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Causes of Vegetation Dieback and Subsequent Wetland Loss in Two Louisiana Coastal Marshes.

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CAUSES OF VEGETATION DIEBACK
AND SUBSEQUENT WETLAND LOSS
IN TWO LOUISIANA COASTAL MARSHES

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
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TABLE OF CONTENTS

LIST OF TABLES	iv
LIST OF FIGURES	v
ABSTRACT	vii
CHAPTER 1: INTRODUCTION	1
<u>Physical Processes</u>	1
<u>Biological Processes</u>	3
<u>References</u>	6
CHAPTER 2: FACTORS ASSOCIATED WITH VEGETATION DIEBACK IN A LOUISIANA SALT MARSH: A THREE-YEAR FIELD STUDY	9
<u>Introduction</u>	9
<u>Methods</u>	10
Study Site Location.....	10
Sampling Design.....	11
Sampling Procedure.....	11
Statistical Analysis.....	14
<u>Results</u>	14
Live Biomass and Plant Cover.....	14
Redox Potential, Interstitial Sulfide Concentration and pH.....	15
Macronutrient Concentration.....	20
Salinity.....	20
Soil Bulk Density.....	20
Correlation Analysis.....	25
<u>Discussion</u>	26
Healthy vs. Deteriorating Marsh Sites.....	26
Pipeline Canal Effects.....	30
<u>Conclusion</u>	30
<u>References</u>	31
CHAPTER 3: CAUSES FOR VEGETATION DIEBACK IN A LOUISIANA SALT MARSH: A BIOASSAY APPROACH	35
<u>Introduction</u>	35
<u>Methods</u>	36
Study Area.....	36
Experimental Design.....	37
Variables Sampled.....	38
Statistical Analysis.....	39
<u>Results</u>	40
Biotic Response.....	40
Abiotic Response.....	40
<u>Discussion</u>	43
<u>References</u>	46
CHAPTER 4: AN INVESTIGATION OF THE FACTORS ASSOCIATED WITH VEGETATION DIEBACK IN A LOUISIANA OLIGOHALINE MARSH	49

<u>Introduction</u>	49
<u>Methods</u>	50
Study Site Location.....	50
Sampling Procedure.....	52
Statistical Analysis.....	53
<u>Results</u>	54
<u>Discussion</u>	65
Reference Marsh vs. Deteriorating Marshes.....	65
Pipeline Canal Effects.....	69
<u>Conclusion</u>	69
<u>References</u>	70
 CHAPTER 5: FACTORS AFFECTING VEGETATION DIEBACK OF AN OLIGOHALINE MARSH IN COASTAL LOUISIANA: FIELD MANIPULATION OF SALINITY AND SUBMERGENCE	73
<u>Introduction</u>	73
<u>Methods</u>	75
Study Area.....	75
Experimental Design.....	76
Variables Sampled.....	77
Statistical Analysis.....	78
<u>Results</u>	78
<u>Discussion</u>	84
<u>References</u>	88
 CHAPTER 6: CONCLUSION	91
<u>References</u>	94
 APPENDIXES: LETTERS OF PERMISSION	96
 VITA	99

LIST OF TABLES

Table 2.1. Correlation matrix between biotic and abiotic variables. Correlation coefficient and significance level are given for each correlation. Asterisk (*) indicates a correlation significant at the 0.05 level.	25
Table 3.1. Interstitial sulfide (mM) and soil macro- and micronutrient concentrations (μ M), pH and salinity (ppt). Mean values given for each elevation; parentheses indicate 1 standard error. Different letters following values indicate a significant difference at the 0.05 probability level.	43
Table 4.1. Relative plant species composition by marsh site for all plant species with more than one occurrence. Dominance for each plant species given as mean percentage of total biomass; 1 standard error of the mean in parentheses.....	56
Table 5.1. Interstitial, pH, salinity (ppt) and interstitial soil cation concentration (mM). Mean values given for each marsh location (DM=donor marsh, RM=recipient marsh, RP=recipient pond); parentheses indicate 1 standard error. Different letters following values indicate a significant difference at the 0.05 probability level.....	84

LIST OF FIGURES

Figure 2.1 Location of reference marsh site (HNP, n=14), deteriorating marsh site bisected by backfilled pipeline canals, indicated with dashed lines, (DP, n=14), and deteriorating marsh site without pipeline canals (DNP, n=14).....	12
Figure 2.2. Aboveground live biomass for each marsh site by sampling date. Bars represent mean live biomass \pm 1 standard error.....	16
Figure 2.3. Live vegetation cover by marsh site. Bars represent mean of all sampling dates \pm 1 standard error.....	17
Figure 2.4. Redox potential at 2 cm depth (a) and at 15 cm depth (b) by marsh site for each sampling date. Bars represent mean \pm 1 standard error.	18
Figure 2.5. Interstitial sulfide concentration by marsh site averaged across all sampling dates (a) and by sampling date averaged across all marsh sites (b). Bars represent mean \pm 1 standard error.....	19
Figure 2.6. Interstitial NH ₄ -N concentration by marsh site averaged across all sampling dates (a) and by sampling date averaged across all marsh sites (b). Bars represent mean \pm 1 standard error.....	21
Figure 2.7. Interstitial P concentration by marsh site averaged across all sampling dates. Bars represent mean \pm 1 standard error.	22
Figure 2.8. Interstitial salinity concentration by sampling date averaged across all marsh sites. Bars represent mean \pm 1 standard error.	23
Figure 2.9. Soil bulk density by marsh site for each sampling date. Bars represent mean \pm 1 standard error.....	24
Figure 3.1. Total standing crop (biomass + necromass) at elevated and ambient elevations for all four plant species. Plant species: DS= <i>Distichlis spicata</i> , JR= <i>Juncus roemerianus</i> , SA= <i>Spartina alterniflora</i> and SP= <i>Spartina patens</i> . (mean \pm 1 standard error).	41
Figure 3.2. Redox potential (mV) at 2 cm depth (a) and at 15 cm depth (b) for elevated and ambient elevations for all four plant species. Plant species: DS= <i>Distichlis spicata</i> , JR= <i>Juncus roemerianus</i> , SA= <i>Spartina alterniflora</i> and SP= <i>Spartina patens</i> . (mean \pm 1 standard error).....	42
Figure 4.1. Location of reference marsh site (HA, n=8), deteriorating marsh site bisected by backfilled pipeline canals (DB, n=8), and deteriorating marsh site without backfilled pipeline canals (DC, n=8).....	51

Figure 4.2. Aboveground live biomass by marsh site averaged across all sampling dates (a) and by sampling date averaged across all marsh sites (b). Bars represent mean live biomass \pm 1 standard error.	55
Figure 4.3. Aboveground live biomass by marsh site averaged across all sampling dates for <i>S. lancifolia</i> (a), <i>S. patens</i> (b), and <i>P. vaginatum</i> (c). Bars represent mean live biomass \pm 1 standard error.	57
Figure 4.4. Redox potential at 2 cm depth (a) and at 15 cm depth (b) by marsh site for each sampling date. Bars represent mean \pm 1 standard error.	59
Figure 4.5. Interstitial sulfide concentration by marsh site for each sampling date. Bars represent mean \pm 1 standard error.	60
Figure 4.6. Interstitial NH ₄ -N concentration by sampling date averaged across all marsh sites. Bars represent mean \pm 1 standard error.	61
Figure 4.7. Interstitial P concentration by marsh site for each sampling date. Bars represent mean \pm 1 standard error.	62
Figure 4.8. Interstitial pH by marsh site for each sampling date. Bars represent mean \pm 1 standard error.....	63
Figure 4.9. Interstitial salinity by marsh site for each sampling date. Bars represent mean \pm 1 standard error.....	64
Figure 4.10. Soil bulk density by marsh site for each sampling date. Bars represent mean \pm 1 standard error.....	66
Figure 5.1. Live biomass by elevation for each marsh location. DM=Donor marsh location, RM=Recipient marsh location and RP=Recipient pond location. Bars represent mean \pm 1 standard error.	80
Figure 5.2. Redox potential (mV) at 2 cm depth (a) and at 15 cm depth (b) by elevation type for each marsh location. DM=Donor marsh location, RM=Recipient marsh location and RP=Recipient pond location. Bars represent mean \pm 1 standard error.	81
Figure 5.3. Interstitial sulfide concentration (mM) by elevation type for each marsh location. DM=Donor marsh location, RM=Recipient marsh location and RP=Recipient pond location. Bars represent mean \pm 1 standard error.	82
Figure 5.4. Interstitial NH ₄ -N (a) and P (b) concentration (μ M) by elevation type for each marsh location. DM=Donor marsh location, RM=Recipient marsh location and RP=Recipient pond location. Bars represent mean \pm 1 standard error.	83

ABSTRACT

Vegetation dieback is an important component of the wetland loss process in coastal Louisiana. To determine the factors causing vegetation dieback in a Louisiana salt marsh and oligohaline marsh, field monitoring and manipulative field experiments were conducted in both marsh types. In the salt marsh, a three-year field study was conducted in which biotic and edaphic characteristics were measured and the effect of elevation on four salt marsh plant species was tested in a manipulative field experiment. In the oligohaline marsh, a two-year field study was conducted which evaluated the relationship between biotic and edaphic characteristics. Additionally, the effects of increased salinity, increased submergence and their combination on oligohaline marsh vegetation was determined in a manipulative field experiment.

Deteriorating salt marshes had more reduced soil conditions and higher sulfide and $\text{NH}_4\text{-N}$ concentrations (which likely indicates decreased $\text{NH}_4\text{-N}$ uptake by vegetation) than a nearby reference marsh. Soil bulk density and interstitial salinities did not differ between marsh sites. When salt marsh vegetation was elevated 20 cm, plant growth improved and soils became less reduced. Additionally, salt marsh soils elevated 20 cm had lower interstitial sulfide and $\text{NH}_4\text{-N}$ concentrations.

In an oligohaline marsh, deteriorating sites had higher interstitial salinities, sulfide and $\text{NH}_4\text{-N}$ concentrations than the nearby reference marsh. Deteriorating oligohaline marshes had a change in species composition from a *Sagittaria lancifolia*-dominated community to a brackish marsh community dominated by *Spartina patens* and *Paspalum vaginatum*. When the elevation of *S. lancifolia*-dominated marsh vegetation was decreased 15 cm, soils became more reduced and plant growth decreased. The combination of a decrease in elevation and saltwater intrusion caused the most dramatic decrease in growth of *S. lancifolia*-dominated vegetation as well as causing a significant

increase in interstitial sulfide and $\text{NH}_4\text{-N}$ concentrations. An increase in salinities alone had no effect on the health of the *S. lancifolia*-dominated community.

In both the salt marsh and the oligohaline marsh, increases in marsh elevation improved soil conditions and caused increased growth of marsh vegetation. To be successful, restoration activities in coastal Louisiana should focus on increasing marsh elevation, even with the threat of saltwater intrusion.

CHAPTER 1

INTRODUCTION

The Louisiana coastal marshes represent 40% of the coastal wetlands of the conterminous United States (Turner, 1990). Louisiana is losing its coastal marshes at an astonishing rate with current wetland loss estimates as high as $65.6 \text{ km}^2 \text{ y}^{-1}$ (Dunbar et al., 1992). Some of this marsh loss can be attributed to shoreline erosion or anthropogenic disturbances such as canalization but 67% of wetland loss in coastal Louisiana occurs within the interior of marshes (Wayne et al., 1994). The physical processes responsible for coastal wetland loss have been well studied, but the biological processes controlling wetland loss are not as well understood.

Physical Processes

During the past 10,000 years the Mississippi River has built the present deltaic plain in coastal Louisiana through a series of overlapping delta lobes (Fisk, 1944; Fisk and McFarlan, 1955; Scruton, 1960; Frazier, 1967; Coleman, 1988). Since the last glaciation, the Mississippi River has changed course several times depositing sediments upon the shallow continental shelf of the Gulf of Mexico creating numerous, overlapping delta lobes. As a new delta lobe is formed, the river course becomes longer and more sinuous and is eventually abandoned for a more efficient course. For a period of time, river flow may continue down both channels, but eventually the older delta lobe is abandoned. Subsidence due to sediment compaction becomes the dominant process within the abandoned delta lobe and the marsh surface elevation decreases. Open water bodies replace vegetated marsh within the abandoned delta. During the time period that the older delta lobe is in a deterioration phase, a new delta lobe is being formed elsewhere along the continental shelf. Through time, the formation of multiple delta lobes through course changes of the Mississippi River has maintained a relative balance

between the creation and loss of coastal wetlands within the Mississippi River deltaic plain.

