1) there wass no difference in the mean elevation of marsh > 30 m from the marsh edge of dredged canals and natural channels, 2) there was a difference in the amount of open water in marshes adjacent to dredged canals compared to marshes with only natural channels, 3) there was no difference in elevation in marsh areas with producing wells or with non-producing wells, compared to natural systems, and, 4) there was more open water formed in marsh areas near producing wells or non-producing wells, compared to natural marsh systems. There was no evidence found in this study to support the theory that deep sub-surface fluid withdrawal affects surface elevation. The formation of open water areas within the marsh was near dredged canals, but the mechanism of marsh-to-water conversion does not appear to involve an intermediate decrease in elevation. The evidence that marsh loss occurs though a break up of interior marsh to form open water ponds, rather than through edge erosion, can be seen in aerial photos and through the results of this and other studies, but the mechanism and time scale of the marsh to pond transition needs further analysis. A time series of LiDAR data would be useful in further exploring associations between elevation changes and marsh loss, and could be combined with "on the ground" studies of plant productivity to better understand marsh-to-water conversions.

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CHAPTER 4

SALT MARSH CORD GRASS, *SPARTINA ALTERNIFLORA*, RIBBED MARSH MUSSEL, *GEUKENSIA DEMISSA DEMISSA*, AND SALT MARSH PERIWINKLE, *LITTORARIA IRRORATA*, DISTRIBUTION IN NATURAL AND DREDGED TIDAL SALT MARSHES OF LOUISIANA

Introduction

Comparisons of the tidal salt marsh edge to the inner marsh, and to shallow non-vegetated areas (e.g., ponds, channels, bays), have shown that the marsh edge is preferentially selected for by decapod crustaceans and resident, transient, and commercially important fishes (Baltz et al. 1993; Peterson and Turner 1994; Rozas and Zimmerman 2000; Stunz et al. 2002), and the amount of marsh edge appears to be positively correlated with densities of natant decapods (Minello et al. 1994; Webb and Kneib 2002). While it is generally accepted that the marsh edge serves as a nursery habitat for many nekton species by providing food and refuge, it is important to consider that marsh edge is not uniform and can include a variety of habitat types and environmental conditions. For example, differences in infaunal food availability and predation pressure have been found between erosional and depositional edges (McIvor and Odum 1988), and fish community structure varies between rivulet and channel marsh edge (Hettler 1989).

The dredging of canals creates tidal marsh edge. The most intensely dredged coastal wetland landscape is in Louisiana, which had 45,866 ha of canals in 1990 (Britsch and Dunbar 1993). No estimate of total increase in marsh edge could be found for these wetlands, but the total addition of edge by canal construction may be roughly estimated by dividing the area of dredged canals by the average canal width, estimated to be 50 m (Chapter 2), to obtain a canal length of 9,173 km, which is then multiplied by two to account for both banks of the channel. This calculation yields an estimate of 18,346 km of edge added to the coastal Louisiana marshes through canal construction. In the following text, the term “creek” refers to naturally formed systems, “canal” to artificially constructed systems, and “channel” refers to any conduit of tidal water, natural or artificial.

It has been hypothesized that, because the marsh edge is essential habitat for many species, habitat value and fisheries landings may actually increase as marsh edge is increased, but eventually decline as open water replaces broken marsh (Chesney et al. 2000). It is possible, however, that the functioning of the marsh edge may be affected by the method of development (i.e., natural vs. anthropogenic), and by the amount of unbroken marsh area with which it is associated. Morphological characteristics that differ between natural and created channels (e.g., width, maximum depth and bank profile) may influence fish assemblage composition (Williams and Zedler 1999). Dredged canals are usually bordered by parallel spoil banks formed by the deposition of dredge material. The soil properties of this dredged sediment, such as bulk density, grain size, and nutrient content, may also differ from that of undisturbed sediment at unmodified marsh edges.

The distribution of plant and animal species on the *marsh surface* near the channel may also be affected by channel modifications. Tidal creek morphology has been linked to the spatial distribution of vegetation (Fischer et al. 2000; Sanderson et al. 2000), and both vegetation type and density have primary roles in structuring the marsh surface habitat. For example, the ribbed marsh mussel, *Geukensia demissa,* and the salt marsh periwinkle, *Littoraria irrorata,* have been shown to be positively correlated with the density and distribution of the salt marsh cord grass *Spartina alterniflora* (Bertness 1984; Hamilton 1978). The distributions of *G. demissa* and *L. irrorata* may also be affected directly by channel modification if the presence of spoil bank makes the marsh edge inaccessible during settlement, or if the soil properties of spoil bank sediment are inappropriate.

S. alterniflora, *G. demissa demissa* (*G. demissa*), and *L. irrorata* are three conspicuous species in the tidal salt marshes of the Gulf coast. The activities of these organisms affect nutrient cycling within the marsh and between the marsh surface and channel system, and may indirectly affect the abundance of prey organisms and the growth of the vegetation that serves as a refuge for many nekton species. The relationships between these organisms and marsh morphology are not well known and there is virtually no data on the effects of changing marsh edge by the creation of dredged channels. This study examines natural creeks and dredged canals of a Louisiana tidal marsh in order to determine if there are: (1) differences in the distributions of *S. alterniflora*, *G. demissa*, and *L. irrorata*, (2) differences in nutrient content of plants and soils, (3) correlations between plant and animal biomass and soil properties, and (4) correlations between biomass and channel morphology or marsh elevation.

Methods

Site Selection

Fifty sites were randomly chosen at natural creek, dredged canal, and open bay marsh edges over an area of approximately 36 km^2 of *Spartina alterniflora* dominated tidal salt marsh in southeastern Barataria Bay, Louisiana. The study area (89.83 ^{o}W, 29.45 ^{o}N and 89.75 ^{o}W, 29.41 ^{o}N; Figure 4.1) is primarily a salt to brackish marsh dominated by *S. alterniflora*, with diurnal 30 cm tides influenced by wind and precipitation. Petroleum exploration and drilling programs in the area began in the late 1930’s with the advent of equipment capable of drilling in aquatic environments, resulting in a coastal plain with a massive network of man-made canals (Davis 1973). The canal networks examined in this study are estimated to have been dredged between 1933 and 1969 based on the date the oil/gas well drill bit pierced the ground (Louisiana Department of Natural Resources, accessed 2005).

Sites were selected by placing a grid of numbered 10 ha hexagons over the area, then using a random number generator to select from those hexagons that fell on natural marsh edge (22 sites), marsh edge that has been modified by dredging (20 sites), and marsh edge at open water bodies (8 sites). The channel width, linear distance to the nearest bay, and channel distance to the nearest bay (Figure 4.2) were measured at each site in Arc View using digital orthophotos taken in 1998. Elevations at 5 m and 10 m from the marsh edge were obtained from LiDAR data (U.S. Army Corps of Engineers 2003).

Sample Collection

Two 0.5 m X 0.5 m quadrats were placed one meter apart at the marsh edge of each of the fifty sites in October, 2004. *Litoraria irrorata* (*L. irrorata*) were collected by hand from the plots, and then all above ground vegetation was cut at the marsh surface and collected. Vegetative litter within the plot was also collected. All live *Geukensia demissa demissa* (*G. demissa*) and shell litter were then collected from the plot by hand, and approximately 0.25 liters of sediment were collected from the surface and sealed in plastic bags while pushing out as much air as possible. The quadrats were then pushed directly backward away from the mash edge, where two more groups of samples were taken for a total of four plots sampled per site (Figure 4.3).

Vegetation samples were washed and separated into live *Spartina alterniflor*a (*S. Alterniflora*), other live vegetation, and dead vegetation. Vegetation samples were dried at 70 ^{o}C for five days and weighed to ± 0.01 g. *G. demissa* samples were frozen prior to processing. After thawing, the shells were cleaned, opened by cutting the posterior adductor, the byssus was removed, and the meat was rinsed in deionized water and removed from the shell. Shell length was measured with calipers from the apex to the posterior margin to 0.1 mm. The shells and meat of animals greater than 20 mm in length were dried at 60^{o}C for 48 hours then weighed to ± 0.01 g. *L. irrorata* were rinsed in deionized water, and then dried at 60 ^{o}C for 6 days. All individuals from a plot were weighed together to ± 0.01 g.

Vegetation and soil samples were analyzed by the Louisiana State University Ag Center. Soil samples were analyzed for nitrogen, phosphorus, sodium, and sulfur by the addition of 20 ml of Mehlich 3 extractant solution to 2 g of soil, shaking for 5 min on fast speed, and results were read on an Inductively Coupled Plasma Mass Spectrometer (ICPMS). The organic matter content was determined by adding 10 ml 1 N K_2Cr_{2O7} and 20 ml concentrated sulfuric acid, waiting 2 hr, then adding 90 ml water, letting the solution equilibrate for 16 hr, and then reading the results on a colorimeter. Dried live and dead leaf tissue samples were ground and analyzed for phosphorus, potassium, iron and sodium by adding 5 ml concentrated HNO_3 to 0.5 g samples, waiting 50 minutes, adding 3 ml H_2O_2 digest for 2.75 hr on a heat block, cooling, and then diluting and reading on an ICPMS. Dry combustion on a Leco N analyzer was used to assess nitrogen content. Soil bulk density was measured by drying 5 ml of soil from each sample site at 60^{0}C for 72 hours, which were then weighed to ± 0.01 g.

Data Analysis

Samples were pooled over the four quadrats at each site and expressed as g dry weight biomass m^{-2}. The total biomass of *S. alterniflora* and *G. demissa* were compared

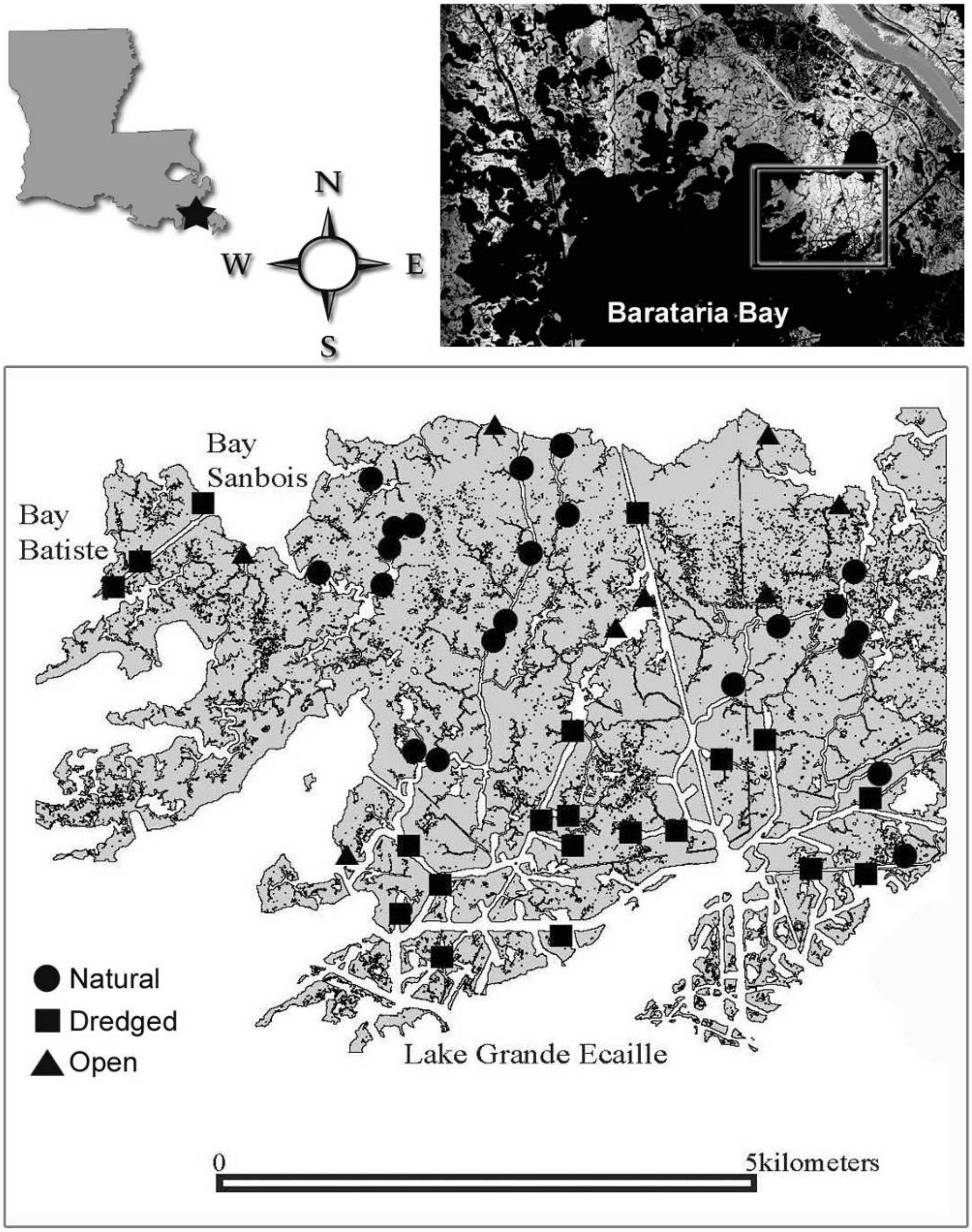

Figure 4.1. The locations of the fifty sites randomly chosen at natural, dredged, and open marsh edges. The area is approximately 36 km^2 of a *Spartina alterniflora* dominated tidal salt marsh in southeastern Barataria Bay, Louisiana.

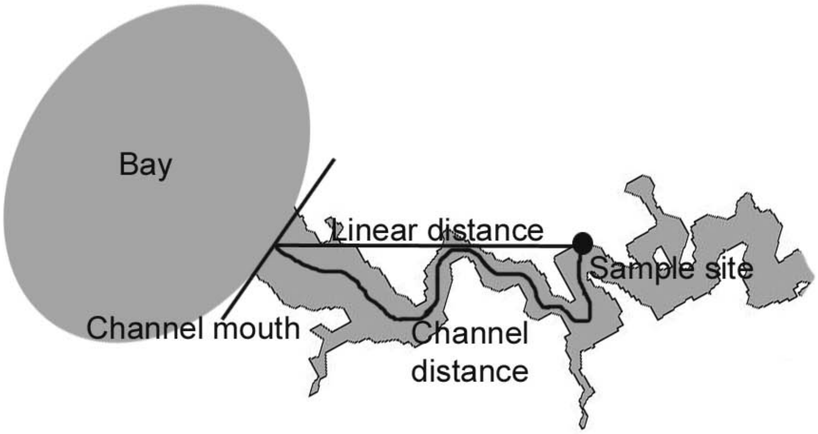

Figure 4.2. An illustration of the methods of measuring channel width, linear distance to the nearest bay, and channel distance to the nearest bay. Measurements were made in Arc View using digital orthophotos of the area taken in 1998.