The balance between the creation and loss of wetlands within the deltaic plain has been interrupted by anthropogenic changes to the river course. River suspended sediment load has decreased nearly 80% since 1850 (Kesel, 1987; Kesel, 1989). The decrease in the Mississippi River sediment load is due to dams on major tributaries as well as landuse changes within the Mississippi River drainage basin (Keown et al., 1986; Kesel, 1989). Overbank flooding of the Mississippi River and its distributaries likely provided the majority of the sediment input to coastal marshes in the deltaic plain. Overbank flooding from the Mississippi River has been retarded by leveeing its entire length to the Gulf of Mexico. As well as the levee system, the construction of the Old River Control Structure in 1963 has maintained the Mississippi within its current course, even though it would have naturally abandoned its course for the more efficient Atchafalaya River by 1975 (Fisk, 1952; Roberts et al., 1980). The changes to the natural Mississippi River delta cycle have caused the sediment deposition and freshwater discharge to occur off the continental shelf into deep water.

The high rate of wetland deterioration in coastal Louisiana is directly linked to the physical processes associated with the delta cycle. Since the Mississippi River's current course is contained, much of the Louisiana deltaic plain is within one of the abandoned delta lobes. The physical processes that dominate within the abandoned delta lobes are subsidence, which is caused by sediment consolidation, settlement, geochemical processes and faulting (Penland et al., 1994). Because of the high rate of subsidence combined with the eustatic sea level rise of the Gulf of Mexico, the Louisiana deltaic plain has one of the highest rates of relative sea level rise in the United States (Penland et al., 1988; Penland and Ramsey, 1990). A combination of the high rate of relative sea level rise and the sediment accretion deficit has caused the Louisiana coastal marshes to

experience increased depths and duration of flooding as well as greater oceanic influences. These are the physical changes responsible for the high rate of wetland deterioration in coastal Louisiana.

Biological Processes

Although the physical processes of wetland deterioration are well studied, vegetation dieback, the degeneration and death of all aboveground plant tissue leaving unvegetated, intertidal mud flats (Goodman et al., 1959) which is an important component of the wetland loss process, is not as well understood. The main causes believed to be responsible for vegetation dieback within the coastal zone of Louisiana are increased plant submergence, saltwater intrusion, the combination of these two processes, and an inadequate supply of mineral sediments.

Increased depth and duration of marsh flooding has been shown to adversely affect a number of wetland plant species. *Spartina alterniflora*, the dominant salt marsh plant species in coastal Louisiana, has decreased productivity in more waterlogged soil conditions (Mendelssohn and Seneca, 1980; Howes et al., 1981; Mendelssohn and McKee, 1988). Additionally, plant species that dominate lower salinity marshes such as *Spartina patens*, *Sagittaria lancifolia*, *Panicum hemitomon* and *Leersia oryzoides* also experience decreased productivity with increased flooding conditions (McKee and Mendelssohn, 1989; Burdick et al., 1989).

Saltwater intrusion is commonly believed to be a cause of vegetation dieback in coastal marshes of Louisiana, but little supporting data exists. A northward movement of more saline marsh types since 1949 has been found using historical maps of dominant marsh vegetation (Chabreck and Linscombe, 1982). *Sagittaria lancifolia*, a dominant plant species of Louisiana tidal freshwater and intermediate marshes, has decreased growth at salinities as low as 4.8 ppt and *P. hemitomon* and *L. oryzoides* can tolerate salinities as high as 9.4 ppt under greenhouse conditions (McKee and

Mendelssohn, 1989). After 30 to 40 days of exposure at salinities of 3 ppt, net carbon assimilation in *S. lancifolia* was reduced 45% (Pezeshki et al., 1987a) and after a seven-day exposure, salinities between 5-7 ppt caused decreased stomatal conductance and photosynthesis in *P. hemitomon*, a common freshwater marsh plant species in coastal Louisiana (Pezeshki et al., 1987b).

Although increased plant submergence and saltwater intrusion independently can be detrimental to vegetation, collectively the two stresses are extremely damaging to many wetland plant species. Increased soil waterlogging decreases the rate of oxygen diffusion through waterlogged sediments (Greenwood, 1961) and, in part, cause more reduced soil conditions (Gambrell and Patrick, 1978). More reduced soil conditions and available sulfate, which is abundant in seawater (Riley and Skirrow, 1975), can cause the formation of sulfides (Gambrell and Patrick, 1978). High sulfide concentrations have been shown to reduce the growth of wetland plant species including *S. alterniflora* and *S. lancifolia* (Koch and Mendelssohn, 1989; Mendelssohn and McKee, 1988; Havill et al., 1985; King et al., 1982; Morris, 1980). High sulfide concentrations have also been shown to inhibit the plant uptake of $\text{NH}_4\text{-N}$, the most abundant form of N in wetlands (Koch et al., 1990; Bradley and Morris, 1990). These edaphic changes associated with increased plant submergence and saltwater intrusion are detrimental to wetland plant growth.

Some researchers have shown a relationship between soil mineral matter content and wetland plant health (DeLaune et al., 1979; DeLaune et al., 1990; Nyman et al., 1994). DeLaune et al. (1979) found a relationship between soil bulk density and the growth of *S. alterniflora*. They determined that soil bulk densities below 0.20 g cm^{-3} would not be capable of supporting the growth of *S. alterniflora*. This indicates that marshes which no longer receive mineral sediment input would not be able to support the growth of some coastal marsh plant species.

Collectively, there are a wide range of stresses that can effect the vegetation within deteriorating marshes of coastal Louisiana. Although the physical processes which drive wetland loss are well understood, the factors that cause vegetation dieback, an important component of the wetland loss process, are not known. The overall goal of this study is to determine the edaphic factors which are responsible for vegetation dieback within two coastal marshes in Louisiana, a salt marsh dominated by *S. alterniflora* and an oligohaline marsh dominated by *S. lancifolia*, through field monitoring and manipulative field experiments.

Chapter 2 “Factors associated with vegetation dieback in a Louisiana salt marsh: a three-year field study”, describes the results of a three year field study to determine the factors causing vegetation dieback in a deteriorating salt marsh in the Terrebonne basin of coastal Louisiana. This chapter has been submitted to the Journal of Coastal Research and is in review.

Chapter 3 “Causes for vegetation dieback in a Louisiana salt marsh: A bioassay approach”, describes the results of a field experiment on the affects of marsh elevation on four salt marsh plant species. This chapter has been published in Aquatic Botany and permission for its use has been obtained.

Chapter 4 “An investigation of the factors associated with vegetation dieback in a Louisiana oligohaline marsh”, describes the results of a two-year study to determine the causes of vegetation dieback in a deteriorating low-salinity marsh in the Terrebonne basin of coastal Louisiana.

Chapter 5 “Factors affecting vegetation dieback of an oligohaline marsh in coastal Louisiana: Field manipulation of salinity and submergence”, describes the results of a field experiment that tested the effects of changes in salinity and submergence and their combination on oligohaline marsh vegetation. This chapter has been published in the American Journal of Botany and permission for its use has been obtained.

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CHAPTER 2

FACTORS ASSOCIATED WITH VEGETATION DIEBACK IN A LOUISIANA SALT MARSH: A THREE-YEAR FIELD STUDY

Introduction

Louisiana's coastal marshes represent 40% of the tidal wetland area of the conterminous United States (Turner, 1990). Currently, Louisiana is experiencing a rapid rate of wetland loss estimated at $65.6 \text{ km}^2 \text{ y}^{-1}$ (Dunbar et al., 1992). Large areas of vegetation dieback, usually occurring in the interior of marshes and forming shallow water bodies with unconsolidated sediments, characterize wetland deterioration in coastal Louisiana. The causes for wetland loss are related to physical processes such as natural subsidence of the Louisiana deltaic plain (Ramsey and Penland, 1989; Penland and Ramsey, 1990), decreased sediment input to surrounding marshes (Kesel, 1987; Reed and Cahoon, 1992) and impoundment by intersecting spoil banks associated with pipeline and access canals (Swensen and Turner, 1987). However, the reasons for vegetation dieback, an important component of the wetland loss process, are not as well understood.

In the Louisiana coastal zone, relative sea level rise, a combination of subsidence and eustasy (Penland et al, 1988), is approximately 1.19 cm y^{-1} , which is ten times the world wide average (Penland and Ramsey, 1990). This high rate of relative sea level rise, coupled with a sediment aggradation deficit, causes the vertical accretion rate of the marsh surface to be less than the rate of water level rise, and results in increased plant submergence and elevated salinities (Morgan, 1977; Turner, 1990). The depletion of soil oxygen due to increased water depths and soil waterlogging stresses wetland plants through an oxygen deficiency in plant roots and promotes the formation of soil phytotoxins such as hydrogen sulfide (Mendelsohn et al., 1981; Burdick et al., 1989). Strongly reduced soils and elevated sulfide concentrations inhibit root respiration,

nutrient uptake and, in turn, plant growth (Morris, 1980; Havill et al., 1985; Koch and Mendelssohn, 1989; Bradley and Morris, 1990; Koch et al., 1990).

Inadequate mineral matter accumulation in Louisiana coastal marshes is another factor possibly involved in vegetation dieback. Less mineral matter accumulation not only causes decreased vertical accretion rates that prevent marshes from remaining intertidal (Hatton et al., 1982), but also influences soil fertility and phytotoxin accumulation. For example, growth of *Spartina alterniflora*, a dominant salt marsh grass, was related to the amount of soil mineral matter and its associated nutrient content (DeLaune et al., 1979). Additionally, iron associated with mineral sediments can precipitate sulfides as relatively insoluble metal sulfides (King et al., 1982; Feijtel et al., 1988).

Thus, increased plant submergence, high sulfide concentrations, increased salinities and inadequate mineral matter availability can reduce plant growth in coastal marshes. However, the spatial and temporal dynamics of these factors and their relationship to vegetation dieback and subsequent wetland loss are not well understood. The objective of our study was to evaluate the factors responsible for vegetation dieback in Louisiana salt marsh environments. We conducted a three-year study to compare the vegetative and edaphic characteristics of two deteriorating salt marsh sites and a healthy salt marsh site to assess the relative contribution of abiotic factors to vegetation dieback. In addition to natural factors affecting vegetation dieback, we also assessed the influence of pipeline canals, a ubiquitous human-generated feature in coastal Louisiana.

Methods

Study Site Location

The study area is located within the Lafourche delta complex along the eastern portion of the Terrebonne delta plain which is an area that is typified by rapid rates of subsidence (Figure 2.1). The average rate of relative sea level rise for this area is 1.04 cm y⁻¹ (Penland and Ramsey, 1990). Two deteriorating salt marsh sites adjacent to

Bayou Chitigue, and a healthy, well-vegetated salt marsh site adjacent to Bayou Terrebonne, approximately 7 km northeast of Cocodrie, LA, were chosen for this investigation of vegetation dieback within a Louisiana salt marsh environment. This area was selected because of the proximity of marshes experiencing vegetation dieback to a healthy, well vegetated marsh site. Also, backfilled, pipeline canals are present in one of the deteriorating marsh sites, making it possible to evaluate the role of this anthropogenic disturbance in marsh deterioration.

Sampling Design

For field monitoring, forty-two permanent plots, 4 m X 5 m, were located within three salt marsh sites dominated by *S. alterniflora* (Figure 2.1): a healthy, well vegetated site without pipeline canals (hereafter referred to as HNP, n=14) that served as a reference site, a deteriorating site without pipeline canals (hereafter referred to as DNP, n=14), and a deteriorating site bisected by two backfilled pipeline canals (hereafter referred to as DP, n=14). The three marsh sites are bordered to the north by an area of extreme internal marsh deterioration (the Madisonville “hotspot”), nearly devoid of vegetation (Leibowitz and Hill, 1987) and to the south by Lake Barre and Terrebonne Bay. All permanent plots were located at random distances south of the Madisonville hotspot and at random distances perpendicular to the bayou.