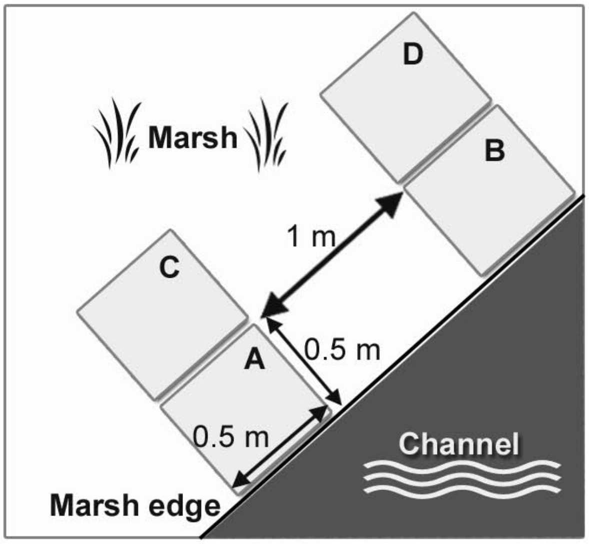

Figure 4.3. An illustration of the placement of four 0.5 m X 0.5 m quadrat samples at the marsh edge. *Littoraria irrorata*, *Spartina alterniflora*, vegetative litter, *Geukensia demsissa demissa*, shell litter, and soil samples were collected.

among natural, dredged, and open bay (open) site types using an analysis of variance (ANOVA) and Tukey's post-hoc test, and between natural and dredged site types using a standard t-test (Insightful Corporation 2001(a)). The variation of the abiotic variables of channel width, elevation at 5 m from the edge, elevation at 10 m from the edge, linear distance, channel distance, the ratio of linear to channel distance (as a measure of sinuosity), soil nutrient composition (carbon, nitrogen, phosphorus, sodium, and sulfur) and soil bulk density was examined by performing a principal components analysis (PCA) in S-Plus using the princomp function and a correlation matrix (Insightful Corporation 2001(b)). A variogram analysis was used to provide measures of spatial correlation in the *S. alterniflora* and *G. demissa* total biomass values over all fifty sites. The variogram provides a measure of how sample data are related with distance and direction, and is calculated as half the average squared difference between points separated by distance h:

$$\gamma(h) = \frac{1}{2\,|N(h)|} \sum_{N(h)} (z_i - z_j)^2$$

where $N(h)$ is the set of all pairwise Euclidean distances $i - j = \mathrm{h}$, $|N(h)|$ is the number of distinct pairs in $N(h)$, and z_i and z_j are data values at spatial locations i and j, respectively (Kaluzny et al. 1998).

Results

A total of 57.4 kg (dry weight) of *S. alterniflora*, 20.2 kg of *G. demissa*, and 0.34 kg of *L. irrorata* were collected from fifty sites. *S. alterniflora* was the dominant vegetation at all sites; *G. demissa* and *L. irrorata* were present at 46 and 48 of 50 sites, respectively. There was no statistical difference between the biomass results for *S. alterniflora*, *G. demissa*, or *L. irrorata* plots sampled closest to the edge (A and B, Figure

4.3) and those sampled at 0.5 m from the edge (C and D), and so the data from all four quadrats was used to calculate site means.

Spartina alterniflora

The mean biomass of live *S. alterniflora* at natural edge sites (927.2 g $m^{-2)}$ was significantly greater than the mean biomass at dredged sites (658.3 g m^{-2}), but the biomass of open edge sites (810.7 g m^{-2}) was not significantly ($p > 0.05$) different from natural or dredged sites (ANOVA, $F_{2,47} = 7.122$, $p = 0.002$). Significantly more dead plant material was also found at natural sites than at the dredged or open sites, but dredged and open sites did not differ significantly from each other (ANOVA, $F_{2,47} = 19.628$, $p < 0.01$) (Table 4.1 and Figure 4.4).

The leaves of live *S. alterniflora* from natural edge sites had significantly more nitrogen and phosphorus than those of dredged sites (nitrogen ANOVA, $F_{2,47} = 4.95$, $p = 0.011$; phosphorus ANOVA, $F_{2,47} = 7.60$, $p<0.01$) (Figure 4.5). The nitrogen in leaves from open sites was not significantly different from leaves collected at either natural or dredged sites, but the phosphorus content was significantly less than in leaves from natural sites. The leaf content of potassium was also significantly higher at natural sites compared to either dredged or open sites (ANOVA, $F_{2,47} = 11.19$, $p<0.01$). There were no significant differences in iron or sodium in leaves at the different site types.

The nitrogen, phosphorus, and iron content of dead vegetation did not differ among natural, dredged, and open site types. The sodium content of dead vegetation from dredged sites was significantly greater than at natural sites, but did not differ significantly from open sites (ANOVA, $F_{2,47} = 3.98$, $p = 0.03$) (Figure 4.6A). The potassium content of dead vegetation was significantly higher at open sites than at natural sites, but there was no significant difference between open and dredged or between dredged and natural sites (ANOVA, $F_{2,47} = 4.27$, $p = 0.02$) (Figure 4.6B).

Geukensia demissa

The highest mean density of *G. demissa*, 98 individuals m^{-2}, was found at open edge sites, and was not significantly different from the mean density of 82 individuals m^{-2} at natural sites. The density of 30 individuals m^{-2} at dredged sites was significantly less than at natural and open sites (ANOVA, $F_{2,47} = 4.91$, $p = 0.01$). The total biomass (shell + body weight) for *G. demissa* was significantly greater at open sites than at natural and dredged sites, but the biomass at natural and dredged sites did not differ significantly from each other (ANOVA, $F_{2,47} = 3.16$, $p = 0.05$) (Figure 4.4). However, the number of open sites sampled (8) was small compared to the number of dredged (20) and natural (22) site types, and the variance of *G. demissa* biomass at open sites was more than twice as great as that of dredged channel and natural sites. The biomass of *G. demissa* was compared between dredged and natural creek sites only (not including open edge) by standard two-sample t-test, and mussel biomass was significantly greater at natural sites than at dredged sites ($p = 0.01$). The mean shell length of all *G. demissa* collected was 54.5 mm, the mean body weight was 0.23 g, and the mean shell weight was 6.60 g. The mean shell length to shell weight ratio was significantly higher at natural sites than at dredged or open sites (ANOVA, $F_{2,3197} = 13.65$, $p < 0.001$), and the mean shell weigh to body weight ratio was also significantly greater at natural sites (ANOVA, $F_{2,3197} = 17.32$, $p < 0.001$). The amount of shell litter collected from the plots did not differ among site types.

Littoraria irrorata

The number of *L. irrorata* found ranged from 0 to 134 m^{-2}. The mean number of individuals for natural plots was 34 individuals m^{-2}, 31 individuals m^{-2} for open edge sites, and 19 individuals m^{-2} for dredged sites, but the means were not significantly different when tested by an ANOVA. There also was no significant difference in the mean snail mass among site types.

Soil Properties

The soils at the open edge sites had the highest values for nitrogen, sodium, and carbon content, and the lowest value for phosphorus content (Figure 4.7). The soil nitrogen and sodium content were significantly greater at open sites and at natural sites than at dredged sites, but there was no difference between open and natural creek sites (nitrogen ANOVA, $F_{2,47} = 7.5$, $p<0.01$; sodium ANOVA, $F_{2,47} = 5.9$, $p<0.01$). Soil carbon was significantly higher at open sites than at dredged sites, but was not different from natural sites, and soil carbon at natural sites was not different from dredged sites (ANOVA, $F_{2,47} = 4.92$, $p = 0.01$). Soil phosphorus was significantly higher at natural sites than at open sites, but there was no significant difference in the soil phosphorus between natural and dredged or open and dredged sites (ANOVA, $F_{2,47} = 3.65$, $p = 0.03$). Soil bulk density values were significantly higher at dredged sites than at natural or open sites (ANOVA, $F_{2,47} = 8.93$, $p < 0.001$).

Elevation and Channel Characteristics

The mean elevations measures obtained from the LiDAR data at 5 m and at 10 m from the edge did not differ among site types. The channel (c) distance, linear (l) distance, and c/l distance (sinuosity) did not differ significantly between dredged canals and natural creeks, but the mean channel width was significantly greater in dredged canals (60.7 m) than in natural (28.2 m) creeks (t-test, $p = 0.002$).

Table 4.1. Means (± standard error) of the biomass and nutrient analysis variables for natural, dredged, and open edge site types. Significant differences between means were tested by ANOVA analysis ($p > 0.05$) and Tukey's post-hoc test and are represented by letters reading horizontally. Results marked with an * were significantly different ($p > 0.05$)when compared to each other by t-test.

	Natural	**Dredged**	**Open Edge**
Spartina alterniflora			
Live biomass (g m^{-2})	927.24 ± 47.03 A	658.31 ± 44.11 B	810.73 ± 115.06 AB
Dead veg. mass (g m^{-2})	453.65 ± 22.39 A	241.47 ± 36.04 B	187.59 ± 32.30 B
Live leaf N (%)	0.92 ±0.04 A	0.74 ± 0.04 B	0.79 ± 0.05 AB
Live leaf P (%)	0.11 ± 0.01 A	0.08 ± 0.01 B	0.08 ± 0.01 B
Live leaf K (%)	0.95 ± 0.06 A	0.62 ± 0.03 B	0.71 ± 0.08 B
Live leaf Fe (ppm)	684 ± 65.	605 ±58	596 ± 112
Live leaf Na (ppm)	22,703 ± 964	23,119 ± 1,375	20,328 ± 1,415
Dead veg. N (%)	0.76 ± 0.30	0.74 ± 0.02	0.84 ± 0.04
Dead veg. P (%)	0.06 ± 0.01	0.05 ± 0.01	0.06 ± 0.01
Dead veg K (ppm)	0.28 ± 0.02 A	0.36 ± 0.02 AB	0.40 ± 0.08 B
Dead veg. Fe (ppm)	1,265 ± 107	1,138 ± 94	837 ± 167
Dead veg. Na (ppm)	20,291 ± 1,317 A	26,205 ± 1,979 B	21,025 ± 877
Geukensia demissa			
Biomass (g m^{-2})	478 ± 68.8 A*	221 ± 67 A*	610 ± 282 B
Individuals m^{-2}	82 ± 12 A	30 ± 9 B	98 ± 41 A
Shell length / shell weight	19.3 ± 0.4 A	16.3 ± 26.9 B	15.9 ± 0.5 B
Shell weight / body weight	30.8 ± 0.5 A	26.9 ± 0.4 B	27.5 ± 0.3 B
Shell length (mm)	58.6 ± 2.4	49.5 ± 6.5	55.6 ± 3.8
Littoraria irrorata			
Density (number m^{-2})	34 ± 6	19 ± 4	31 ± 4
Biomass (g m^{-2})	50.91 ± 10.11	22.80 ± 6.38	45.12 ± 14.5
Soil			
N	0.72 ± 0.03 A	0.52 ± 0.05 B	0.80 ± 0.09 A
P	39.88 ± 4.05 A	33.88 ± 2.92 AB	23.04 ± 1.00 B
Na	15,569 ± 760 A	12,216 ± 1,059 B	18,082 ± 2,011 A
S	3,558 ± 183	3,034 ± 215	3,684 ± 660
C	9.22 ± 0.47 AB	7.22 ± 0.91 A	11.82 ± 1.86 B

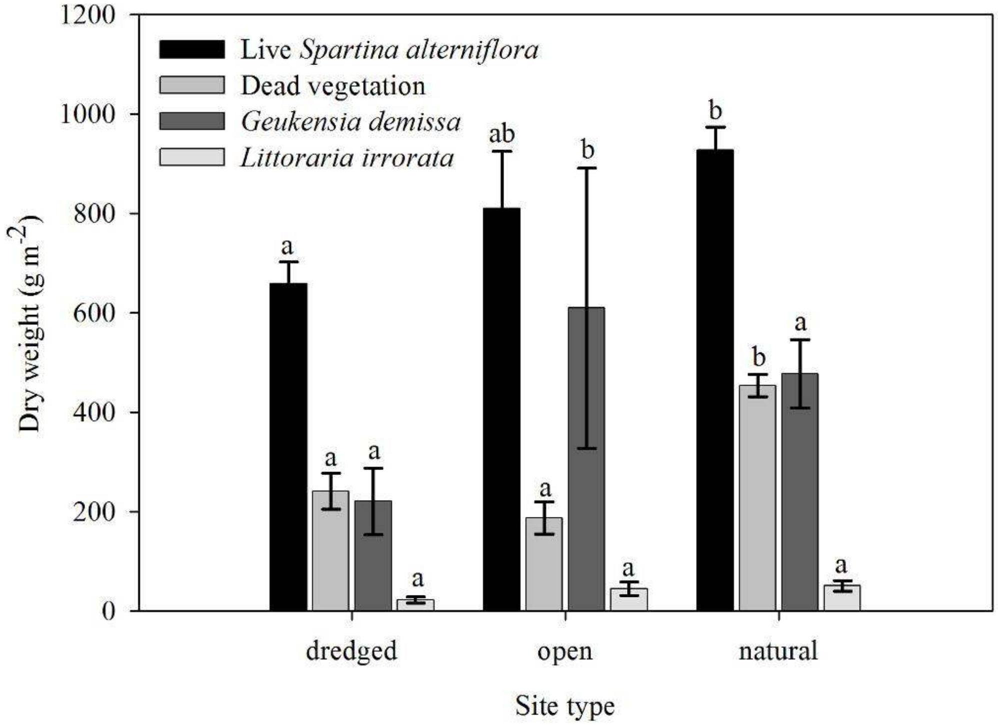

Figure 4.4. Biomass of live and dead *Spartina alterniflora*, *Geukensia demissa demissa*, and *Littoraria irrorata* among the three site types: dredged channel edge, open bay edge, and natural creek edge. The mean ± 1 standard error is shown and significant differences (ANOVA, $p < 0.05$) are show by letters for each species. See Table 4.1 for p values.

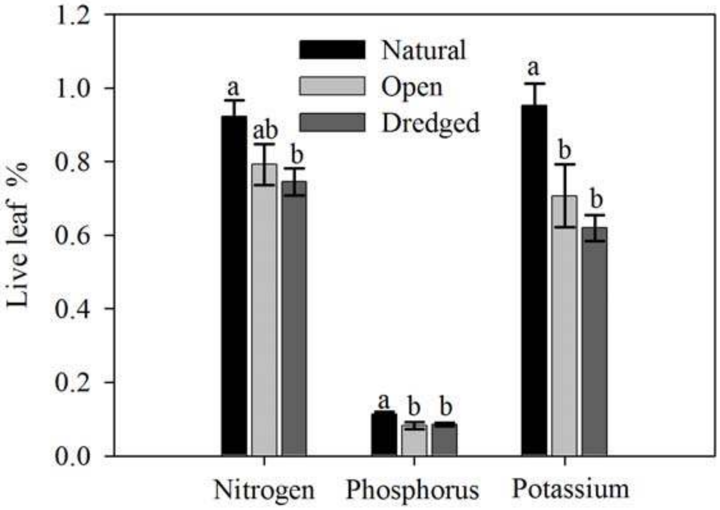

Figure 4.5. The live leaf content of nitrogen, phosphorus, and potassium at natural, dredged, and open marsh edges. The means ± 1 standard error are shown. Significant differences between site types for each nutrient (ANOVA, $p < 0.05$) are marked with differing letters.

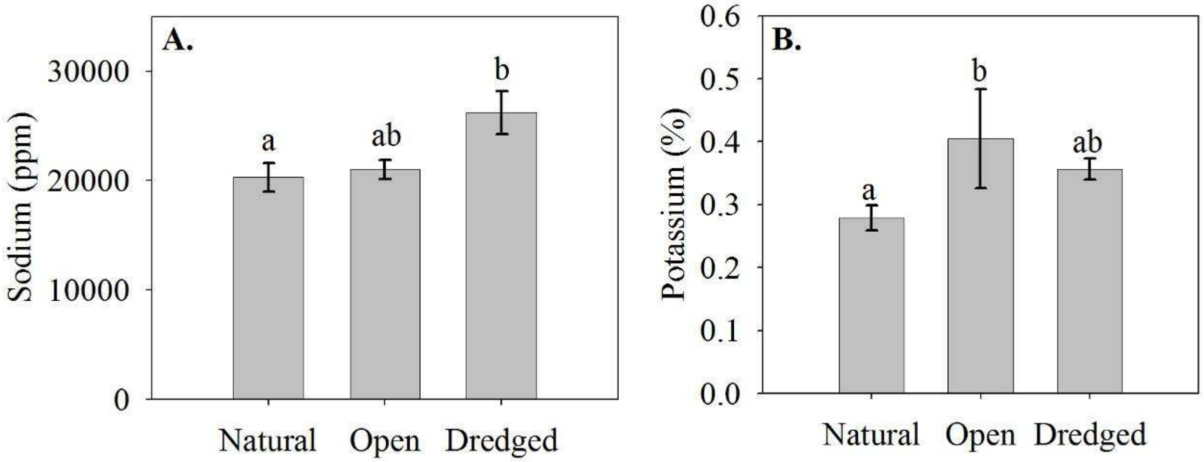

Figure 4.6. The sodium (A) and potassium (B) content of dead vegetation collected from natural, open, and dredged marsh edge. The means ± 1 standard error are shown. Significant differences (ANOVA, $p < 0.05$) are shown by different letters.

Spatial Distribution

Because values of *L. irrorata* were low and similar among site types, only *S. alterniflora* and *G. demissa* biomass values were examined for patterns of spatial distribution. Regressions of biomass values against (x, y) location showed there was no significant trend in values in either north-south or east-west directions. The spherical variogram model fitted to the empirical variogram for *S. alterniflora* had a range distance of 0.31, which corresponds to a distance of approximately 3 kilometers, and is the distance over which points are considered to be spatially correlated. The model variogram for *G. demissa* had a range of 0.25 (spatial correlation over 2.5 km) similar to *S. alterniflora* (Figure 4.8). The directional variograms of *S. alterniflora* for 90^o and 135^o showed a range of correlation in biomass values of approximately 0.2, or 2 kilometers, in west-east and northwest-southeast directions, while the 0^o and 45^o directional variograms showed that there is little or no spatial correlation in the north-south and northeast-

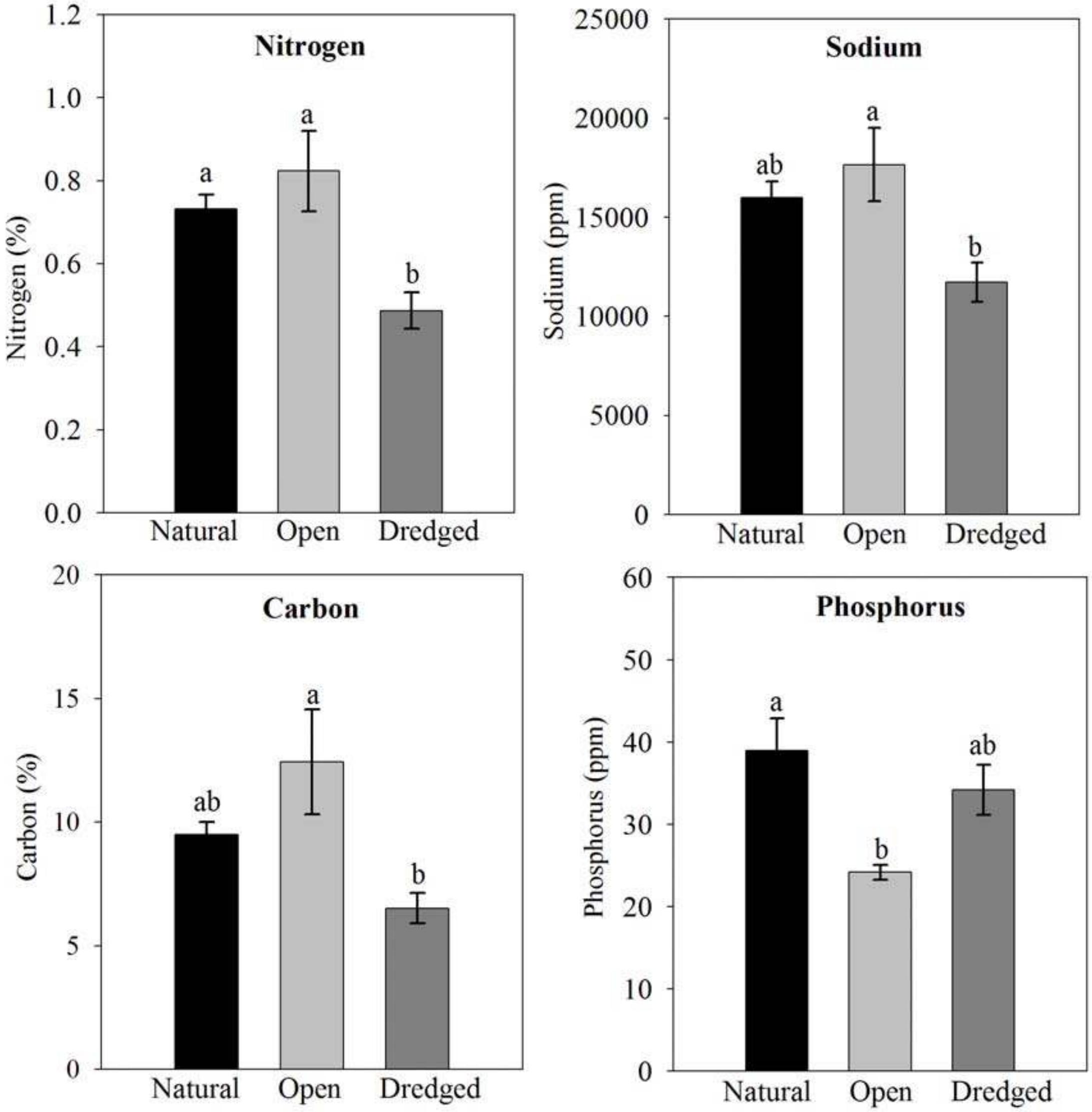

Figure 4.7. The soil nitrogen, sodium, carbon, and phosphorus.content of salt marsh soils. Significant differences (ANOVA, *p* < 0.05) are shown by differing letters.

southwest directions (Figure 4.9). The directional variograms for *G. demissa* did not show a trend of spatial correlation in any direction. There appeared to be no spatial correlation between sites of the same type for either *S. alterniflora* or *G. demissa* when the site types were examined independently (Figure 4.10). In summary, when all site types are examined together, values for *S. alterniflora* and *G. demissa* biomass appear to be spatially correlated over a distance of 2.5 to 3 kilometers, which is about half of the distance across the study site, but when the data is divided by site type, there appears to be no spatial correlation in values for biomass for either species.

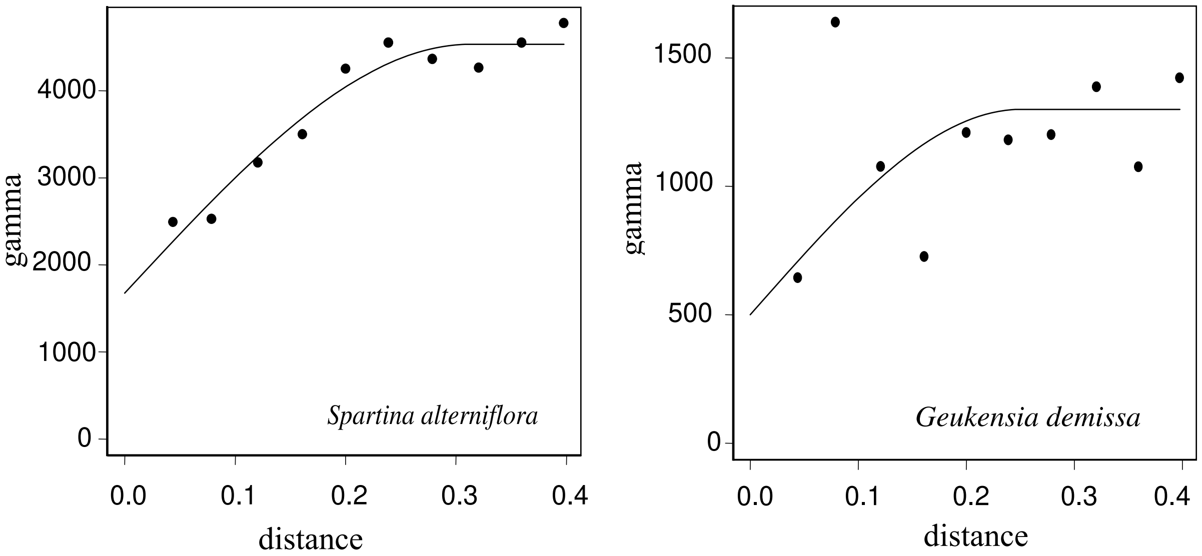

Figure 4.8. Fitted spherical variogram of *S. alterniflora* (top) and *G. demissa* (bottom) biomass values at all sites. Variogram values for *S. alterniflora*: range = 0.31, sill = 2,857, nugget = 1,675, nlag = 10, and for *G. demissa*: range = 0.25, sill = 7,595, nugget = 4,595, nlag = 10. Distance units are grid units, 1 grid unit is approximately 10 kilometers.

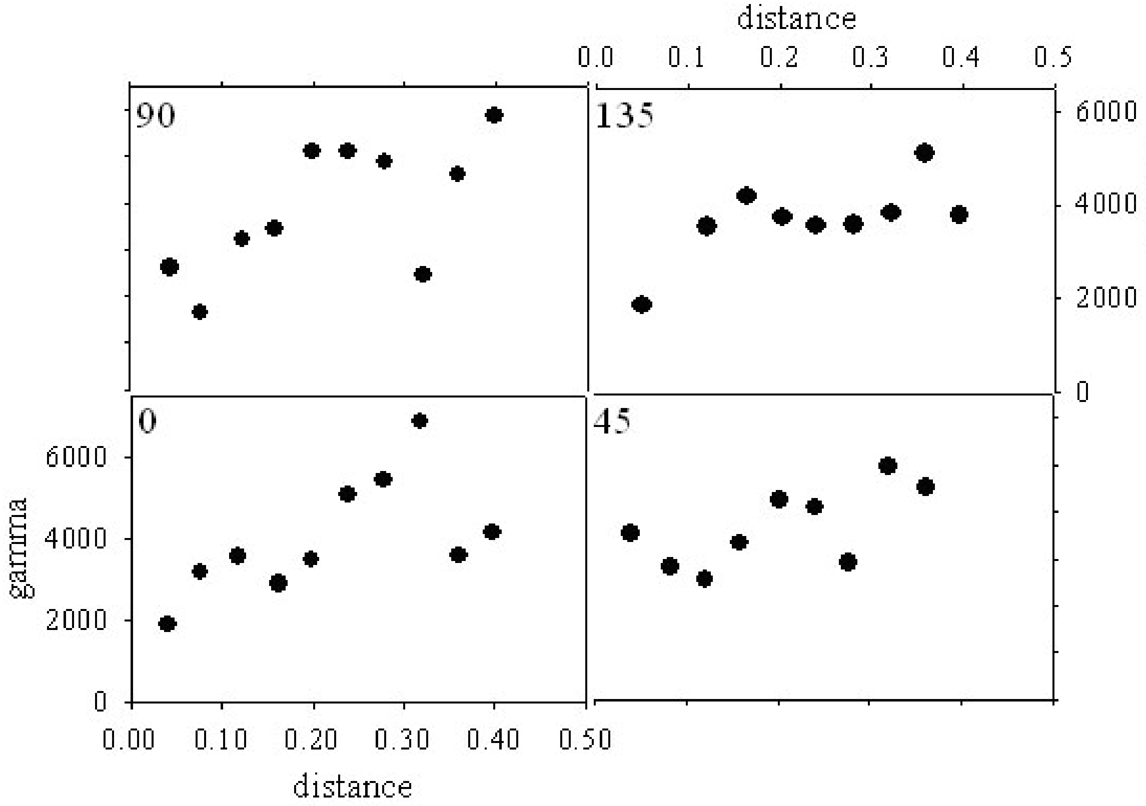

Figure 4.9. The directional variograms for *S. alterniflora* mass values over angles 0^o (north -south), 45^o (southwest-northeast), 90^o (east-west) and 135^o (southeast-northwest). The variograms for 90^o and 135^o show a range of correlation in biomass values of approximately 0.2, or 2 kilometers, in west-east and northwest-southeast directions, while the 0^o and 45^o directional variograms show that there is little or no spatial correlation in the north-south and northeast-southwest directions