Sampling Procedure

Within each 4 m x 5 m permanent plot, specific biotic and abiotic variables were sampled at randomly located 0.25 m² plots. The selected plots were marked after each sampling in order to avoid sampling the same plot again.

Aboveground biomass was harvested at ground level from a 0.25 m² quadrat, transported to the laboratory and separated by plant species into live and dead components, dried to a constant weight at 65^o C, and dry weight determined to the

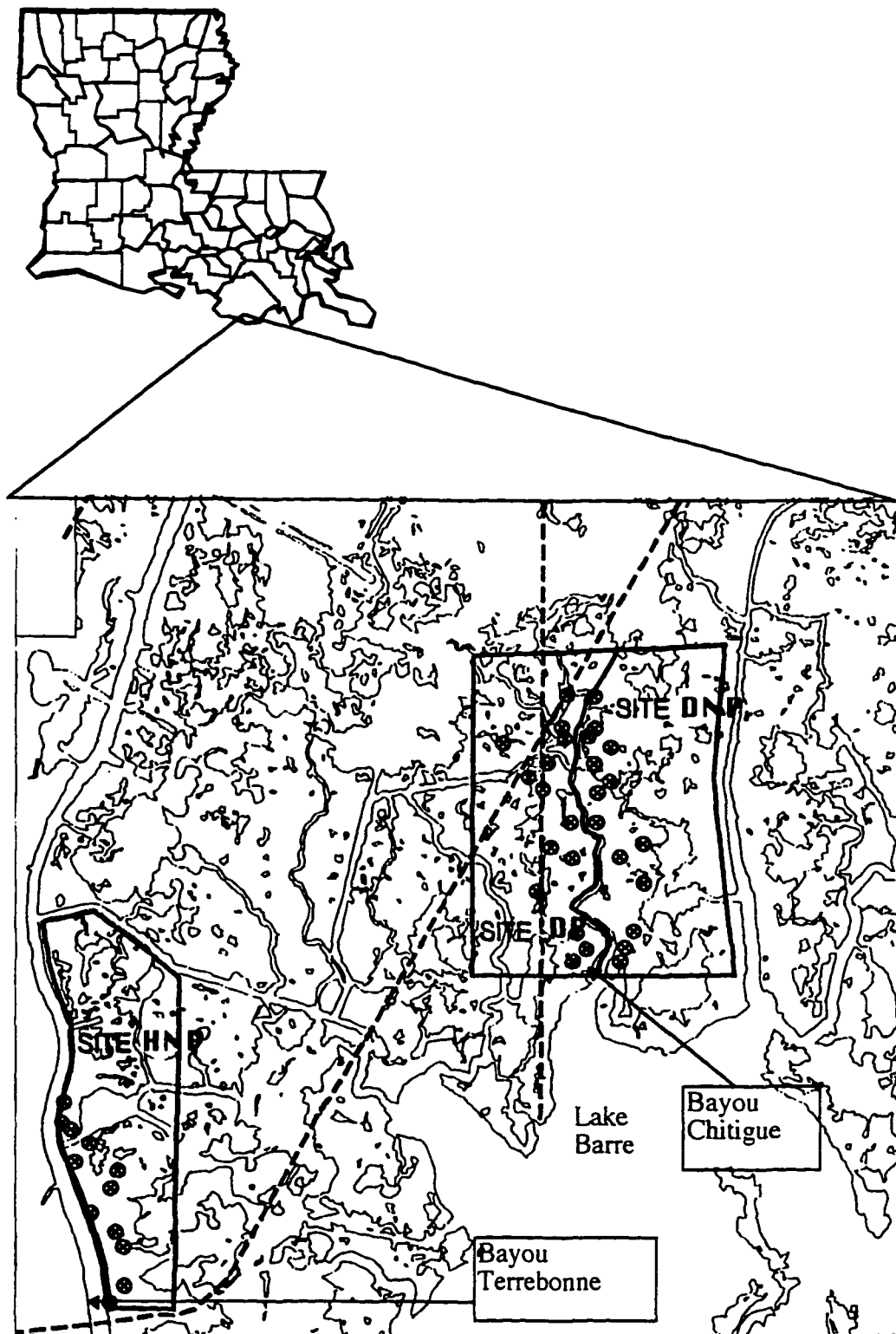


Figure 2.1 Location of reference marsh site (HNP, $n=14$), deteriorating marsh site bisected by backfilled pipeline canals, indicated with dashed lines, (DP, $n=14$), and deteriorating marsh site without pipeline canals (DNP, $n=14$).

nearest 0.1 g. Within a 1 m² permanent quadrat, located in each 4 m x 5 m permanent plot, percent plant cover and species composition was determined.

Soil redox potentials were measured at 2 and 15 cm depths in an undisturbed area adjacent to the 0.25 m² clip plot with bright platinum electrodes. Redox potentials (Eh), which were determined in triplicate per plot, were calculated by adding the potential of the calomel reference electrode (+244 mV) to each mV value.

Soil bulk density was determined by collecting a 190.5 cm³ soil core from each clip plot. Compaction of the core was minimized by cutting a groove in the soil prior to the insertion of the coring device. The soil core was dried at 65^o C to a constant weight and weighed to the nearest 0.1 g.

A second soil core, 15 cm long and 5 cm in diameter, was removed from each clip plot with an aluminum core tube for the extraction of interstitial water. Immediately upon removal from the soil, the core was placed into a 500 ml centrifuge tube, sealed, purged with N₂ gas for two minutes in order to maintain an anaerobic environment, and transported on ice to the laboratory. In the laboratory the core was centrifuged at 10,000 g at 4^o C to separate the interstitial water from the sediments. A 5 ml aliquot of the interstitial water was immediately removed from the centrifuge tube and added to antioxidant buffer, and dissolved sulfide (S⁻²) concentration was determined (Sulfide electrode, Lazar Research Laboratories, Los Angeles, CA). A second aliquot of unfiltered water was analyzed for pH (Altex Model 3560 digital pH meter), and salinity (Fisher Model 152 salinity/conductivity meter). A third aliquot was filtered through a 0.45 µm Uniflo syringe tip filter (Schleicher and Schull) and analyzed for P with a Fisher inductively coupled argon plasma emission spectrometer (Atom Comp Series 800) and for NH₄-N with a Technicon Auto Analyzer (Technicon Industrial Method at the Institute of Ecology, Soil Biology Laboratory, University of Georgia).

In order to sample all 42 sites, two days of consecutive sampling were required. Samples were obtained three times in the first year of the study: March 10/11, 1991, June 26/27, 1991 and October 5/6, 1991, and twice in the second and third years of the study: June 23/24, 1992 and October 23/24, 1992; June 29/30, 1993 and November 8/9, 1993.

Statistical Analysis

All statistical analyses were computed using SAS (GLM Procedure, CORR Procedure, Statistical Analysis Systems, 1985). The study was analyzed as a split plot design with time, treatment and distances from bayou as main effects. Differences between healthy and deteriorating marsh sites as well as differences between the deteriorating sites with and without pipeline canals were tested using contrast statements. Tests for normality and homogeneity of variance were conducted on all variables. Transformation was required for interstitial sulfide (log), Mn (log), Fe (reciprocal) and P (reciprocal) to meet the assumptions of analysis of variance.

Results

Live Biomass and Plant Cover

Initially we observed that HNP had robust plant growth dominated by a relatively continuous cover of *S. alterniflora* while DP and DNP had patchy vegetation, also dominated by *S. alterniflora* but separated by small ponds, with some areas completely devoid of vegetation. Aboveground biomass and plant cover were measured to quantify these biotic differences between the marsh sites.

The species composition of all three marshes was similar. The dominant plant species was *S. alterniflora*, making up 90% of the mean live biomass of DP, 89% of the mean live biomass of DNP and 82% of the mean live biomass of HNP. Other plant species found in all three marshes were *Juncus roemerianus*, *Distichlis spicata* and *S. patens*.

Live aboveground biomass (Figure 2.2) and percent cover (Figure 2.3) were significantly greater in the reference marsh compared to the two deteriorating marshes ($P < 0.01$), which did not differ significantly ($P = 0.67$). Live aboveground biomass was lowest on May 1991 (main effect of time significant [$P < 0.01$]) and increased with each sampling date until a maximum mean biomass of 765.7 g m^{-2} was reached in October 1992 (Figure 2.2). This was the first sampling date after the passage of Hurricane Andrew which crossed South Louisiana on August 26, 1992 making landfall ca. 65 km west of the study site. The pattern of change through time for biomass showed less variation in the reference site compared to the deteriorating sites, but the interaction was significant only at the 6% probability level. No significant difference in plant cover was found between DNP and DP, and percent cover did not significantly change over time at any site ($P = 0.28$).

Redox Potential, Interstitial Sulfide Concentration and pH

The soil at the deteriorating sites was more reducing than the soil at the reference site ($P < 0.05$), but this pattern was more consistent at 15 cm depth (Figure 2.4). There was substantial variation through time with periods in which the soils at all three sites were either more or less reduced. Soil Eh was not significantly different between the deteriorating marshes.

Interstitial sulfide concentration (Figure 2.5a) was significantly higher in the deteriorating marshes than in the reference marsh ($P < 0.01$) but did not differ between the deteriorating marsh with pipeline canals and those without pipeline canals. The highest sulfide concentration (Figure 2.5b) occurred in June 1991 and the lowest in May 1991 (main effect of sampling date significant [$P < 0.01$]). Mean interstitial sulfide concentration was higher than 1 mM on all sampling dates except May 1991 and October 1991. There was no significant interaction between marsh site and sampling date ($P = 0.21$).

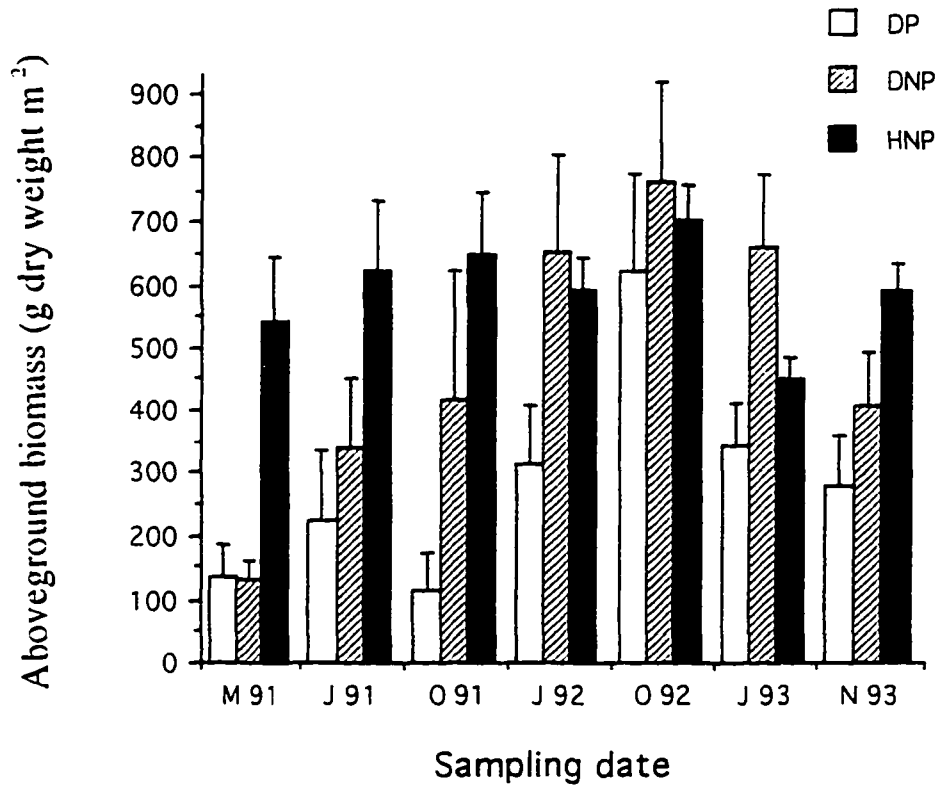


Figure 2.2. Aboveground live biomass for each marsh site by sampling date. Bars represent mean live biomass ± 1 standard error.