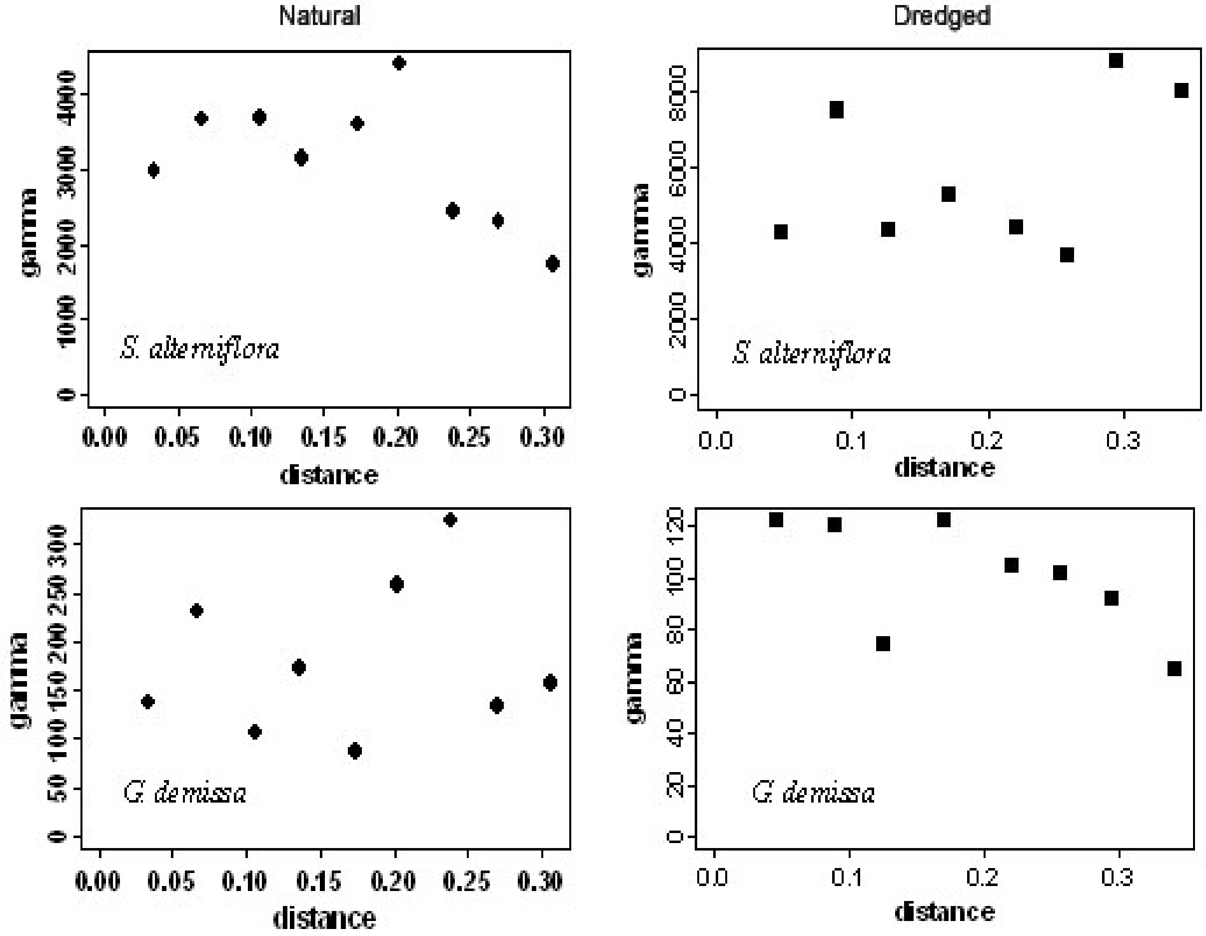

Figure 4.10. *S. alterniflora* and *G. demissa* mass at natural (left) and dredged (right) site types examined separately for spatial correlation using variogram analysis. There appears to be no spatial correlation among sites of the same type.

Correlations and Principal Components Analysis

With the data from the three site types combined, no statistical relationship was found between the biomass of *G. demissa* and that of *S. alterniflora* or *L. irrorata*, but there was a linear relationship between *S. alterniflora* biomass and the number of *L. irrorata* ($R^2 = 0.38$, $p < 0.01$). *S. alterniflora biomass* was also linearly related to the amount of phosphorus in the leaves ($R^2 = 0.16$, $p < 0.01$) and with soil nitrogen ($R^2 = 0.12$, $p = 0.01$), and was weakly correlated with soil sodium, ($R^2 = 0.07$, $p = 0.06$). The biomass of dead plant material was found to be linearly related to the biomass of live *S. alterniflora* ($R^2 = 0.19$; $p < 0.01$), the live leaf nitrogen ($R^2 = 0.38$; $p < 0.01$), and also with the linear distance of the site from the bay ($R^2 = 0.21$, $p < 0.01$). *G. demissa* biomass was positively related to live leaf nitrogen ($R^2 = 0.17$, $p < 0.003$) and shell litter weight ($R^2 = 0.42$, $p < 0.001$).

The natural and dredged sites had different patterns in correlations when analyzed separately. Natural sites showed significant positive relationship between live *S. alterniflora* biomass and the number of *L. irrorata* ($R^2 = 0.18$, $p = 0.04$), soil nitrogen ($R^2 = 0.25$, $p = 0.02$), and soil sodium ($R^2 = 0.23$, $p = 0.02$), but not between *S. alterniflora* and *G. demissa* biomass or leaf phosphorus. For dredged sites, *S. alterniflora* biomass was positively related to the number of *L. irrorata* ($R^2 = 0.36$, $p < 0.01$), leaf phosphorus ($R^2 = 0.22$, $p = 0.04$), and *G. demissa* biomass ($R^2 = 0.24$, $p = 0.03$), but not with soil nitrogen or soil sodium.

A principal components analysis (PCA) of the twelve abiotic variables for the natural creeks and dredged canal sites (n = 42) identified 3 components with eigenvalues greater than one, which explained 69 % of the variance (Table 3.2). Soil carbon, nitrogen, and sodium loaded negatively and soil bulk density loaded positively on component 1. Channel width loaded positively and 5-10 m elevation and soil phosphorus loaded negatively on component 2, while 0-5 m elevation, linear distance, channel distance, and c/l distance loaded negatively on component 3.

Discussion

The mean *S. alterniflora* biomass value at natural creek edges (927.2 g m^{-2}) was higher than that found by Darby (2006) in a Louisiana marsh in September (876.8 g m^{-2}), but both the natural edge mean value and the mean value at dredged canal edges (658.3 g m^{-2}) were within the wide range of average peak biomass values previously reported for Louisiana marshes. For example, Visser and Sasser (2006) reported values from 473 g m^{-2} to 1698 g m^{-2}. *G. demissa* values in this study ranged from a low of 30 individuals m^{-2} at dredged sites to 98 individuals m^{-2} at open sites. These densities are generally lower than those reported for east coast marshes. Jordan and Valiela (1982) reported *G. demissa* densities from 364 to 734 individuals m^{-2} along creek banks, 140 individuals m^{-2} in tall *S. alterniflora*, and 34 individuals m^{-2} in short *S. alterniflora* in a New England salt marsh. Bertness (1980) found mean densities of 422, 88, and 186 individuals m^{-2} in three marshes of the Chesapeake Bay. On the Gulf coast, densities of 2.8 individuals m^{-2} were found in Alabama (West and Williams 1986).

Some of the suggested controls on *S. alterniflora* production include salinity, hydroperiod (or redox potential), and nutrient limitation (Tyler and Zieman 1999, Howes et al. 1981). The average range of channel water salinity in the study area is 8 to 15 psu and the gradient across the study area is typically not more than 2 psu (Granados-Dieseldorff 2006, Spicer, unpublished data). Pore water salinity was not measured, but soil sodium was highest at the natural sites. *S. alterniflora* has been shown to tolerate salinity of up to 115 psu (Hester et al. 1998). Although it has been shown that salinity can become a significant controlling factor on *Spartina* spp. following disturbance (Baldwin and Mendelssohn 1998), all of the dredged canals examined in this study were at least 30 years old and therefore much older than the effects of disturbance would be expected to persist. I conclude that it is unlikely that salinity varies enough between natural and modified sites, or reached high enough concentrations in the soil or

Table 4.2. The factor loadings from a principal component analysis of 12 abiotic variables (channel dimensions, bank elevation, and soil nutrients) from natural creeks and dredged canals. Loadings in bold were used to characterize factors.

Abiotic Variable	Principal component factors		
	1	2	3
Channel width	0.18	**0.34**	-
0-5 m height	-	-0.23	**-0.39**
5-10 m height	-	**-0.63**	0.11
Linear distance (l)	-0.22	-0.28	**-0.47**
Channel distance (c)	-0.27	-	**-0.58**
c/l distance	-0.17	0.29	**-0.35**
Soil C	**-0.43**	0.15	0.15
Soil N	**-0.43**	0.11	0.11
Soil P	-	**-0.39**	0.19
Soil Na	**-0.44**	-	0.11
Soil S	-0.30	-0.26	0.23
Soil density	**0.40**	-	-0.16
	-	-	-
Standard deviation	2.16	1.36	1.32
Proportion of variance explained	0.39	0.15	0.15
Cumulative proportion explained	0.39	0.54	0.69

leaf tissue to account for the differences in *S. alterniflora* production among site types.

Changes in hydroperiod, such as prolonged flooding events, can cause a shift from aerobic to anaerobic production in root metabolism, limiting above ground production (Howes et al. 1981). The timing and duration of flooding events at a marsh edge site is strongly affected by marsh surface topography (Cahoon and Reed 1995). Dredged spoil banks restrict water flow both above and below the adjacent marsh (Swenson and Turner 1987), and may cause interior "ponding" of water on the marsh surface. There were no significant differences among site types in the 5 m or 10 m elevations measured using the LiDAR data, however all samples were taken within the first meter of the marsh edge and edge profiles were not measured. The data collected are insufficient to determine if hydroperiod was a cause of the differences in biomass values between natural and dredged marshes.

The aboveground accumulation of *S. alterniflora* biomass is generally believed to be nitrogen limited (Morris 1982), and increases in primary production and plant nitrogen content have been shown to occur with nitrogen fertilization (Valiela and Teal 1979). *S. alterniflora* biomass, soil nitrogen, and leaf nitrogen all had significantly lower values in dredged marshes than in natural marshes. There was a weak but statistically significant correlation across site types in plant biomass and soil nitrogen, suggesting that nitrogen limitation may be one factor affecting the growth of *S. alterniflora* at dredged sites.

Nitrogen fertilization may occur through pseudofeces deposition by mussels. Nitrogen absorbed by mussels is excreted as ammonia or dissolved organic nitrogen. Jordan and Valiela (1982) estimated that as much as 50 % of the nitrogen absorbed by *G. demissa* is deposited as feces and pseudofeces, and that 55 % of the nitrogen absorbed is excreted as ammonia. Bertness (1984) showed that the experimental removal of mussels from *Spartina* marsh plots decreased the soil nitrogen by 0.18 %, which is approximately the difference in soil nitrogen found between natural and dredged sites in this study (0.2 %). When the data are examined for natural and dredged sites (Figure 4.11A), there does not appear to be a strong relationship between soil nitrogen and total mussel biomass per m^2, but there does appear to be a weak relationship between the *number* of mussels and soil nitrogen (Figure 4.11B). Plots with a high total *G. demissa* biomass can be composed of a few large individuals or many smaller individuals. Soil nitrogen is inversely correlated with the mean individual weight (shell plus body) of mussels (Figure 4.11C), and the mean individual weight decreases as density increases (Figure 4.11D), meaning that plots with a few large individuals generally had lower soil nitrogen than those with many smaller individuals. Smaller mussels may have higher metabolic rates than large mussels as a consequence of their higher surface to volume ratios (Jordan and Valiela 1982). Mussels in natural plots were more dense and had a smaller mean body length than the mussels in dredged plots. The effect of *G. demissa* on soil nitrogen fertilization may be dependent upon not only the number of mussels, but also their size distribution.

Kreeger et al. (2000) suggest that *G. demissa*, too, may be nitrogen limited, because the estimated supply of nitrogen sources, primarily phytoplankton, do not meet the estimated demand. Chintala et al. (2006) found that *G. demissa* density and total biomass increase with nitrogen loading, possibly because increasing nitrogen enhances food sources such as *S. alterniflora* detritus, microheterotrophs, particulate organic matter, or phytoplankton.

The relationship between *S. alterniflora* and *G. demissa* may be one of facultative mutualism: mussels fertilize *Spartina* through nitrogen excretion, and the increased *Spartina* density and biomass reduce mussel predation and increase food sources. In this study, shell litter was collected as evidence of predation, but the mean dry weight of shell litter was not significantly different between natural and dredged sites, and there was no relationship between litter weight and either *S. alterniflora* biomass or stem number. The primary predator on *G. demissa* is the blue crab *Callinectes sapidus*, with predation by mud crabs (*Panopeus herbstii*) also found in some studies. West and Williams (1986) found no size selectivity of mussels by blue crab, and Hughes and Seed (1981) found that crabs consumed mussels up to 80 mm in length, however mud crab predation may be restricted to mussels under 50 mm (Seed 1980). Evidence of predation may be seen in length frequency histograms of *G. demissa* if predation is strongly size class restricted. The length frequency histograms for *G. demissa* (see next chapter) do not show any patterns that would resemble selective size class predation which, based on the literature, may be expected. It cannot be determined from this data if predation differed among site types and if it is a factor controlling mussel distribution.

Food resource limitation can result from a low abundance of phytoplankton and other food resources, or because of limited inundation reducing feeding time. The growth in shell and soft tissue are not coincidental (Borrero and Hilbish 1988), and mussels may respond to food-limitations by reducing absolute tissue growth rates more severely than absolute shell growth (Franz 1993), although Lent (1967) found there was no relationship between intertidal height and shell-weight/meat-weight ratio. Franz (1997) demonstrated that limits in feeding time and food supply resulted in mussels of a

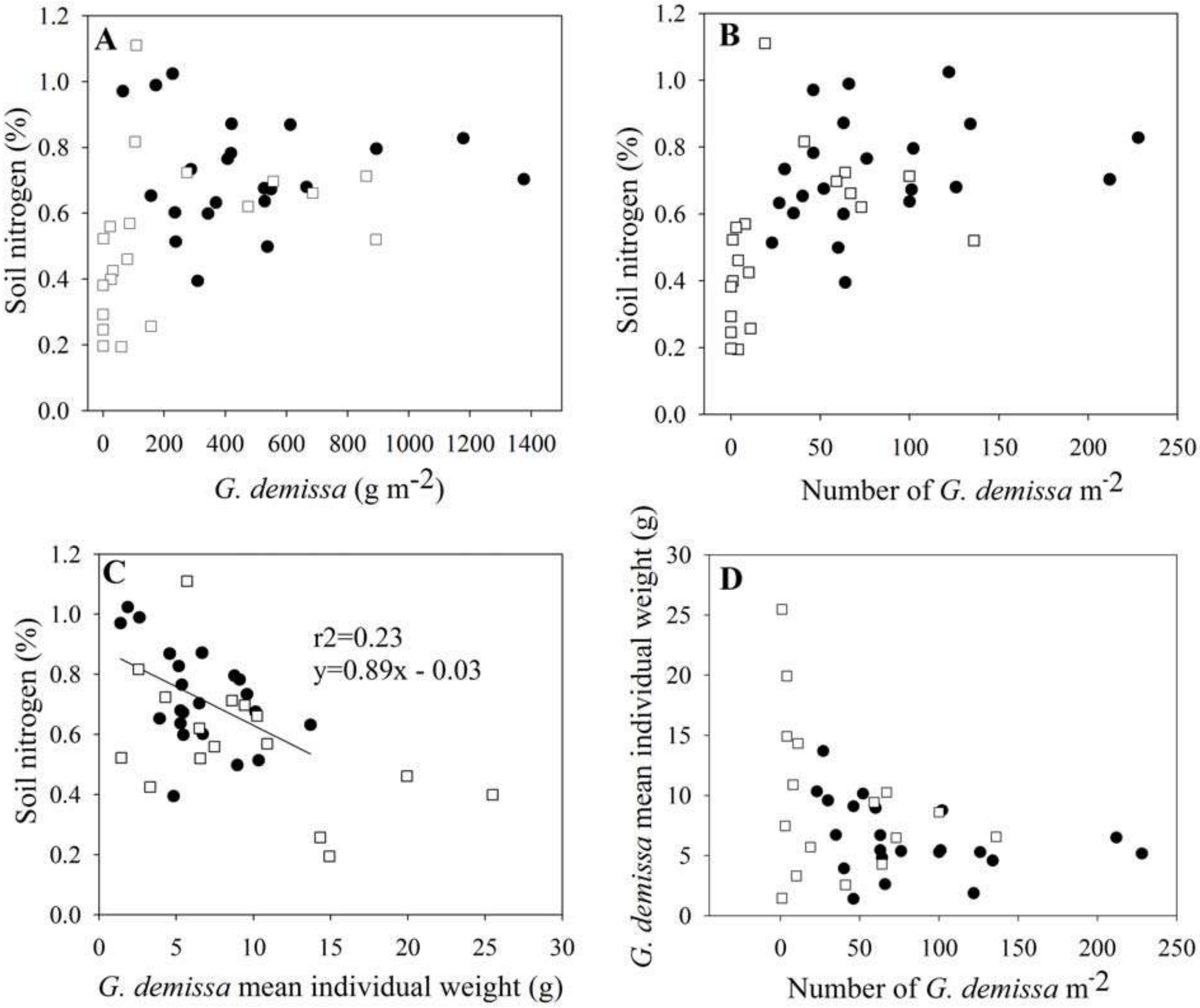

Figure 4.11. A. The relationship between soil nitrogen and total mussel biomass m^{-2}. B. The relationship between the *number* of mussels and soil nitrogen. Plots with a high total *G. demissa* mass can be composed of a few large individuals or many smaller individuals. C. The relationship between soil nitrogen and the mean individual weight (shell plus body) of mussels. D. The mean individual weight and density relationship. Plots with a few large individuals generally had lower soil nitrogen than those with many smaller individuals.

given shell length having a lower body mass at higher elevations than mussels of equivalent length at the marsh edge. The mussels that were removed from natural sites in this study had higher mean shell length-to-shell weight and shell length-to-body weight ratios than those removed from dredged or open sites. Based on the above findings, higher shell-to-body weight ratios would be expected to occur under conditions with limited inundation, or under conditions were food sources are limited. The LiDAR data does not show a difference in elevation over the first 5 m of the marsh edge, but, again, differences in bank slope would not show up at this resolution and were not measured in the field, and so the effect of inundation cannot be determined. There was a greater density of mussels at natural sites, which could limit the amount of food available for individual mussels due to pre-filtering by neighboring mussels.

The principal components analysis showed that 69 % of the variability in the abiotic factors was accounted for by the nitrogen and carbon content and bulk density of the soil. This is consistent with the hypothesis that nitrogen limitation is a significant factor in *S. alterniflora* distribution. Soil bulk density is not linearly related to *G. demissa* biomass in the plots (Figure 4.12), but there does appear to be an optimal range of approximately 0.25 to 0.4 g cm^{-3} for *G. demissa*. The mean bulk density for natural sites of 0.3 g cm^{-3} falls within this range, while the mean value of 0.44 g cm^{-3} for modified sites was above this range. The higher bulk density at modified sites probably reflects the source of the dredge spoil sediments from the channel bottom. These denser sediments may be more difficult for mussels to settle on.

Landscape factors, including channel width, elevation, and distance from the site to the channel mouth accounted for the second and third components of the principal components analysis. Both physical and ecological conditions within channels are affected by channel morphologic parameters such as outlet width and sinuosity. For example, water velocity at the surface has been shown to be positively correlated with channel outlet width, and the amount of organic material in channel sediments scaled negatively with channel size (Hood 2002). In this study, dredged channels had a mean width that was more than twice that of natural channels, and dredged channel width did not scale with channel sinuosity as measured by the ratio (channel distance) : (linear distance to the channel mouth). Wider, less sinuous dredged canals may be more subject to erosion at their edges. Banks subject to erosion may be less suitable for mussel survival (Jordan and Valiela 1982).

Summary and Conclusions

Nitrogen limitation may be one cause of the greater biomass values of *S. alterniflora* at natural compared to dredged sites; there was not sufficient data to determine if salinity and hydroperiod differences affected plant growth. *G. demissa* density was greater at natural than at dredged sites, which could indicate better food resources, less predation on settled mussels, or higher rates of settlement. It has been suggested that the relationship between *S. alterniflora* and *G. demissa* may be mutually facultative, with *G. demissa* supplying nitrogen to *S. alterniflora* through psuedofeces deposition, and *S. alterniflora* providing protection and increased food resources for *G. demissa.* In this study, there did appear to be a relationship between soil nitrogen and the number of mussels, and this relationship is affected by the size distribution of the mussels. There was no evidence of a relationship between *S. alterniflora* density and predation on *G. demissa* as measured by shell litter. Allometric variation in the ratios of

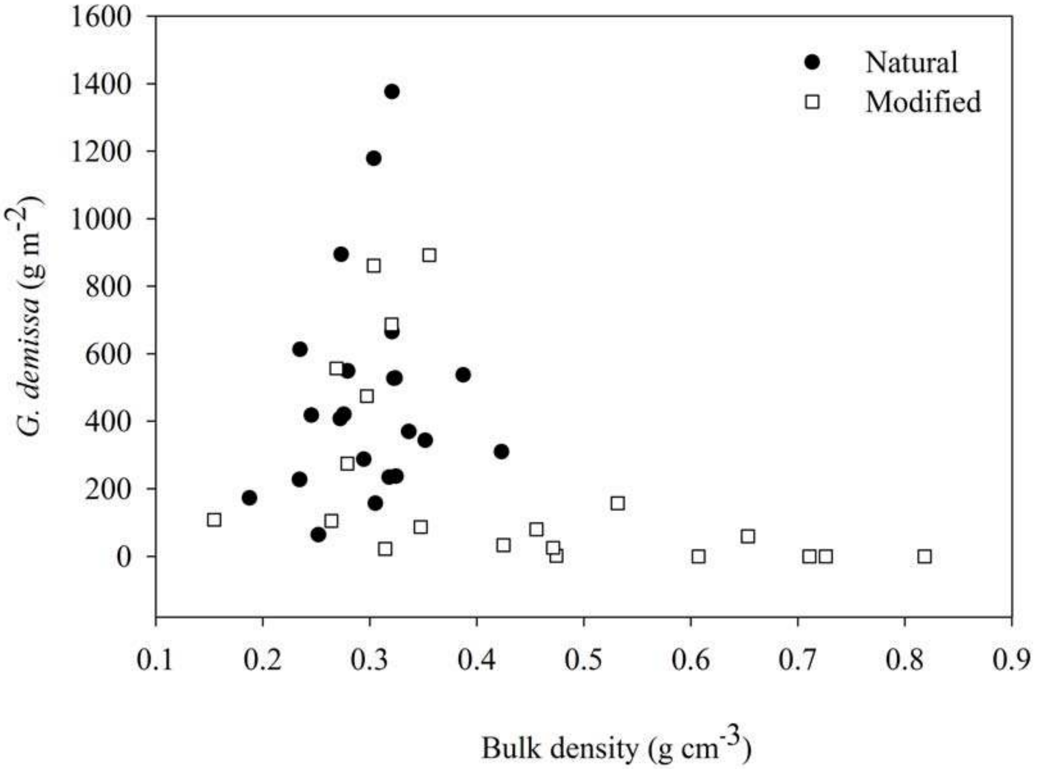

Figure 4.12. Soil bulk density is not linearly related to *G. demissa* biomass, but there does appear to be an optimal range for *G. demissa* of 0.25 to 0.4 g cm^{-3}.

shell length-to-body weight and shell weight-to-body weight suggest there may be differences in food resources between natural and dredged sites. The amount of food available, inundation time, and mussel density have been shown to affect mussel growth patterns. Further research is needed to measure fine scale bank slope and inundation differences between site types, as well as to examine differences in predation and recruitment.

The nitrogen and carbon content and bulk density of soils accounted for 39 % of variation in the abiotic variables, and channel morphology and marsh elevation accounted for 69 % of the variance. This suggests that the source of the sediment (i.e. natural deposition or dredge spoil) may be more important that the effects of channel shape or marsh elevation, at least within the range of elevations measured in this study. This may be an important consideration when artificially introducing tidal channels into constructed wetlands. It is also worth noting that these differences in soil properties occurred even though the canals examined in this study were at least thirty years old.

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CHAPTER 5

TOWARD A MODEL FOR LINKING LANDSCAPE CHANGE TO ECOSYSTEM FUNCTION: POPULATION STRUCTURE OF *GEUKENSIA DEMISSA DEMISSA* AT NATURAL AND DREDGED MARSH EDGES

Introduction

The development of major tidal networks in undisturbed tidal marshes involves a direct feedback between marsh elevation and the tidal prism (French and Stoddart 1992). The colonization of the marsh surface by halophytic vegetation is related to elevation and tidal fluctuations, and the presence of vegetation stabilizes the configuration of the tidal networks (Mudd et al. 2004, Rinaldo et al. 2004). These processes give rise to the physical properties of tidal channels, such as edge slope and the presence of rivulets, which have been shown to affect infaunal food availability and the rates of piscivorous predation, as well as access to the marsh surface for foraging and refuge (McIvor and Odum 1988, Rozas et al. 1988).

Dredging marshes creates artificial canal systems with characteristics that are different from natural creeks in several important ways. Dredged canal networks are morphologically dissimilar to naturally formed creek systems (Chapter 2), and they are typically straighter, deeper, and have steeper banks, higher mean dissolved oxygen concentrations, and lower mean salinities than natural creeks (Williams and Zedler 1999; Rozas and Reed 1994). These structural and functional differences between naturally formed tidal creeks and dredged canals may have important effects on the biological functioning of the system. In the following text, the term "creek" refers to naturally formed systems, "canal" to artificially constructed systems, and "channel" refers to any conduit of tidal water, natural or artificial.

The population structure of the ribbed marsh mussel, *Geukensia demissa demissa* (*G. demissa*), has the potential to be highly affected by channel morphology. There are several reasons that this potential is likely. For example, the survival of mussel spat released into tidal channels is mediated by currents, channel water temperature and oxygen concentration, and the presence of predators. Settlement occurs as tides flood the marsh edge, and recruitment rates are positively correlated with tidal coverage (Nielsen and Franz 1995). *G. demissa* feed by filtering primarily plankton from the water column during flooding tide (Peterson and Howarth 1987), and their growth rate, as well as reproduction, may be food-limited by submergence time (Franz 1993, Borrero 1987). Desiccation can occur when flooding tides are too infrequent, and mussels in low tide zones may be more susceptible to predation (Stiven and Gardner 1992).

A population's length frequency and age structure can provide insight into how physical and structural processes affect the distribution of organisms. The structural and functional differences between natural and dredged channel systems are examined in this study by comparing the age distribution of mussels at sites with divergent environmental conditions. This information provides a starting point for the development of a conceptual model of how dredged canal systems will function as mussel habitat in comparison to natural creeks.

Methods

G. demissa were collected from four 0.5 m X 0.5 m quadrants placed at the marsh edge of 22 natural creek and 20 dredged canal sites in Barataria Bay, Louisiana. See Chapter 4 for a description of the marsh site. Shell length (± 1 mm) was measured with calipers from the apex to the posterior margin. Age was estimated by counting the external growth rings across the length range on 195 right-valve shells that were randomly selected. Shells were dried at 60^0C for 48 hours and then placed in a solution of 50% bleach for approximately 1 hour to make the growth rings more visible. Growth rings (Figure 5.1) are caused by retraction of the shell-secreting mantle edge into the shell during harsh environmental conditions, and have been previously used to approximate seasonal growth (Bertness 1980).

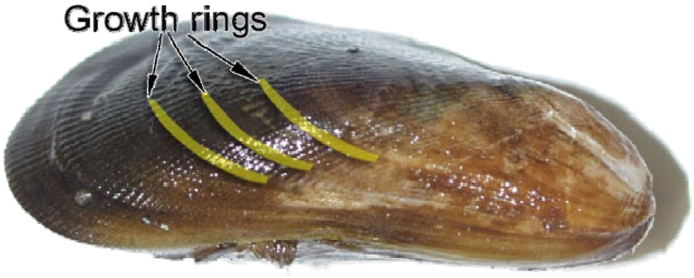

Figure 5.1. Image of a *G. demissa* shell with examples of growth rings highlighted.

The age frequencies of mussels were compared by site type (natural creek edges and dredged canal edges) and for the following variables: channel distance from the nearest outlet, bank elevation, channel width, *Spartina alterniflora* dry weight, and soil bulk density. Channel distance from the sample site to the nearest outlet was measured in ArcView using digital orthophotos of the area taken in 1998. Channel width was measured *in situ* for channels less than 10 m wide, and using the aerial photos for channels greater than 10 m wide. Bank elevation values were obtained from LiDAR data with a horizontal resolution of 5 m and a vertical relative precision of ± 0.07 m (U.S. Army Corps of Engineers 2003). *S. alterniflora* was clipped at the ground and removed from the plots prior to the removal of the mussels, and was washed and then dried at 70

^{0}C for five days and weighed to ± 0.01 g. Soil bulk density was measured by drying 5 ml of soil from each sample site at 60^0 for 72 hours, and then weighing to ± 0.01 g.