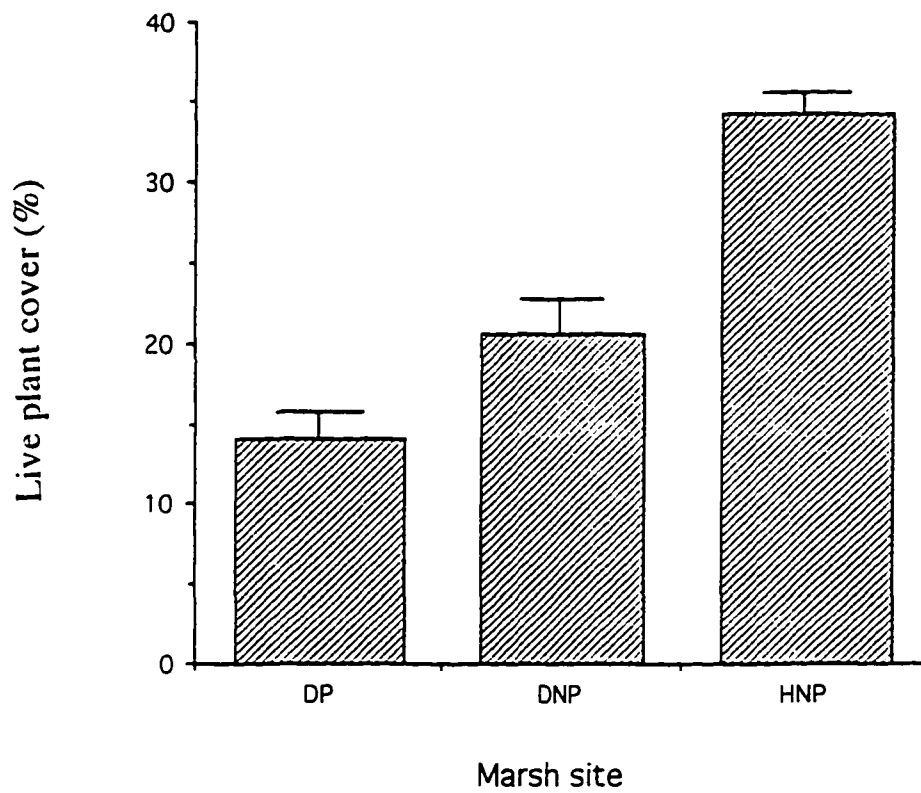


Figure 2.3. Live vegetation cover by marsh site. Bars represent the mean of all sampling dates \pm 1 standard error.

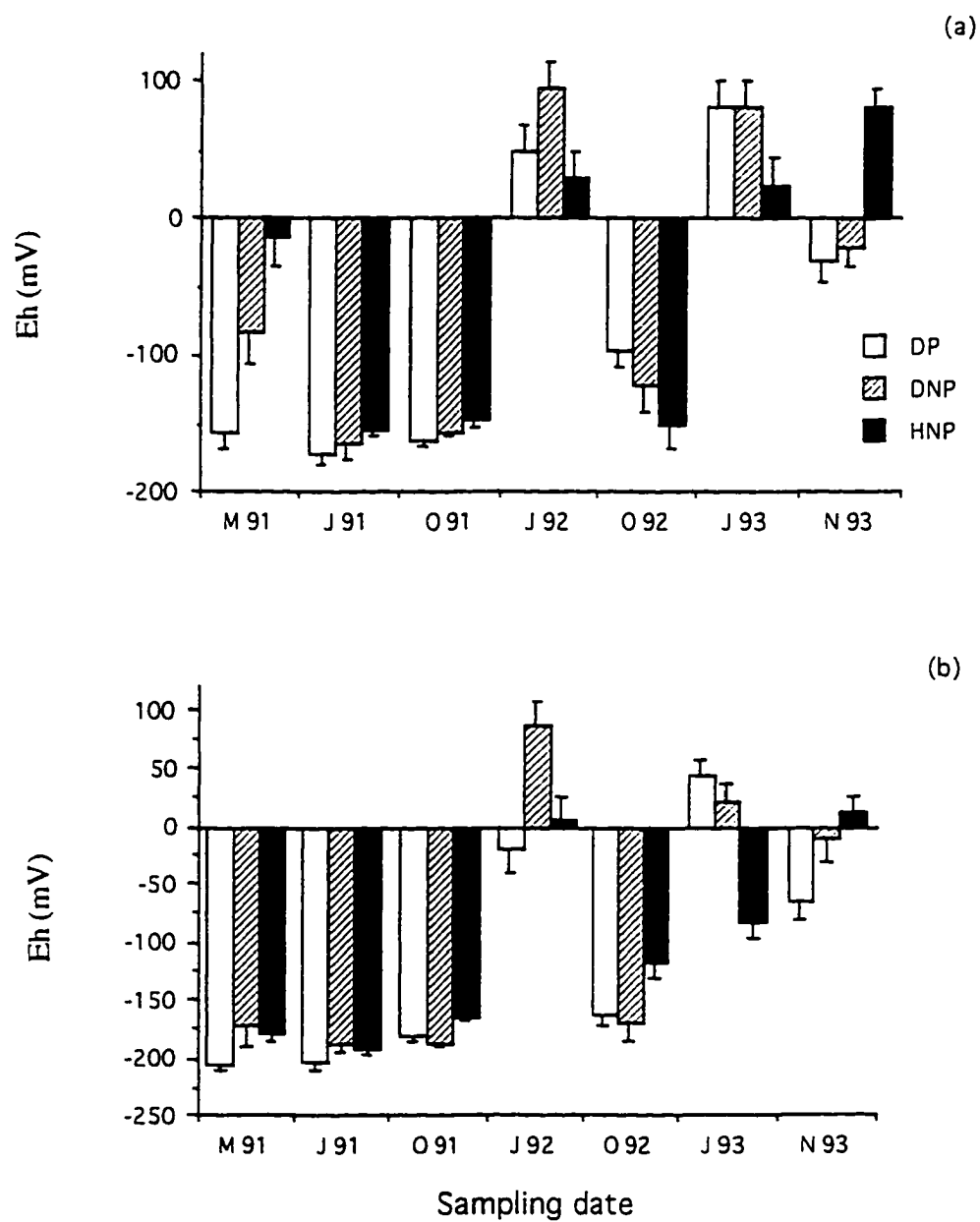


Figure 2.4. Redox potential at 2 cm depth (a) and at 15 cm depth (b) by marsh site for each sampling date. Bars represent mean \pm 1 standard error.

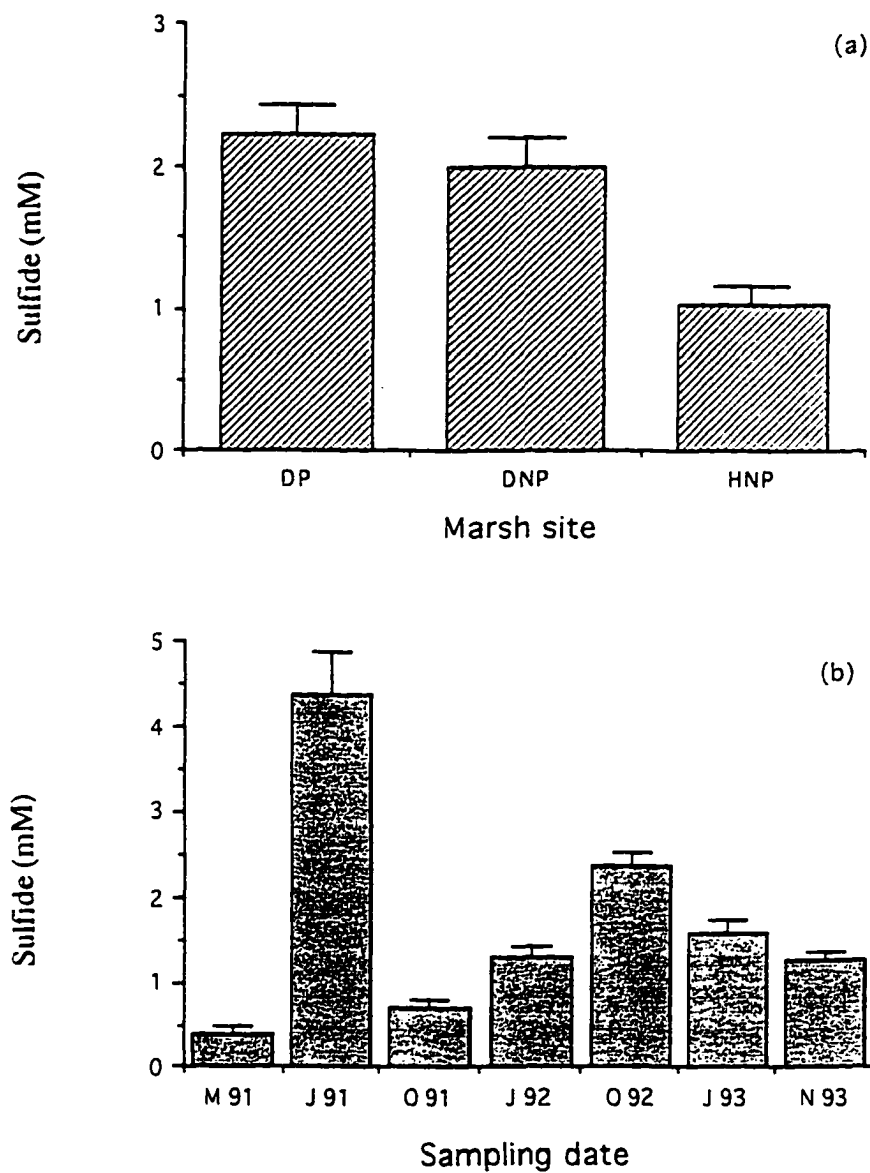


Figure 2.5. Interstitial sulfide concentration by marsh site averaged across all sampling dates (a) and by sampling date averaged across all marsh sites (b). Bars represent mean ± 1 standard error.

The pH of all three sites was near neutral for all sampling dates although the pattern of temporal variation differed among marsh sites ($P < 0.01$). Mean interstitial pH ranged from 6.5 ± 0.1 to 8.0 ± 0.1 during the three year study.

Macronutrient Concentration

The deteriorating marshes had significantly higher $\text{NH}_4\text{-N}$ concentrations than the reference marsh ($P < 0.01$) but $\text{NH}_4\text{-N}$ concentrations did not differ significantly between DNP and DP (Figure 2.6a). The highest $\text{NH}_4\text{-N}$ concentrations occurred in the fall each year (Figure 2.6b): October 1991, October 1992 and November 1993 (main effect of sampling date significant [$P < 0.01$]). No significant interaction between marsh site and time was found ($P = 0.88$).

Interstitial P concentration (Figure 2.7) was significantly higher in the reference marsh than at the deteriorating marshes ($P < 0.01$) but no significant difference in interstitial P concentration was found between the two deteriorating marsh sites. There was no significant temporal effect ($P = 0.97$) nor a significant sampling date x marsh site interaction ($P = 0.99$) for interstitial P concentration.

Salinity

Interstitial salinity (Figure 2.8) did not differ between marsh sites ($P = 0.22$), but salinities did differ significantly between sampling dates ($P < 0.01$). Overall the salinities were low for a salt marsh environment and ranged between 5.1 ppt in March, 1991 to 13.3 ppt in June, 1992. Temporal variation was similar for all marshes (marsh site x sampling date interaction not significant [$P = 0.13$]).