The data for each of these variables was divided into quartiles (Table 5.1). Mussels from sites with values less than the 1st quartile were compared with those with values greater than the 3rd quartile. For bulk density, mussels from sites with a bulk density less than the 1st quartile were compared to those from sites values greater than the 3rd quartile and to those with values *between* the 1st and 3rd quartiles.

Table 5.1. The minimum, 1st quartile, mean, 3rd quartile, and maximum values for each of the five environmental variables by which *G. demissa* age distributions were compared.

Variable	minimum	1st quartile	mean	3rd quartile	maximum
Channel distance (m)	84	627	1,315	1,877	3,293
Bank elevation (m)	0.2	0.5	0.7	0.9	1.2
Channel width (m)	3	23	43	51	218
S. alterniflora dry weight (g m^{-2})	347	602	799	982	1,382
Soil bulk density (g cm^{-3})	0.15	0.28	0.36	0.41	0.81

Results

More than twice as many *G. demissa* were collected from natural creek edges than from dredged canal edges, with a total of 1,816 from natural sites and 600 from dredged sites. The density of mussels (± SE) at natural sites was 82 ± 12 m^{-2}, which was significantly greater than the density of 30 ± 9 m^{-2} at dredged sites (t-test, $p = 0.001$). The length frequency histogram of mussels from the natural sites (Figure 5.2) approximated a normal distribution, but the length frequency at dredged sites was skewed toward larger individuals.

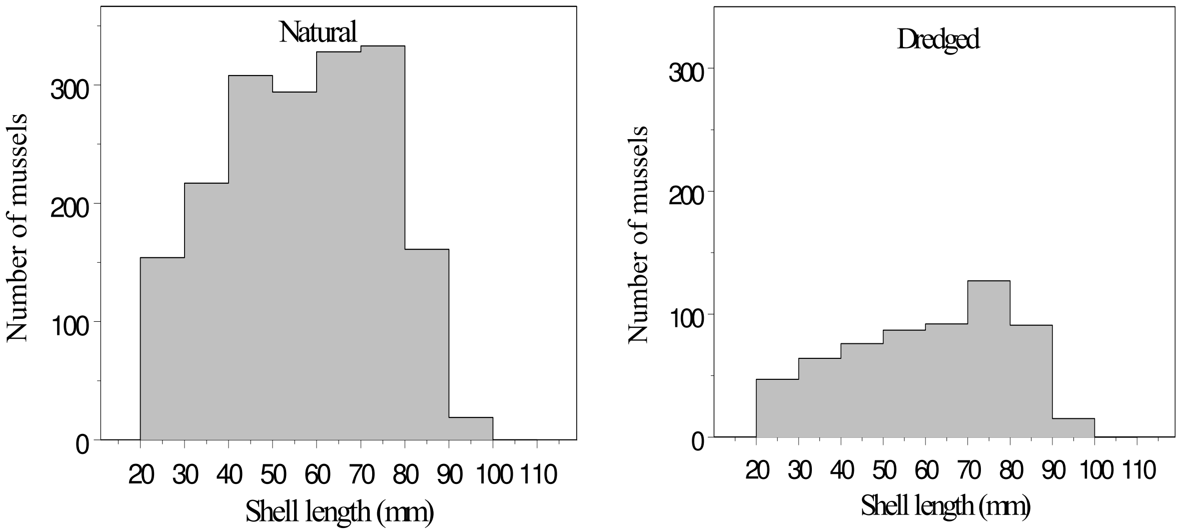

Figure 5.2. Shell length frequency histograms of mussels from marsh at natural creek edges (left) and dredged channel edges (right).

The exponential equation $y = 100*(1-0.65^x)$ was fit to the length to age curve (Figure 5.3) with an $r^2 = 0.97$. The equation

$$\text{Age} = \frac{\log((\text{length}-100) / 100)}{\log(0.65)}$$

was used to calculate the age distributions for mussels at natural and dredged sites (Figure 5.4). Thirty percent of the mussels at natural sites and 24 % at the dredged sites were aged 1 to 2 years; this age range accounted for the greatest proportion of mussels at both site types. The low number of mussels aged 0 to 1 years is a sampling artifact from only measuring mussels greater than 20 mm in length. Also, smaller mussels are more difficult to find in the sediment, and are therefore probably under-represented in the population sample. There is a constant attenuation in the number of mussels with increasing age at the natural sites. At dredged sites the number of mussel age 5 to 6 years is slightly greater than those aged 4 to 5 years.

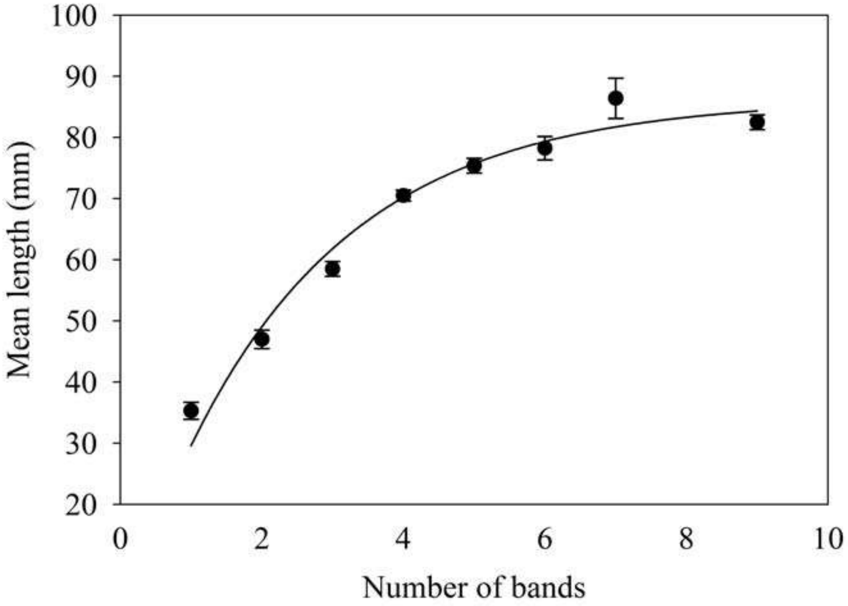

Figure 5.3. The relationship between the number of shell growth rings and mean length of *G. demissa*. The exponential equation $y = a(1-b^x)$ was fit to the data with an $r^2 = 0.97$.

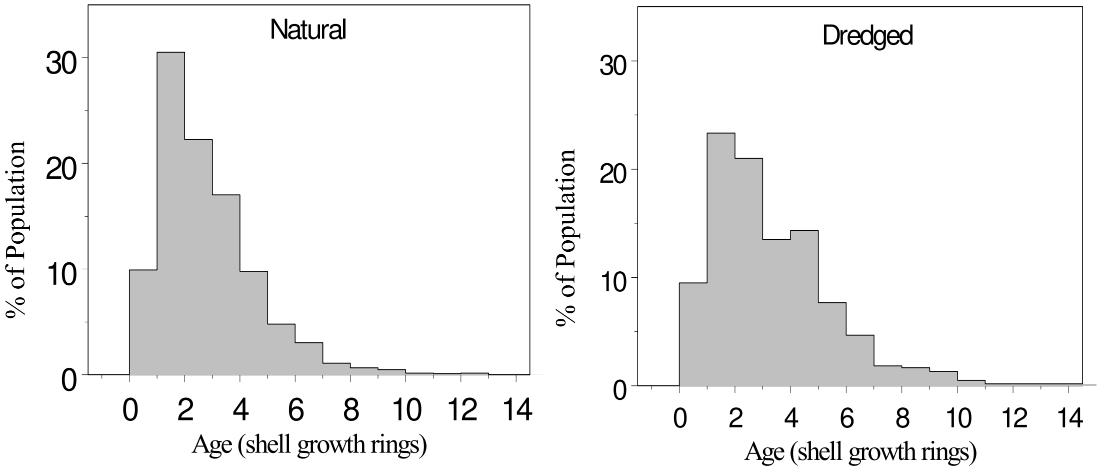

Figure 5.4. Age frequency histograms for mussels from natural (left) and dredged (right) systems calculated from age at length equation. This method assumes that 1 band equals one year of growth. The low number of 0 to 1 year aged mussels is probably because small sized mussels were missed during collection.

The age distribution at sites nearest channel outlets had a peak at ages 3 to 4 years, with 28 % of the population in this age range (Figure 5.5). The age distribution at sites farthest from channel outlets is skewed toward individuals aged 1 to 2 years, with a constant attenuation with age. The highest elevation sites had a more even distribution of mussels aged 1 to 5 years than the lowest elevation sites, whereas individuals aged 1 to 2 years were 31 % of the population (Figure 5.6). A similar pattern was observed for the widest versus narrowest sites. The mussels in the 1 to 2, 2 to 3, 3 to 4, and 4 to 5 year classes accounted for 19 to 20 % each at the widest channel sites; in contrast, 40% of mussels were aged 1 to 2 years at the narrowest channels (Figure 5.7). The lowest *S. alterniflora* dry weight sites had a low proportion of individuals that were aged 0 to 3 years and the majority of individuals were in the 3 to 4 year age class (Figure 5.8). The highest *S. alterniflora* dry weight sites had a high percentage of 1 to 2 year aged mussels, with constant attenuation with age, as did the lowest and highest bulk density sites (Figure 9). The sites with an intermediate bulk density had fewer individuals aged 1 to 2 years, than age 2 to 3 years, and no individuals that were 7 to 10 years in age.

Discussion

Counting external growth rings (annuli) of mussel shells as a measure of age has been reported to under-estimate the age of older individuals (Brousseau 1981), but the shell ring count to shell length curve did demonstrate the expected decrease in growth rate with size. Each growth ring may not truly represent one year, and rings were often difficult to distinguish, but this method did prove useful for comparing population structure across environmental conditions.

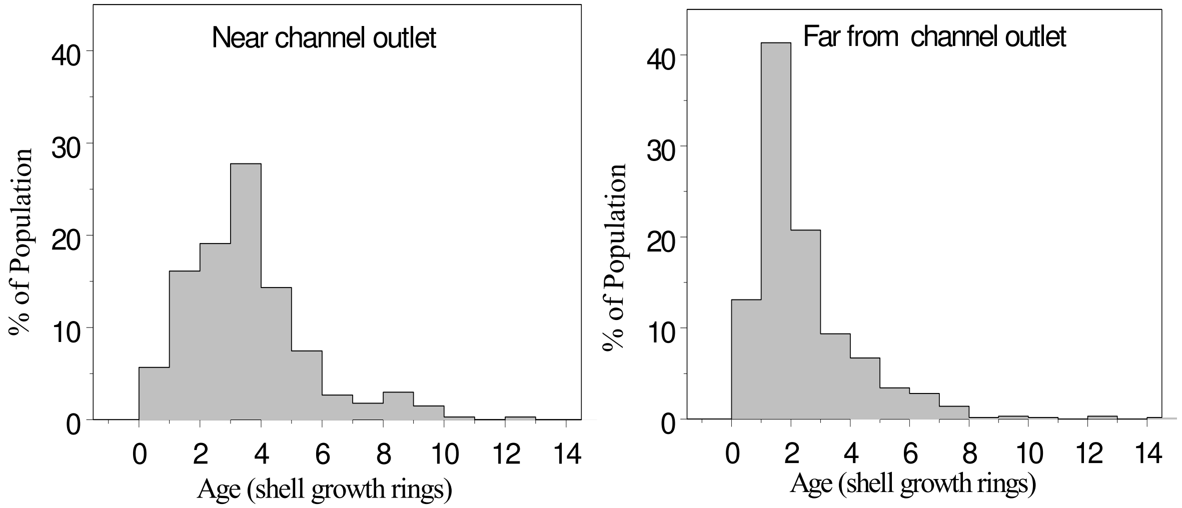

Figure 5.5. Left: Age frequency histograms of mussels from sites with a channel distance ≤ 627 m (1st quartile). Right: Age frequency histograms of mussels from sites with channel distance ≥ 1877 m (3rd quartile).

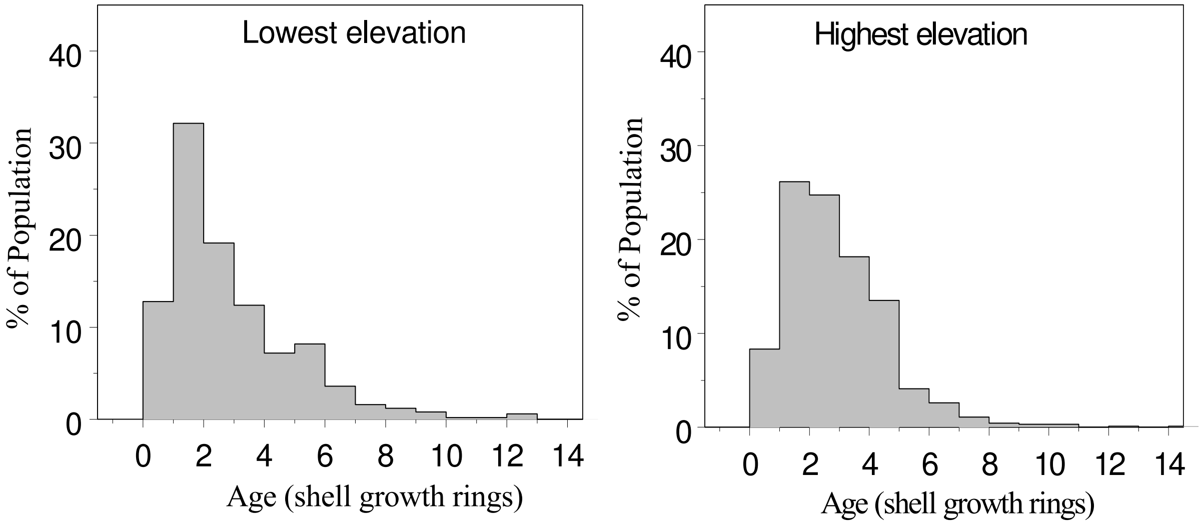

Figure 5.6. Left: Age frequency of mussels at sites with an elevation ≤ 0.5 m (1st quartile). Right: Age frequency of mussels at sites with an elevation ≥ 0.9 m (3rd quartile).