Soil Bulk Density

Soil bulk density (Figure 2.9) varied significantly with time ($P < 0.01$) and had a significant marsh site x sampling date interaction ($P < 0.05$); sampling dates previous to August, 26, 1992, the date of the passage of Hurricane Andrew, had mean soil bulk

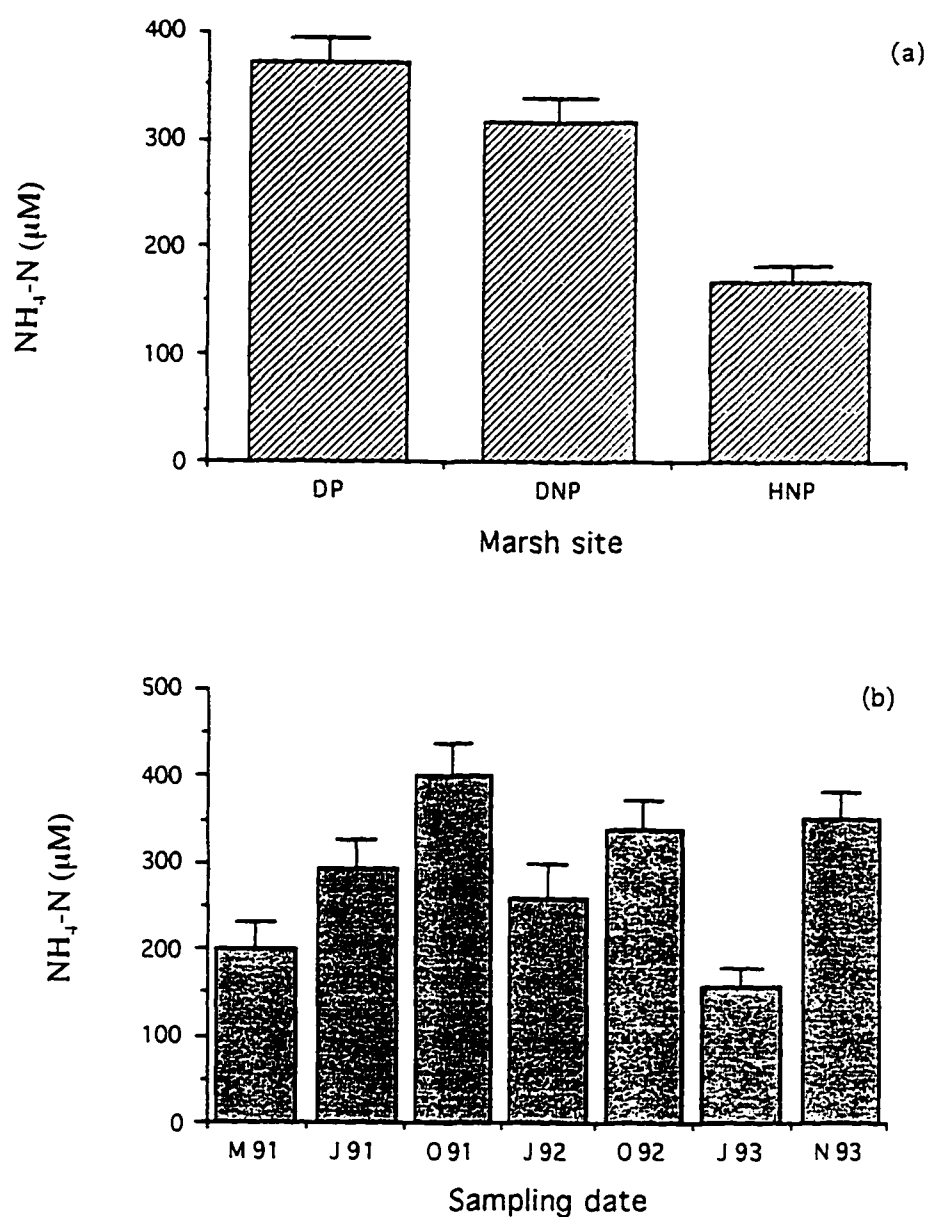


Figure 2.6. Interstitial $\text{NH}_4\text{-N}$ concentration by marsh site averaged across all sampling dates (a) and by sampling date averaged across all marsh sites (b). Bars represent mean ± 1 standard error.

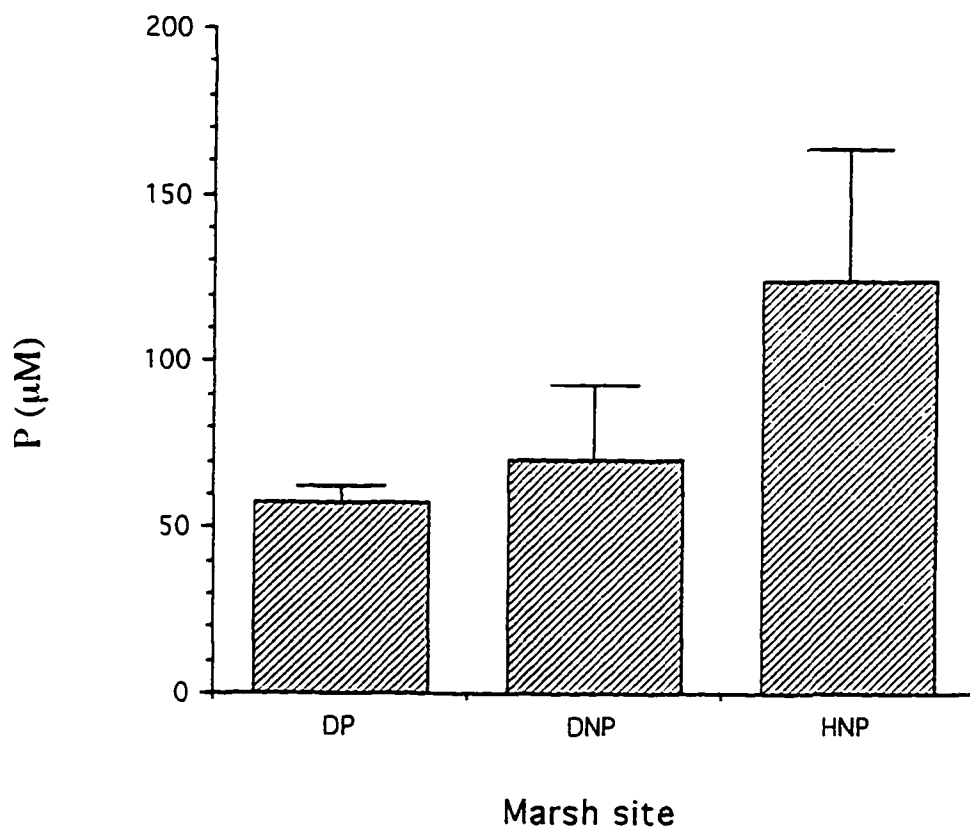


Figure 2.7. Interstitial P concentration by marsh site averaged across all sampling dates. Bars represent the mean \pm 1 standard error.

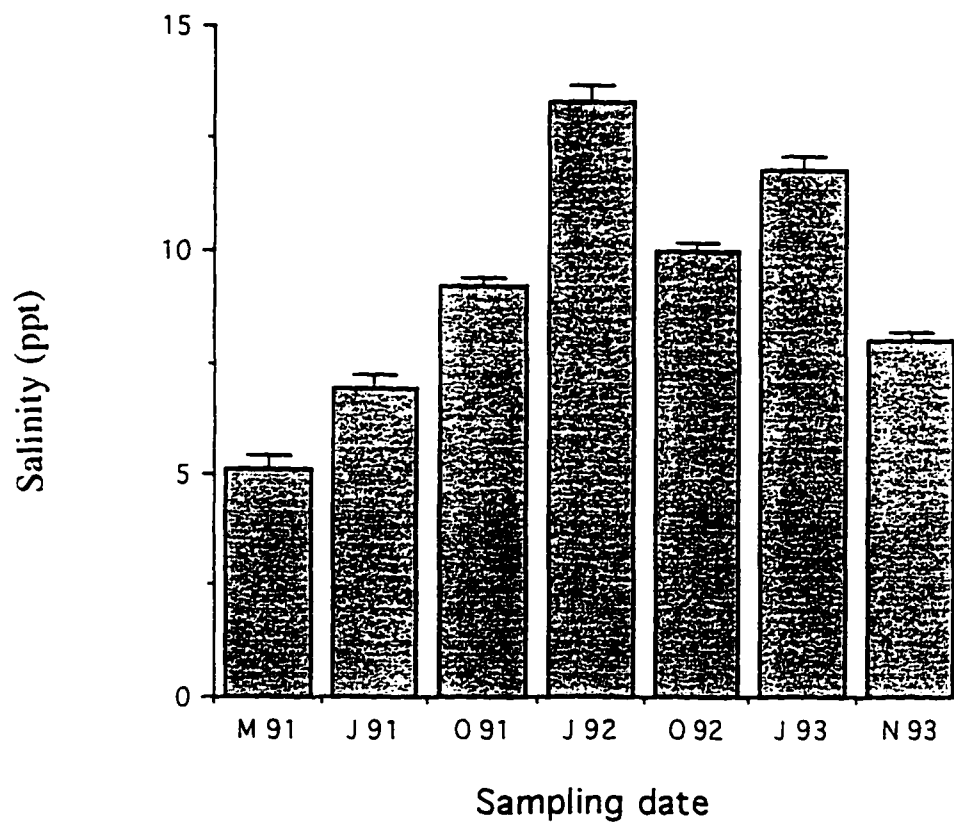


Figure 2.8. Interstitial salinity concentration by sampling date averaged across all marsh sites. Bars represent mean \pm 1 standard error.

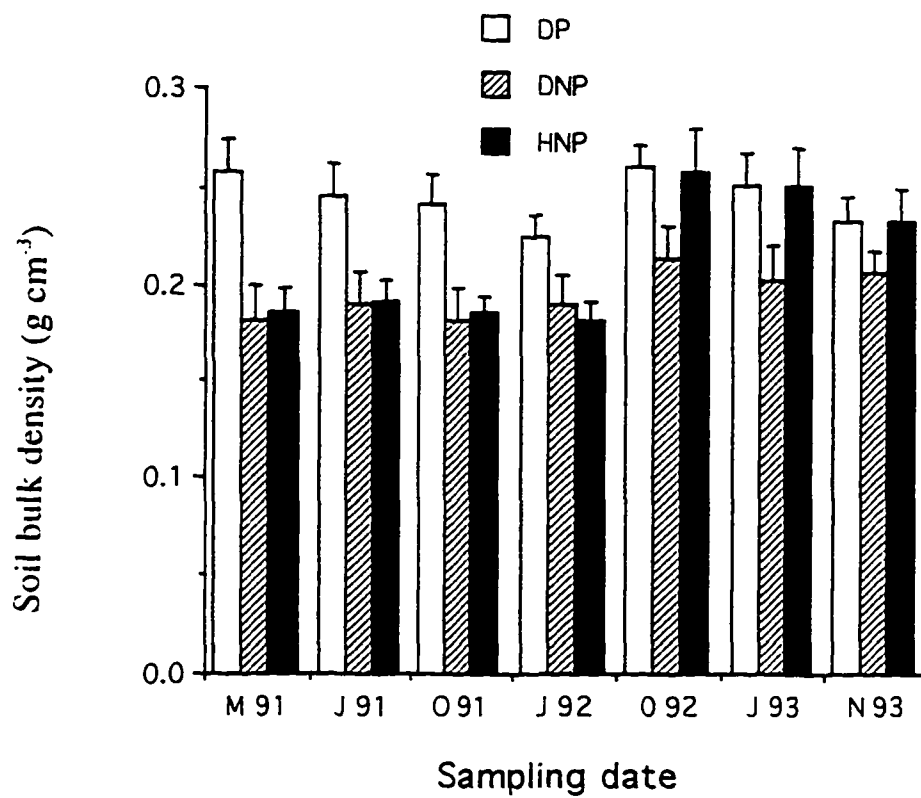


Figure 2.9. Soil bulk density by marsh site for each sampling date. Bars represent mean \pm 1 standard error.

densities, averaged across all marsh sites, ranging between 0.20 g cm^{-3} and 0.21 g cm^{-3} . On sampling dates after the passage of Hurricane Andrew (October 1992, June 1993 and November 1993), mean soil bulk density ranged between 0.22 g cm^{-3} and 0.24 g cm^{-3} . Soil bulk density was higher in the deteriorating sites than in the reference site prior to the passage of Hurricane Andrew ($P < 0.05$).