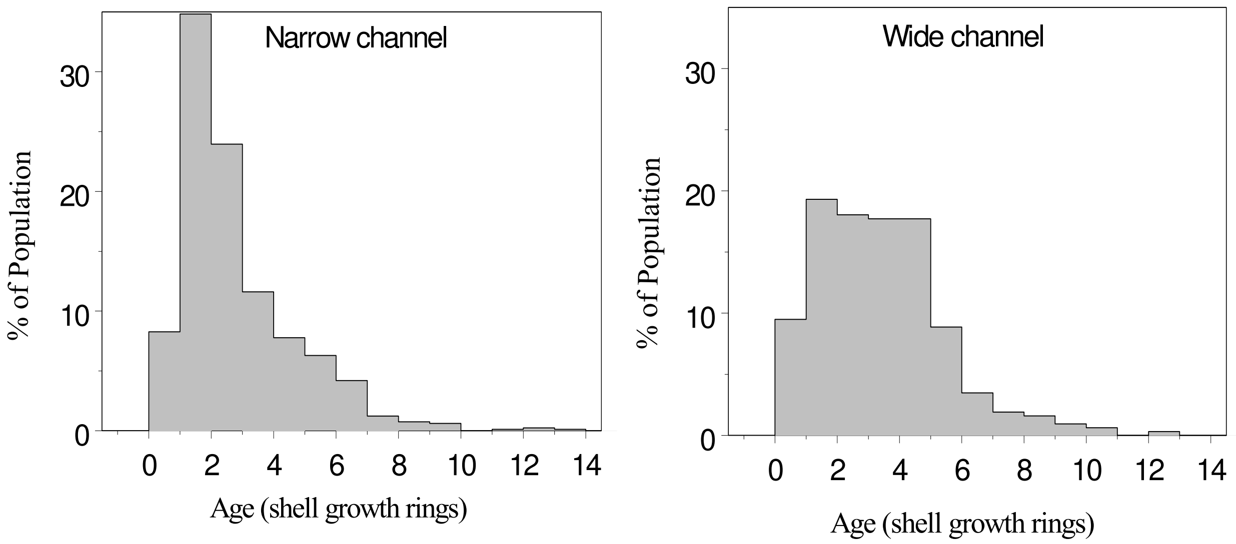

Figure 5.7. Left: Age frequency histograms of mussels from sites with a width ≤ 22.9 m (1st quartile). Right: Age frequency histograms of mussels from sites with a width ≥ 50.9 m (3rd quartile).

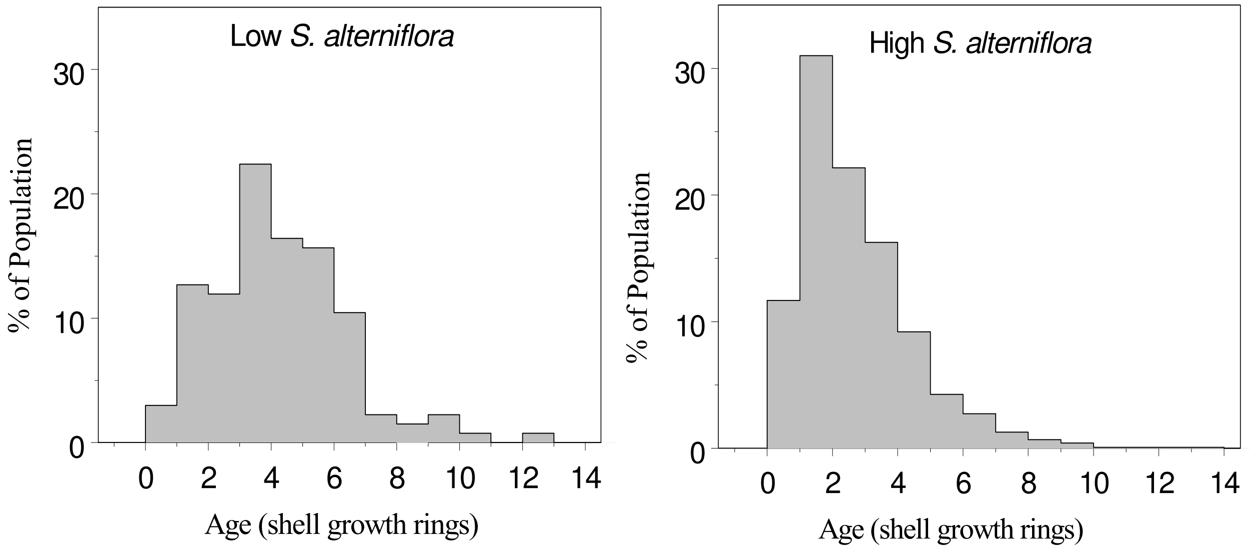

Figure 5.8. Left: Age frequency histograms of mussels from sites with *S. alterniflora* dry weight of ≤ 602 g m^{-2} (1st quartile). Right: Age frequency histograms of mussels from sites with *S. alterniflora* dry weight of ≥ 982 g m^{-2} (3rd quartile).

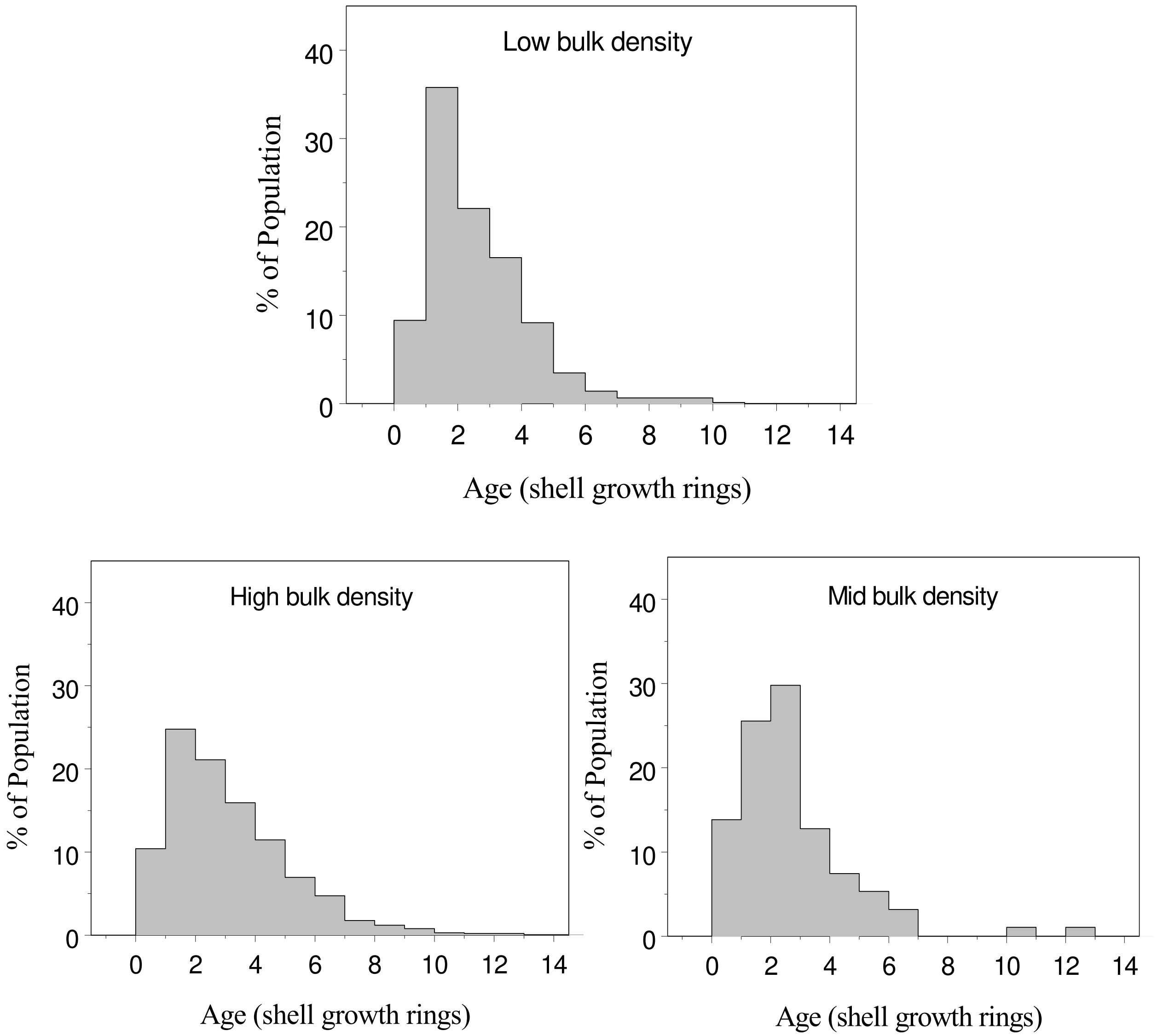

Figure 5.9. Top: Age frequency histograms of mussels from sites with a bulk density ≤ 0.28 g cm^{-3} (1^{st} quartile). Bottom left: Age frequency histograms of mussels from sites with a bulk density ≥ 0.41 g cm^{-3} (3^{rd} quartile). Bottom right: Age frequency histograms of mussels from sites with a bulk density > 0.28 and < 0.41 g cm^{-3}.

At dredged sites there were fewer mussels and the age distribution was skewed toward larger, older individuals compared to natural sites. The widest channel sites and lowest *S. alterniflora* sites also showed a similar lack of a peak in 1 to 2 year aged recruits. Dredged canals are typically wider than natural creeks, and significantly lower plant biomass was found at dredged canal edges than at creek edges (Chapter 4), so it is not surprising that these variables would show similar results. The attenuation in the number of mussels with increasing age at the natural sites suggests regular recruitment and mortality rates. At the dredged sites, there is a decrease in individuals in the 3 to 5 year age range. Predation could account for this decrease. Mussels in this age range are approximately 65 to 80 mm in length, and Hughes and Seed (1981) found that crabs consumed mussels up to 80 mm in length.

Sites farthest from the channel outlet and those with the smallest widths had the greatest proportion of the population that was individuals 1 to 2 years old. The population of mussels at the far ends of narrow channels appears to be made of mostly young small mussels and have fewer older individuals than the wider channels. This suggests high recruitment, and either high adult mortality or slower growth. Retention time may be a factor in high recruitment at distal ends of channel systems. A greater proportion of *G. demissa* larvae produced in a system will be retained through settlement if retention time exceeds the larval development period (Franz 2001). Hood (2002) showed that allochthonous detrital inputs are more likely to be retained in small sloughs and in the smaller and more distal portions of large sloughs. In wider channels closer to the outlet, recruitment may be lower due to a decrease in retention time, but adult survival or growth may be greater. Bertness (1980), for example, found that mussels showed progressively decreasing growth rates and maximum size from the open coast to the more severe estuarine environment.

Age frequencies for lowest and highest elevations did not differ greatly. Previous studies have shown that for mussels living at the seaward edge of the marsh, those from lower elevation displayed substantially higher growth rates than their counterparts at higher elevation (Lin 1989). Elevation differences between sites may not have been great enough to see a difference.

The results of a previous analysis of mussel biomass at natural and dredged sites suggested there is an optimal range for *G. demissa* recruits of 0.25 to 0.4 g cm^{-3} (Chapter 4). Mussel distributions were compared at low, medium and high bulk density sites. Based on the histograms, low bulk density sites appear to have the greatest proportion of recruits. Medium bulk density sites had a greater proportion of individuals age 2 to 3 years, and also had mussels older than 10 years. The finding of higher mussel biomass at the mid-bulk density sites may not be a result of higher recruitment due to better soil properties, but soil properties may contribute to the longevity of individuals that do recruit to sites with bulk density values in this range.

Conceptual Model

A conceptual model (Figure 5.10) was created to link structural differences between natural creek systems and dredged canal systems to population processes in *G. demissa*. Information used in developing the model came from studies of natural and dredged channel shape complexity (Chapter 2), marsh elevation adjacent to natural and dredged systems (Chapter 3), collections of *G. demissa*, *S. alterniflora*, and soil samples from random dredged and natural channel marsh edge (Chapter 4), and from the population structure analysis presented in the paper, as well as information from other sources of published literature on *G. demissa* and *S. alterniflora*. The box numbers in parenthesis in the following text refer to the boxes numbers in Figure 4.10.

Structural differences between natural creek systems and dredged canal systems include changes in the shape of the channel system and in channel bank properties. Natural creek networks were found to be significantly more complex in shape (box 1) through the analysis of perimeter-to-area ratio, shape index, and patch fractal dimension of 5 natural and 5 dredged channel systems (Chapter 2). The bank elevation (box 2) of marsh adjacent to natural creeks was significantly lower than dredged canals. Natural creek levels measured using LiDAR elevation data averaged 10 m in width and had a maximum height of 0.85 m, while the elevated area of marsh adjacent to dredged canals examined extended approximately 30 m from the channel edge and the mean highest elevation of 0.98 m occurred at 10 m from the edge of canals (Chapter 3). The mean channel width (box 3) measured at random points was more than twice that of natural creeks, and substrate bulk density (box 4) was lower at natural marsh edges than at dredged canals (Chapter 4).

Recruitment of mussels to the marsh edge (box 5) is affected by structural changes in the shape of channel systems, elevation of adjacent marsh, and substrate properties at the marsh edge. Shape complexity can regulate both the amount of edge and quantity of mussel spat available for settlement. The amount of edge available for settlement box 6) is directly related to the shape index (Chapter 2). The amount of mussel spat available for settlement is partially determined by the retention time (box 7) of materials in the channel system. Compared to dredged networks, natural networks had a significantly greater percent channel area that was < 5m from the channel edge (Chapter 2). This result can be attributed to the presence of small side channels that are < 10m wide in natural channels that are rarely present in the dredged channels examined in this study. Retention time may therefore be greater in natural creek systems, which would create a higher density of spat available for settlement. The greater proportion of age 1 to 2 year individuals at the sites that had the narrowest channel widths and in those that were furthest from the channel outlet seems to suggest this.

Inundation time (box 8) controls how often the marsh edge is available to settling spat. A wider, higher bank at a dredged canal means the marsh behind it may rarely be flooded enough for recruits to access it during flooding tides. Jordan and Valiela (1982) hypothesized that the abundance of adult mussels must be determined ultimately by the settlement and survival of spat because mortality of large mussels is very low.