Correlation Analysis

A correlation analysis was conducted to determine if live plant biomass and live plant cover were associated with edaphic characteristics. Initially, correlation analysis was determined from the raw data but due to the large variability across sampling dates, variables were found to be only weakly correlated. In order to decrease the variation

Table 2.1. Correlation matrix between biotic and abiotic variables. Correlation coefficient and significance level are given for each correlation. Asterisk (*) indicates a correlation significant at the 0.05 level.

	Live biomass	Live plant cover
Eh, 2cm depth	0.1009 0.53	-0.0036 0.98
Eh, 15 cm depth	0.0455 0.77	-0.0701 0.66
Interstitial sulfide	-0.4325* <0.01	-0.4435* < 0.01
Interstitial pH	-0.3180* 0.04	-0.4057* < 0.01
Interstitial $\text{NH}_4\text{-N}$	-0.4438* < 0.01	-0.5426* < 0.01
Interstitial P	0.2234 0.16	0.0946 0.55
Interstitial Salinity	0.1661 0.29	0.0291 0.85
Interstitial Fe	-0.2570 0.10	-0.0741 0.64
Interstitial Mn	0.1260 0.43	0.0864 0.59

associated with time, correlation analysis was conducted on treatment means. Interstitial sulfide, $\text{NH}_4\text{-N}$ and pH were negatively correlated with live biomass and live plant cover (Table 2.1).

Discussion

Healthy vs. Deteriorating Marsh Sites

The analysis of biotic variables indicated that the deteriorating marshes had significantly poorer plant growth than the reference marsh. This supports our initial observation that vegetation dieback was occurring within the deteriorating marsh sites during the study period. Furthermore, our results indicate that plant stressors such as more reduced soil conditions and high sulfide concentrations caused by prolonged soil waterlogging can be related to vegetation dieback within the deteriorating marsh sites.

Decreased tidal flushing and prolonged inundation of soils and plants lead to dramatic decreases in rates of oxygen diffusion (Greenwood, 1961). As oxygen and other electron acceptors are consumed, the soil becomes increasingly reduced and phytotoxins such as sulfide, accumulate (Gambrell, DeLaune and Patrick, 1991). These conditions have been shown to limit *S. alterniflora* productivity (Mendelssohn and Seneca, 1980; Howes et al., 1981; Mendelssohn and McKee, 1988). The deteriorating marshes had higher sulfide concentrations and less vegetative growth than the reference marsh. Sulfide is a known phytotoxin and can reduce the growth of some salt marsh plant species (Linthurst, 1979; Morris, 1980; Linthurst and Seneca, 1980; King et al., 1982; Havill et al., 1985; Pearson and Havill, 1988; Koch and Mendelssohn, 1989).

Sulfide, in particular, inhibits the uptake of $\text{NH}_4\text{-N}$ by *S. alterniflora* (Koch et al., 1990; Bradley and Morris, 1990). Ammonium is the most bioavailable form of N in flooded soils and the most important growth limiting nutrient in salt marshes (Valiela and Teal, 1974). The higher $\text{NH}_4\text{-N}$ concentrations in the deteriorating marshes may reflect an inhibition of $\text{NH}_4\text{-N}$ uptake by the vegetation as suggested previously

(Mendelssohn and McKee 1988). The correlation analysis supports this conclusion since interstitial sulfide and $\text{NH}_4\text{-N}$ concentration were inversely related to live biomass and live plant cover. The direct effect of sulfide on root respiration, combined with inhibition of plant $\text{NH}_4\text{-N}$ uptake was sufficient to inhibit growth of plants (Koch and Mendelssohn, 1989; Koch et al. 1990) and thus contribute to the dieback in the deteriorating marsh sites. However, the reciprocal effect of the vegetation (or lack of it) on these soil factors cannot be discounted. Wetland plants have been shown to influence soil redox potential and sulfide through the release of oxygen from the roots to the soil (Jaynes and Carpenter 1986; McKee, 1993).

Although P is not a primary limiting factor to the growth of salt marsh plants, it has been shown to limit the growth of *S. alterniflora* in areas where inorganic phosphorous occurs at low concentrations (Broome et al., 1975). We found lower interstitial P concentrations at the deteriorating marshes than at the reference marsh (Figure 7). The processes effecting phosphorus cycling in estuarine sediments is not well understood empirically (Day et al., 1989, p. 127) and the data in this study is insufficient to explain the differences in P concentration between treatments. Although, phosphorus may be limiting plant growth at the deteriorating marshes, it is generally believed that P concentration is considered secondary to N as a limiting factor to *S. alterniflora* growth (Mendelssohn et al., 1982). Further studies may be necessary to understand P cycling in the dynamic salt marshes of Louisiana as well as its overall effect on plant growth.

Saltwater intrusion is often cited as an important cause of vegetation dieback in coastal Louisiana, but little empirical data is available to support this contention. At Bayou Chitigue, interstitial salinities did not differ with marsh type, and salinities were relatively low for a salt marsh environment, not exceeding 13.9 ppt. Therefore, vegetation dieback in this salt marsh was not caused by high salinities during the study period.

Salt marsh loss in coastal Louisiana has been attributed to a deficiency in mineral sediment deposition, and several studies have shown a relationship between salt marsh plant growth and soil mineral matter content (DeLaune et al., 1979; DeLaune et al., 1990; Nyman et al., 1994). Although some researchers have found that low soil bulk density can be correlated to decreased *S. alterniflora* productivity and that soil bulk densities below 0.20 g cm^{-3} are likely incapable of supporting the growth of *S. alterniflora* (DeLaune et al., 1979), we found that soil bulk density was higher in the deteriorating marshes prior to Hurricane Andrew and did not differ between the well vegetated, healthy marsh and the poorly vegetated, deteriorating marshes after the passage of Hurricane Andrew. Furthermore, the site with the poorest plant growth, DP, had the highest mean soil bulk density ($> 0.20 \text{ g cm}^{-3}$ on all sampling dates) of all three sites, and live biomass and live plant cover were not correlated with soil bulk density. These findings indicate that low soil bulk density was not responsible for the dieback of vegetation in this salt marsh and highlights the need for more information to determine the relative importance of this factor.

The role of storm events in the deposition of mineral matter in Louisiana coastal marshes is well documented (Reed, 1989). We attribute the significant increase in soil bulk densities in October 1992 and subsequent sampling dates to the deposition of reworked mineral matter on the marsh surface during passage of Hurricane Andrew. This sediment deposition caused 3 cm of vertical marsh accretion at this location (Cahoon et al., 1995). Also, the sampling date with the greatest live biomass was October 1992 which was the first sampling date after the passage of Hurricane Andrew. Although the deposition of fresh mineral matter and related nutrients may be, in part, responsible for the increased biomass, it is more likely that the 3 cm of vertical accretion (Cahoon et al., 1995) was responsible for the improvement in plant growth. These results illustrate the importance of large storm events, such as Hurricane Andrew, on the

deposition of reworked mineral matter in both healthy and deteriorating coastal marshes in Louisiana.

Comparisons of vertical marsh accretion and marsh elevations revealed however, that sediment deposition, even on this scale, was insufficient to counterbalance the subsidence rates in these marshes (Cahoon et al., 1995). Measurements conducted at Bayou Chitigue between 1992 and 1994 showed that during the two-year period, the marsh at Bayou Chitigue had 5.19 cm of vertical accretion, 3 cm of which occurred after the passage of Hurricane Andrew. Although a high rate of vertical marsh accretion occurred at Bayou Chitigue, the increase in surface marsh elevation was only 0.29 cm during this same time period. This finding indicates that there was 4.90 cm of shallow subsidence (vertical accretion - elevation change, Cahoon et al., 1995) of the marsh at Bayou Chitigue between 1992 and 1994. Thus, mineral matter additions as well as increased vertical marsh accretion may not be enough to counteract the effects of increased waterlogging on vegetation caused by the high rate of subsidence in deteriorating marshes of Louisiana.

Nyman et al. (1993) have proposed a hypothetical model of wetland loss based on a positive feedback loop between vegetation stresses and inadequate vertical accretion. An important component of soil formation in Louisiana's coastal marshes is organic matter accumulation (Nyman et al., 1990). Therefore, a decrease in plant productivity in coastal marshes causes a decrease in the organic matter contribution to marsh soils and hence vertical accretion. Failure to maintain vertical accretion results in more frequent and prolonged inundation of plants and increased stress leading to reduced growth (Havill et al., 1985; Pearson and Havill, 1988; Koch and Mendelssohn, 1989). As plant productivity decreases further due to these stresses, less organic matter is contributed to the soil, vertical marsh accretion is less, and stresses associated with excessive waterlogging cause further decreases in plant productivity. The results of our

study provide evidence in support of this positive feedback loop (Nyman et al., 1993) and further indicate that the causes of decreased plant productivity is plant submergence rather than salt stress due to saltwater intrusion.

Pipeline Canal Effects

The effect pipeline canals have on coastal marshes of Louisiana can be dramatic. For example, approximately 8 % of the coastal marshes of Louisiana have been converted to open water canals and their associated spoil banks (Neill and Turner, 1987). Backfilling is utilized as a method for mitigating the impacts of canals on coastal marshes in Louisiana. After backfilling, canals typically remain unvegetated with no more than 10 % vegetation cover five years after backfilling (Neill and Turner, 1987). Even though the majority of marsh vegetation in coastal Louisiana grows between 15 cm below and 15 cm above mean water level (Sasser, 1977), most backfilled pipeline canals remain unvegetated even though their surface elevation is less than 50 cm below the marsh surface (Neill and Turner, 1987). Though we could not determine what direct influence the backfilled pipeline canals have had on past rates of vegetation dieback and subsequent wetland deterioration at the experimental marshes, the backfilled pipeline canals are not contributing to vegetation dieback at site DP. This conclusion is supported by the fact that vegetative and edaphic characteristics are similar at the two deteriorating marshes during the study period.

Conclusion

Visual differences in vegetative cover and vigor between the reference marsh and deteriorating marshes were supported by the data. The factors associated with vegetation dieback within these deteriorating salt marshes are related to increased plant submergence and not to saltwater intrusion. Additionally, growth of *S. alterniflora* was not related to a specific soil bulk density level as has been previously suggested (DeLaune et al., 1979).

Current wetland restoration efforts in coastal Louisiana include deteriorating salt marshes of the Mississippi River deltaic plain. This study indicates that increasing marsh surface elevation, possibly by sediment addition, is necessary to counteract the effects of plant submergence on these salt marsh systems in Louisiana. However, it is necessary that any restoration efforts that include sediment addition raise marsh elevation enough to fully counteract the effects of local relative sea level rise.

Global sea level may rise as much as 200 cm during the next century (Titus et al., 1991). This scenario will cause increased inundation of salt marshes worldwide. Louisiana provides a model of the long-term effects of changes in sea level on coastal wetlands. Increased plant submergence due to changes in global sea level will likely cause vegetation dieback in many salt marsh systems throughout the world.

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CHAPTER 3

CAUSES FOR VEGETATION DIEBACK IN A LOUISIANA SALT MARSH: A BIOASSAY APPROACH*

Introduction

Wetlands in coastal Louisiana are rapidly converting to open water. Wetland loss rates for the Mississippi River deltaic plain have been estimated at $65.6 \text{ km}^{-2} \text{ year}^{-1}$ (Dunbar et al., 1992) and are, in part, caused by subsidence rates that exceed the rates of vertical marsh accretion (Ramsey and Penland, 1989). Though the high rates of wetland loss in coastal Louisiana are related to natural processes of the Mississippi River delta cycle (Scruton, 1960), wetland loss rates have been accelerated by anthropogenic disturbances. An important component of wetland loss in coastal Louisiana is the dieback of marsh vegetation. As opposed to shoreline erosion, marsh interior fragmentation and degradation as a result of plant dieback are the primary causes of marsh loss (Turner and Cahoon, 1987). Though the causes for wetland loss in coastal Louisiana are numerous and well studied, the factors affecting vegetation dieback within tidal marshes are not as well understood (Mendelssohn and McKee, 1988).

The high rates of relative sea level rise (land subsidence plus water level increase) in coastal Louisiana result in increased depths and duration of flooding and increased plant submergence leading to plant death (Mendelssohn and McKee, 1988). Salt marsh plants are exposed to a variety of stressors as submergence increases. Increased soil waterlogging leads to more reduced soil conditions (Gambrell and Patrick, 1978) that can cause the formation of toxic compounds such as hydrogen sulfide (King et al., 1982; DeLaune et al., 1983; Mendelssohn and McKee, 1988) and decrease the plant uptake of essential nutrients (Morris, 1980; Koch et al., 1990).

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Marine influence increases as relative sea level rises, resulting in higher salinities as well as increased plant submergence. High salinity is detrimental to many marsh plant species (McKee and Mendelssohn, 1989) and is often cited, but not well documented, as the most common cause of vegetation dieback in Louisiana. Saltwater intrusion is responsible for decreased productivity of some brackish marsh plants. *Spartina patens* (Aiton) Muhl., a dominant brackish species in coastal Louisiana, exhibits decreased stomatal conductance and reduced net photosynthesis at elevated soil salinities (Pezeshki et al., 1987). However, Burdick et al. (1989) have found that salinity stress alone does not explain the decreased productivity of *S. patens* in natural marsh environments. Nonetheless, saltwater intrusion, as well as marsh submergence, may control wetland loss in coastal Louisiana.

Understanding the causes of vegetation dieback is a prerequisite for successful management of these deteriorating systems. Based on this premise, we designed a field bioassay experiment to assess the potential causes for plant dieback in a salt marsh in coastal Louisiana. The objective of this study was to determine if salinity or submergence is the primary factor controlling the recent dieback of the vegetation in this marsh.

Methods

Study Area

The study area was located in a tidal salt marsh within the Terrebonne basin approximately 7 km east of Cocodrie in southern Terrebonne Parish, Louisiana (29° 19' N, 90° 35' W). A manipulative field experiment was conducted in a deteriorating salt marsh, dominated by *Spartina alterniflora* Loisel, located adjacent to Bayou Chitigue. The Bayou Chitigue area is characterized by high rates of vegetation dieback, forming large expanses of marsh interspersed with shallow ponds. The location selected for the experiment, although devoid of vegetation, still had plant stubble present at the soil

surface indicating the dieback of the vegetation was recent and no erosion of the marsh surface had occurred

Average interstitial salinities measured at 42 sites twice annually between 1991 and 1993 in the Bayou Chitigue area ranged from 5 ppt to 13 ppt. Tidal amplitude for the Bayou Chitigue area varies between 30 and 60 cm during spring tides and between 10 and 20 cm during neap tides (Wang and Sikora, 1991).

Experimental Design

In order to determine the relative importance of salinity and submergence in controlling the dieback of the vegetation, a field bioassay experiment was conducted in which the elevation of different marsh plant species that vary in their salinity and flooding tolerances was manipulated. This was accomplished by transplanting from the same marsh four native salt marsh plant species, *S. alterniflora*, *S. patens*, *Distichlis spicata* (L.) Greene, and *Juncus roemerianus* Scheele, into the unvegetated Bayou Chitigue marsh at two elevations: (1) ambient with the marsh surface and (2) 20 cm above the marsh surface.

The elevated treatments were created by removing unvegetated marsh sods (30 cm high and 23 cm diameter) and placing them into 14.5 l plastic pots with eighteen 2 cm holes in each pot to allow for drainage. The pots were then placed on the marsh surface so that 20 cm of the sod extended above the marsh surface and 10 cm remained below the marsh surface. An increase in elevation of 20 cm was selected so that the surface of the elevated treatments was equivalent in elevation to nearby healthy salt marshes. The ambient elevation treatment was established by removing unvegetated marsh sods, originating from the deteriorating marsh, and replacing them into their vacated holes at their original elevation. This created a similar physical disturbance as for the +20 cm treatment. A ring containing six 2 cm holes to allow for drainage and made from the same material as the pots was inserted into the soil 10 cm to simulate the same

impairment to groundwater drainage that was likely experienced by the elevated sods. Ramets of equivalent size of the four plant species were collected from a nearby marsh and transported to the experimental location. Three ramets of each plant species were then transplanted into randomly assigned sods.

The plant species used in this experiment have differential flooding and salinity tolerances. The order of flooding tolerance is *S. alterniflora* > *J. roemerianus* \geq *S. patens* > *D. spicata* (Eleuterius and Eleuterius, 1979; Gleason and Zieman, 1981; Gardner et al., 1992). The order of salinity tolerance is *D. spicata* > *S. alterniflora* \geq *J. roemerianus* > *S. patens* (Gosselink 1970; Smart and Barko, 1978; Parrondo et al., 1978). The experiment was initiated in August, 1991 and terminated in August, 1992.

Variables Sampled

Prior to plant harvest, redox potential was determined in each sod at two depths: 2 cm and 15 cm. Redox potential was measured using two brightened platinum electrodes and a calomel reference electrode at each depth. The electrodes were allowed to stabilize for 30 minutes prior to measurements. The potential of the calomel reference electrode against a standard hydrogen electrode (+245 mV) was added to the measured potential in order to determine Eh.

Aboveground vegetation was harvested at the soil surface from each sod. The plants were brought to the laboratory, sorted into their live and dead components and dried at 65° C for 48 hours. Plant samples were weighed to the nearest 0.1 g for biomass determination.

After the vegetation was harvested, soil cores, 5 cm in diameter and 15 cm long, were extracted from the root zone of each sod using an aluminum coring device. The soil cores were immediately placed in 500 ml, polypropylene centrifuge tubes. The centrifuge tubes were sealed and purged with nitrogen gas for two minutes, placed on

ice and brought to the laboratory for analysis. Soil samples were centrifuged at 10,000 g at 4°C for 15 minutes in a refrigerated centrifuge. The interstitial water was immediately removed from the centrifuge tube and the first aliquot was placed in an antioxidant buffer and analyzed for hydrogen sulfide concentration (Sulfide Electrode, Lazar Research Laboratories, Los Angeles, CA). A second aliquot was analyzed for pH (Altex Model 3560 digital pH meter), salinity and conductivity (Fisher Model 152 salinity/conductivity meter). The third aliquot was filtered using a 0.45 µm Uniflo syringe tip filter (Schleicher and Schull, Keene, NH), preserved with nitric acid (US EPA, 1974) and analyzed for P, Fe, Mn, Zn and Cu concentrations with a Fisher inductively coupled argon plasma emission spectrometer (Atom Comp Series 800). The fourth aliquot was filtered through a 0.45 µm filter (same as above), preserved with sulfuric acid (US EPA, 1974) and analyzed for NH₄-N concentration using the Technicon Industrial Method at the Institute of Ecology, Soil Biology Laboratory, the University of Georgia.

Statistical Analysis

Analysis of variance was used to analyze the data (GLM Procedure, SAS Institute, Cary, N.C., 1985). Eight treatments (four plant species at two elevations) were analyzed as a randomized block design (n=5) where elevation and plant species were blocked on distance from the bayou edge. All variables were tested for normality and homogeneity of variance to meet the assumptions of analysis of variance. All variables except interstitial Fe and Zn concentration were found to be normal with homogeneous variances. A square root transformation was used to normalize interstitial Fe and Zn concentration data prior to analysis. Untransformed means are presented.

Results

Biotic Response

Plant species responded differently to the change in elevation (interaction of plant species with elevation significant [$P < 0.01$]). At the ambient elevation of the deteriorating marsh, *D. spicata*, had the lowest total standing crop (biomass + necromass) and *S. alterniflora*, had the highest total standing crop ($P < 0.05$). The total standing crop of *J. roemerianus* and *S. patens* was intermediate between that of *D. spicata* and *S. alterniflora* (Figure 3.1).

When the surface of the deteriorating marsh was elevated 20 cm, the total biomass of *D. spicata* ($P < 0.01$) and *S. patens* ($P < 0.05$) significantly increased compared to that at the ambient elevation. The biomass of *S. alterniflora* and *J. roemerianus* was not significantly affected by an increase in elevation, although both species tended to have higher biomass when elevated 20 cm.

Abiotic Response

The surface soils (Figure 3.2a) of the elevated plots had significantly higher Eh's than those at the ambient elevation ($P < 0.01$) for all four plant species [plant species x elevation interaction significant ($P < 0.05$)]; at the 15 cm depth (Figure 3.2b), the soils of the elevated plots had significantly higher Eh's than the soils at the ambient elevation ($P < 0.01$).

Interstitial, pH, salinity, sulfide, $\text{NH}_4\text{-N}$, P, Fe, Mn, Cu and Zn concentrations did not significantly differ among plant species so means averaged over plant species are given for both elevations (Table 3.1). Interstitial sulfide, $\text{NH}_4\text{-N}$ and P, were significantly higher at the ambient elevation than when elevated ($P < 0.01$), while interstitial Fe, Cu and Zn were significantly higher in the elevated plots ($P < 0.01$). Interstitial Mn did not significantly differ with elevation ($P = 0.08$). Interstitial pH did not differ significantly ($P = 0.23$) between treatments, and the mean pH of all of the

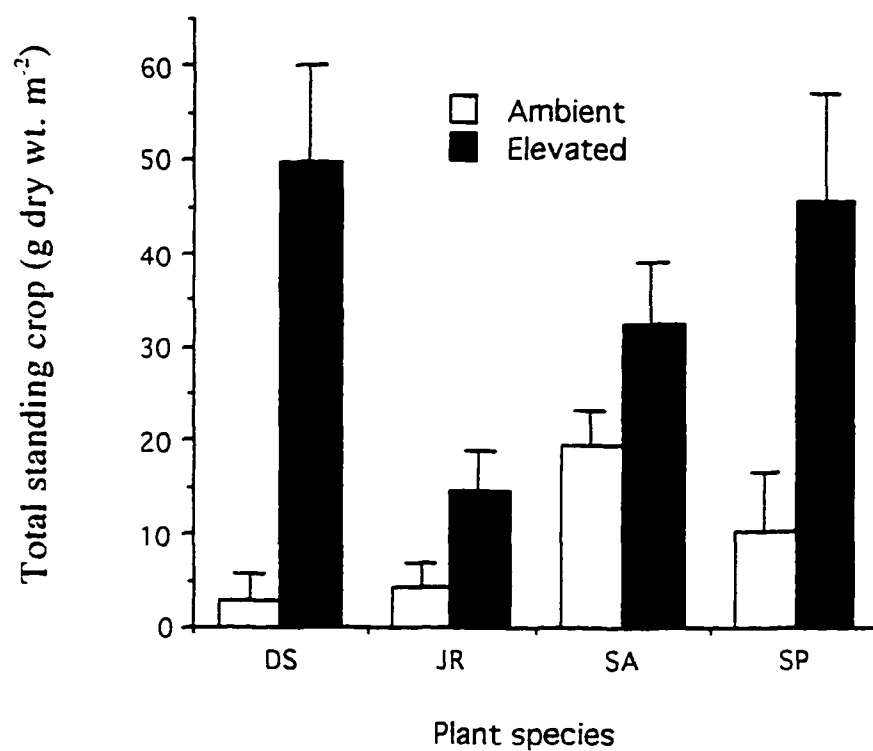


Figure 3.1. Total standing crop (biomass + necromass) at elevated and ambient elevations for all four plant species. Plant species: DS=*Distichlis spicata*, JR=*Juncus roemerianus*, SA=*Spartina alterniflora* and SP=*Spartina patens*. (mean \pm 1 standard error).