Inundation also affects the growth (box 9) and reproductive effort (box 10) of mussels (Franz 1993). *G. demissa* feed by filtering primarily plankton from the water column during flooding tide (Peterson and Howarth 1987), but *Geukensia* can also feed on locally generated food sources such as *Spartina* detritus and suspended benthic algae (Peterson et al. 1985). Growth rates may be directly linked to submergence time, although Lin (1989) found that the magnitude of differences in mussel growth rates between low and high tidal elevations were larger than the quantitative differences in average submergence time. Variability in the quantity and quality of food between habitats may be an important factor, but ultimately would also be governed by physical conditions (Bertness 1980). Mussels growing at higher tidal levels are exposed to

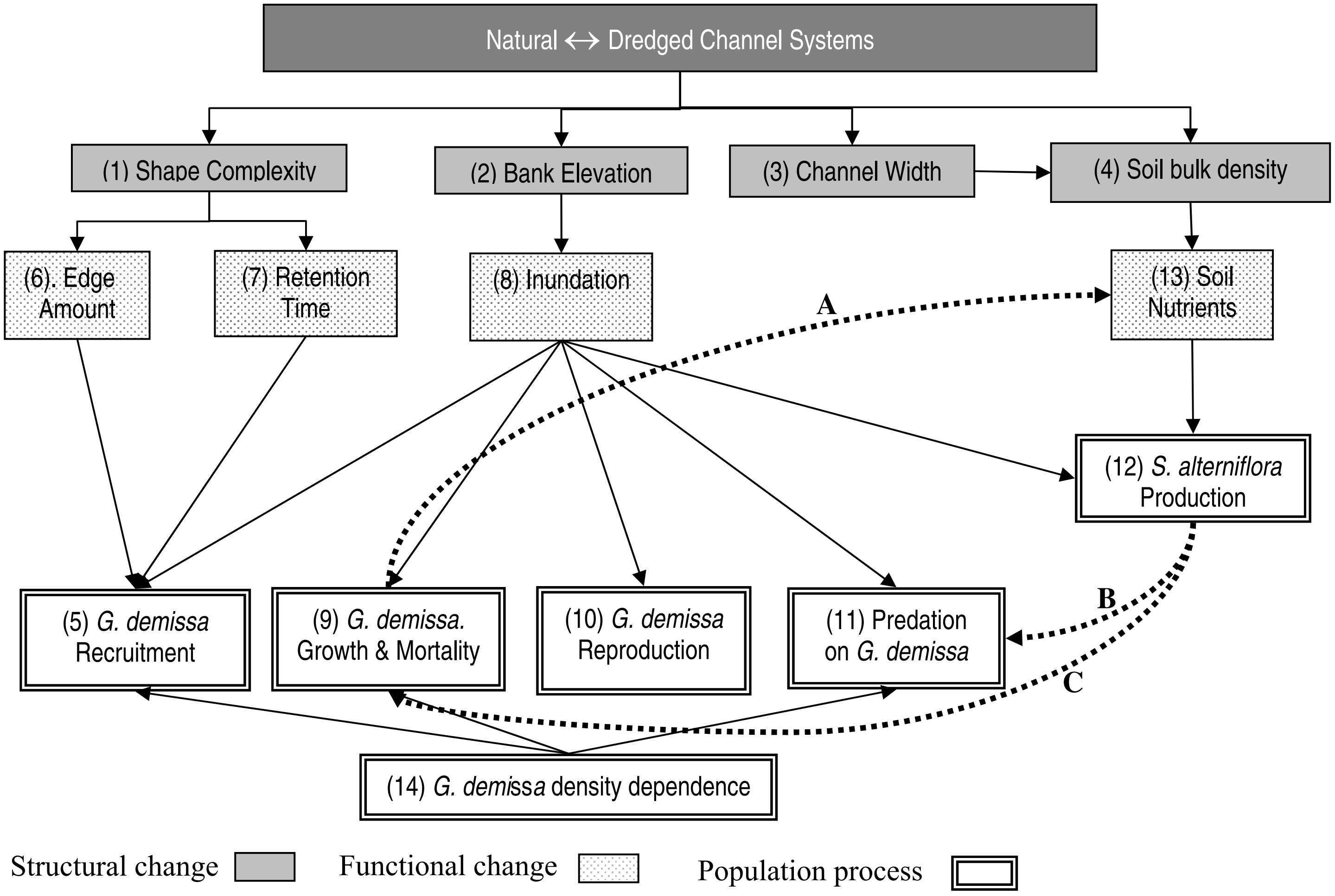

Figure 5.10. Conceptual model linking structural differences between natural and dredged marsh channel systems to population processes in *G. demissa* and production of *S. alterniflora*. The numbers in parentheses and letters A, B, and C refer to process descriptions in the text.

potential desiccation and thermal stress (Lent 1969), and may allocate a smaller proportion of total production to growth, but exceed the reproductive effort of lower-shore populations (Franz 1997). Mussels growing lower in the tidal zone may be more susceptible to predation (box 11) by blue crab.

Inundation is also a limiting factor in the production of *S. alterniflora* (box 12). The growth of *S. alterniflora* is controlled to some extent by the relative oxidation state of the sediments (Howes et al. 1981). *S. alterniflora* growth is also limited by nutrient availability in the soil (box 13), which is affected by the structural parameter of soil organic matter. Soil bulk density was found to be significantly higher in dredged canals than in natural creeks, which may be the result of sediment source since dredged spoil banks are made from sediment dredged from the channel bottom (Chapter 4). Channel width may also affect the percentage of organic material (Hood 2002).

Density-dependent recruitment and survival are represented by box 14. It has been suggested that the recruitment of mussels to the marsh edge is correlated primarily with the amount of available substrate composed of other living mussels (Nielsen and Franz 1995). The number of mussels gathered that were 20 to 25 mm in length were compared to number of mussels that were > 25 mm in length (Figure 5.11) to determine if the number of recruits in the Barataria marsh was related to the number of conspecifics, and there does appear to be a weak relationship between the numbers of recruits and established mussels. The densities of mussels also tends to increase with decreasing elevation (Jordan and Valiela 1982), and evidence of density-dependent mortality (Bertness 1980) suggests that intraspecific competition may be a significant mortality factor at lower elevation sites.

The relationship between *G. demissa* and *S. alterniflora* may be one of facultative mutualism. Mussels increase soil nitrogen (line A) through excretion and pseudofeces deposition. Bertness (1984) showed that the experimental removal of mussels from *Spartina* marsh plots decreased the soil nitrogen by 0.18 %, and in the Great Sippewissett Marsh the mussel population has the highest biomass of any animal population and releases more ammonia than any population of either plants or animals. Since nitrogen limits productivity in the salt marsh, increased retention of nitrogen due to filtration by mussels may ultimately enhance the productivity of the marsh (Jordan and Valiela 1982). The increased *Spartina* density and biomass can reduce predation on mussels (line B) by producing a firm base for attachment of the byssi (Kuenzler 1961), and increased mussel growth (line C) by providing an additional food source (Kreeger et al. 2000), as well as by serving as a shield against the hot summer sun (Kuenzler 1961).

Summary

The examination of mussel populations by age distribution provides evidence that structural differences between natural creeks and dredged canal systems can influence the recruitment, growth, and mortality of mussels. Channel distance, channel width, and *S. alterniflora* standing biomass affected the age distribution of mussels, and all three parameters may differ between natural and dredged systems. The conceptual model presented in this paper is a first step toward developing a predictive model of how landscape change will affect mussel populations. The relationships between retention time and recruitment, inundation time and growth and reproduction, and soil bulk density and growth and mortality need further investigation. The model could also be refined by the incorporation of density dependent growth and mortality.

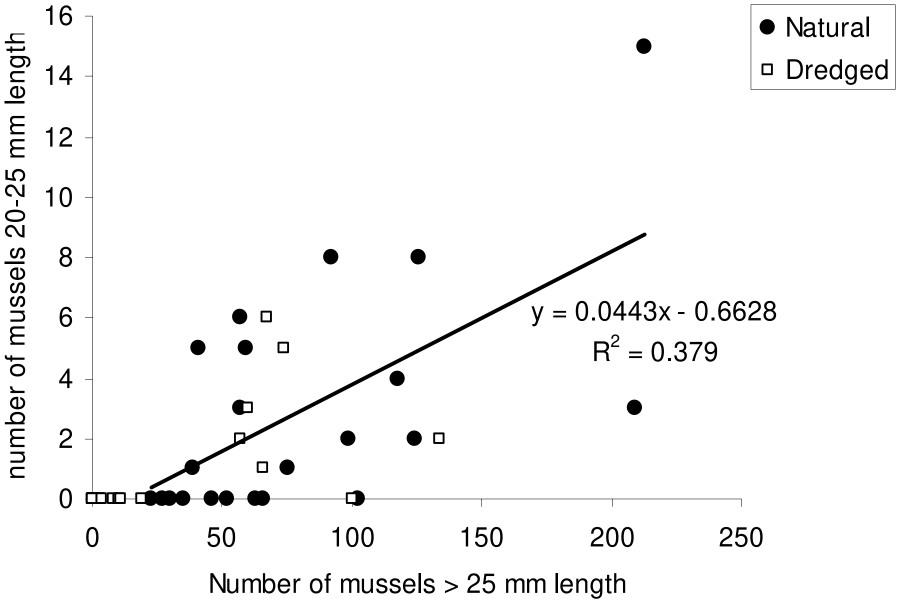

Figure 5.11. The number of recruits (mussels 20 – 25 mm in length) compared to the number of individuals considered to be established (greater than 25 mm in length). The regression line is for natural and dredged site results combined.

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CHAPTER 6

SUMMARY

I used GIS methods and LiDAR data to analyze differences in the shape complexity of natural and dredged channel networks and to compare the elevation of the marsh adjacent to natural and dredged channels and productive and non-productive oil and gas wells in a tidal salt marsh in Louisiana. I then examined the distribution of *Spartina alterniflora*, *Geukensia demissa demissa*, and *Littoraria irrorata* in relation to channel morphology, soil properties and elevation of the marsh edge at natural creeks, dredged canals, and open bay edges in order to correlate landscape patterns with ecological processes.

Dredged canals were significantly less complex in shape, contained less channel habitat that was less than 5 m from the marsh edge, and had lower drainage area indices than natural creeks systems. The marsh around dredged canals appears to be more broken up, but not lower in elevation. Densities of *S. alterniflora* and *G. demissa* were lower, and soil and leaf nitrogen were significantly less in marsh adjacent to dredged canals than in marsh adjacent to natural creeks.

Shape complexity affects the amount of edge available and the area of marsh that can be accessed from the channel, and may also have an effect on the retention time of particles in the channel system, current speeds, and habitat diversity. The results of the spatial heterogeneity analysis support the idea that channel shape affects microhabitat distribution. Nekton populations may be affected by channel shape on a microhabitat level if channels with less complex edges provide less variability in edge structure. The application of the habitat patch metrics of perimeter-to-area ratio, shape index, and patch fractal dimension to tidal marsh channel systems was useful for identifying differences in shape complexity between natural creeks and dredged channels. A more detailed analysis of fine scale bank slope, channel cross section profiles, and elevation would provide a better understanding of structural differences between natural and dredged channels. The tidal channel network and the adjacent marsh edge actually are a mosaic of many microhabitat patches. Dredged channels have a reduced shape complexity over natural systems, which likely results in reduced habitat diversity. The organisms that have evolved to survive in the difficult environment of the tidal marsh are dependent upon specific inundation regimes; a reduction in habitat diversity may remove species niches along the inundation gradient.

The results of both the transect and buffer methods of measuring marsh elevation surrounding natural creeks and dredged channels demonstrated that there is significantly more open water in dredged than in natural marshes. The formation of open water areas within the marsh was near dredged canals, but the mechanism of marsh-to-water conversion does not appear to involve an intermediate decrease in elevation. The results of this study are consistent with the pattern of marsh loss which would result from the internal break up of the marsh into open water ponds. There was no evidence found in this study to support the theory that deep sub-surface fluid withdrawal affects surface elevation. The mechanism and time scale of the marsh-to-water transition needs further analysis. A time series of LiDAR data would be useful in further exploring associations between elevation changes and marsh loss, and could be combined with “on the ground” studies of plant productivity to better understand marsh-to-water conversions.

Nitrogen limitation may be one cause of the lower biomass values of *S. alterniflora* at dredged compared to natural sites; there was not sufficient data to determine if salinity and hydroperiod differences affected plant growth. The greater *G. demissa* density at natural marsh sites could indicate better food resources, less predation on settled mussels, or higher rates of settlement than at dredged marshes. The examination of mussel populations by age distribution provided evidence that structural differences between natural creeks and dredged canal systems influences the recruitment, growth, and mortality of mussels. Channel distance, channel width, and *S. alterniflora* production affected the age distribution of mussels, and all three parameters differ between natural and dredged systems. At dredged sites there were fewer mussels and the age distribution was skewed toward larger, older individuals compared to natural sites. The populations of mussels at the far ends of narrow channels appear to be made of mostly young, small mussels and have fewer older individuals than the wider channels. This suggests that high recruitment, and either high adult mortality or slower growth occurs in the distal ends of natural creek, while low recruitment and low adult mortality occurs along dredged canals. The relationships between retention time and recruitment, inundation time and growth and reproduction, and soil bulk density and growth and mortality need further investigation in order to continue development of a model that can predict how mussel populations will change as the marsh landscape is changed.

This work demonstrates that landscape scale changes to tidal salt marshes have the potential to alter ecological processes at the microhabitat scale. The dredging of massive canal networks is no longer a common practice in Louisiana marshes, but it is important to understand how these artificial systems compare to natural ones so that we do not underestimate the impact dredged systems continue to have on the ecosystem long after they are created. Understanding these impacts and the relationships between the natural hydrology and habitat quality will aid in the restoration of these salt marshes.

VITA

Jennifer Spicer was born in Portland, Maine, to Robert and Carolyn Spicer. She grew up on the coast of Maine with her older brother Jason, and graduated from Marshwood High School in 1993. She then moved to South Carolina to study marine science at Coastal Carolina University. Her Senior Honors Thesis examined vegetative growth in a wetland creation project in Georgetown, South Carolina. She also worked as a microbiology laboratory technician and as an ecology student assistant while at CCU. She graduated with a Bachelor of Science degree in marine science and biology, with minors in chemistry and environmental science in 1998, and became a Research Technician at the University of Georgia Marine Institute on Sapleo Island, Georgia. Jennifer began graduate studies at Louisiana State University in 2001 in the Department of Oceanography and Coastal Sciences. While at LSU, she was active in the graduate student group Marine Environmental Researchers, and helped to organize the 6th annual Graduate Student Symposium. She also worked as a Graduate Assistant on a project to develop educational materials about the ecology of the Pacific Coast of Mexico. She moved back to South Carolina in 2006 to work as the Stewardship Coordinator at the North Inlet-Winyah Bay National Estuarine Research Reserve. The degree of Doctor of Philosophy will be awarded to her at the 2007 December Commencement.