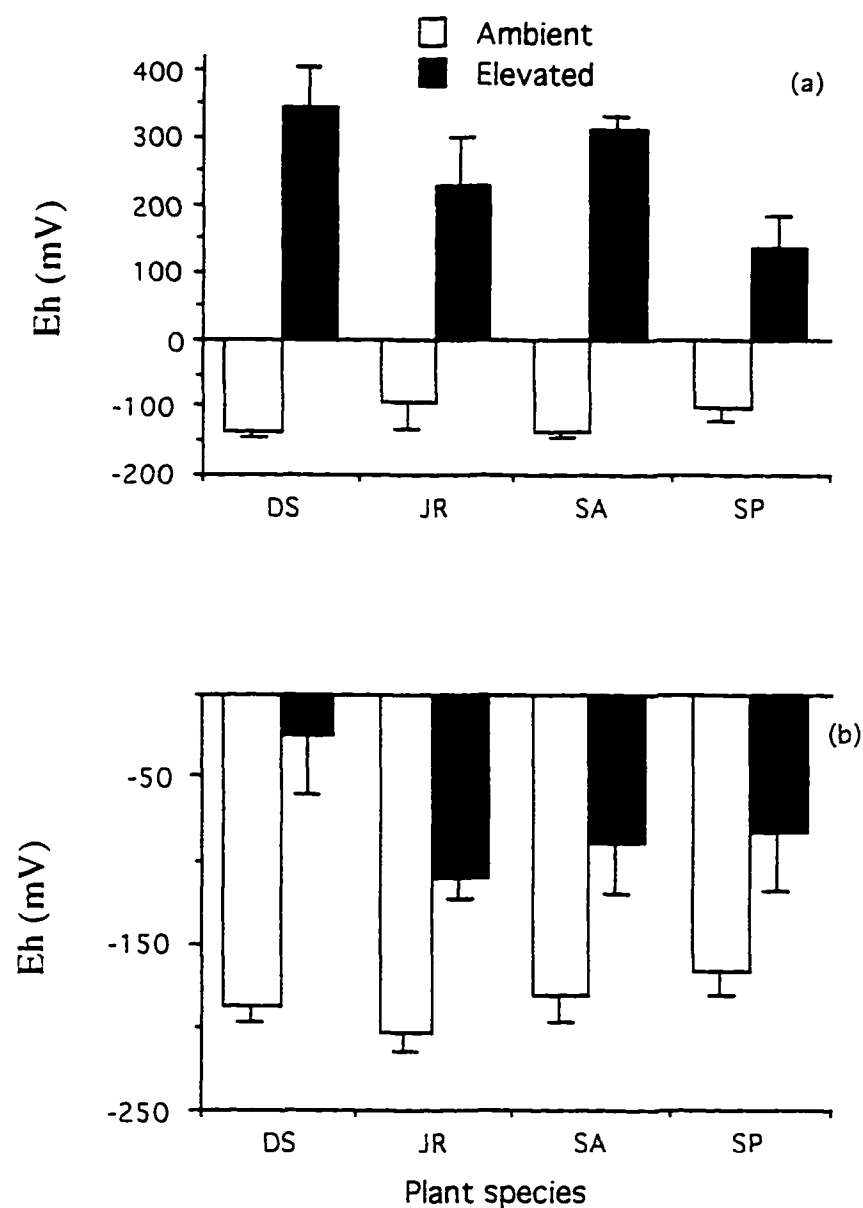


Figure 3.2. Redox potential (mV) at 2 cm depth (a) and at 15 cm depth (b) for elevated and ambient elevations for all four plant species. Plant species: DS=*Distichlis spicata*, JR=*Juncus roemerianus*, SA=*Spartina alterniflora* and SP=*Spartina patens*. (mean \pm 1 standard error).

Table 3.1. Interstitial sulfide (mM) and soil macro- and micronutrient concentrations (μM), pH and salinity (ppt). Mean values given for each elevation; parentheses indicate 1 standard error. Different letters following values indicate a significant difference at the 0.05 probability level.

Variable	Ambient		Elevated	
Sulfide	1.41 a	(0.20)	0.04 b	(0.02)
NH ₄ -N	450.0 a	(47.9)	81.4 b	(12.9)
P	87.5 a	(4.2)	32.0 b	(2.6)
Fe	9.1 a	(2.0)	44.2 b	(7.5)
Mn	23.1 a	(2.0)	28.1 a	(2.0)
Cu	0.315 a	(0.031)	0.472 b	(0.031)
Zn	0.153 a	(0.011)	0.316 b	(0.031)
pH	7.6 a	(0.1)	7.4 a	(0.1)
Salinity	12.6 a	(0.2)	12.4 a	(0.3)

sods was near neutral. Salinity did not differ between treatments ($P = 0.56$) and was relatively low for a tidal salt marsh (Table 3.1)

Discussion

The plant standing crop results were consistent with the flood tolerance of the four plant species. At the ambient elevation, *S. alterniflora*, the most flood tolerant plant species, had the highest biomass, while *D. spicata*, the least flood tolerant plant species, had the lowest biomass. When marsh elevation was increased to that equivalent to adjacent healthy marshes, only the least flood tolerant species, *D. spicata* and *S. patens* showed significant increases in standing crop.

Higher Eh's in the elevated plots at the surface and 15 cm depth indicates a decreased intensity of soil reduction at the higher elevation (Gambrell and Patrick, 1978). By

elevating the deteriorating marsh 20 cm, the soils became less reduced than at the ambient deteriorating marsh elevation.

As soils become more reduced, the production of potentially toxic compounds becomes more likely. One reduced compound which is commonly produced in salt marsh systems upon flooding and is a known phytotoxin is free sulfide (Mendelssohn and McKee, 1988). Sulfide concentrations at the ambient elevation averaged above 1 mM and were 35 times higher than at the + 20 cm elevation (Table 3.1). Sulfide concentrations above 1 mM have been shown to reduce the growth of salt marsh vegetation (Havill et al., 1985; Pearson and Havill, 1988; Koch and Mendelssohn, 1989).

The uptake of the most available form of N in salt marshes, $\text{NH}_4\text{-N}$, can be limited by high sulfide concentration (Koch et al., 1990; Bradley and Morris, 1990) and N is the most important limiting nutrient to salt marsh primary productivity (Valiela and Teal, 1974). Interstitial $\text{NH}_4\text{-N}$ concentration was greater in the ambient plots than in the plots elevated 20 cm. Interstitial sulfide concentration was also significantly higher at the ambient elevation of the deteriorating marsh. Because high sulfide concentrations and more reduced soil conditions are known to inhibit the uptake of $\text{NH}_4\text{-N}$ (Koch et al., 1990; Bradley and Morris, 1990), it is likely that the higher $\text{NH}_4\text{-N}$ concentration at the ambient elevation of the deteriorating marsh was due to decreased plant uptake of $\text{NH}_4\text{-N}$. This, in turn, could reduce plant growth (Mendelssohn and McKee, 1988; Koch et al., 1990). The more reduced soils at the ambient elevation would also prevent nitrification, hence, allowing for $\text{NH}_4\text{-N}$ accumulation (Gambrell and Patrick, 1978).

By increasing the elevation of the deteriorating marsh 20 cm, more oxidized conditions persisted and caused a decrease in sulfide concentration, possibly allowing for improved uptake of $\text{NH}_4\text{-N}$ by vegetation. However, other macronutrients such as P may be affected differently under more oxidized soil conditions. Phosphorus is

typically bound in insoluble complexes under aerobic conditions (Gambrell and Patrick, 1978). The more reduced soil conditions in the plots at the ambient elevation of the deteriorating marsh caused a higher soluble P concentration because P was likely released from insoluble complexes (Gambrell et al., 1991). Since greater soluble Fe concentrations were present in the elevated plots, the lower interstitial P concentration in the elevated plots may have been due to the precipitation of ferric phosphates under more oxidized conditions (Gambrell and Patrick, 1978).

Iron concentration was higher in the elevated plots than in the ambient plots and sulfide concentrations were lower in the elevated plots than in the ambient plots. Lower sulfide concentrations due to more aerobic conditions in the elevated plots decrease the binding and precipitation of Fe and Mn as insoluble metal sulfides (Gambrell and Patrick, 1978).

Raising the deteriorating marsh elevation 20 cm produced less reduced soils, decreased sulfide concentration and lower $\text{NH}_4\text{-N}$ concentrations the latter indicative of greater plant $\text{NH}_4\text{-N}$ uptake. These improved soil conditions led to an increase in total standing crop. The concentration of Fe, an element that can be important in precipitating toxic sulfides, increased with an increase in elevation of the deteriorating marsh surface. These results support the proposition that submergence is an important factor causing vegetation dieback in this system.

Salinities were not significantly different between ambient and elevated plots and did not explain any differences found in the biotic variables. If salinity was affecting vegetation dieback, then the most salt tolerant plant species, *D. spicata*, should grow equally well at the ambient elevation of the deteriorating marsh as at the elevated treatment. Instead the growth of *D. spicata* improved with increased elevation even though salinity did not differ between elevational treatments. This illustrated that plant growth was stimulated at natural salinities when flooding stress was removed.

Vegetation dieback in this marsh was not caused by increased salinity levels; instead, excessive submergence of the vegetation is the causative agent.

Vegetation dieback and subsequent wetland loss is threatening many marshes in coastal Louisiana. The high rate of relative sea level rise (Ramsey and Penland, 1989) and a sediment accretion deficit (DeLaune et al. 1978) are the main causes of increased plant submergence. In this study, plants growing at the ambient elevation of a deteriorating marsh experience highly reduced soils and toxic sulfide concentrations. Plants growing at an increased elevation were subjected to less reduced soil conditions, lower sulfide concentrations, apparently adequate $\text{NH}_4\text{-N}$ uptake and higher plant growth than at the ambient elevation. These results indicate that plant dieback and lack of plant recruitment is due to excessive plant submergence and the associated conditions that develop in highly reduced soils.

In salt marsh systems in coastal Louisiana, the stress from increased submergence associated with high rates of relative sea level rise is responsible for vegetation dieback. Management strategies, such as sediment additions that increase marsh elevation and relieve waterlogging stress, are needed to reduce coastal land loss rates in Louisiana.

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CHAPTER 4

AN INVESTIGATION OF THE FACTORS ASSOCIATED WITH VEGETATION DIEBACK IN A LOUISIANA OLIGOHALINE MARSH

Introduction

Louisiana has 40% of the conterminous United States' coastal wetlands, and is currently experiencing one of the most rapid rates of wetland loss in the world (Turner and Gosselink, 1975; Turner, 1990). The problem of wetland loss in Louisiana is well studied (Turner and Cahoon, 1987) and is known to be related to the high rate of relative sea level rise (land subsidence plus eustasy) in the Louisiana deltaic plain (Penland and Ramsey, 1990) and decreased sediment availability to coastal marshes (Kesel, 1987). Wetland loss is not only dominated by physical processes but it includes a biotic component as well. The dieback of vegetation is an important part of the wetland loss process and is characterized by the formation of large expanses of marsh separated by shallow ponds which coalesce through time forming larger lakes (Turner and Rao, 1990). Although the exact causes of vegetation dieback in low-salinity marshes are not known, several edaphic changes associated with the high rate of relative sea level rise in coastal Louisiana may be responsible. Increased plant submergence (Mendelssohn and Seneca, 1980; Howes et al., 1981; Mendelssohn and McKee, 1988), higher salinities (Pezeshki et al., 1987a; McKee and Mendelssohn, 1989), increased sulfides (Koch and Mendelssohn, 1989; Koch et al., 1990) and decreased mineral matter availability (DeLaune et al., 1979) can be debilitating to wetland vegetation, but how each of these processes affect vegetation dieback and subsequent wetland loss in low-salinity marshes of coastal Louisiana are not well understood.

The goal of this study was to determine the factors responsible for vegetation dieback within a low-salinity marsh in coastal Louisiana. In order to meet this objective, biotic and edaphic characteristics were measured during a two-year period within deteriorating

marshes and compared to biotic and edaphic characteristics of a relatively healthy, well-vegetated oligohaline marsh. Also, in order to assess possible effects of backfilled pipeline canals on vegetation dieback, biotic and edaphic characteristics of two deteriorating marshes, one with backfilled pipeline canals and one without backfilled pipeline canals, were compared.

Methods

Study Site Location

For this study of the causal mechanisms of vegetation dieback within low-salinity marshes of coastal Louisiana, three adjacent oligohaline marsh sites within the Pointe-au-Chien Wildlife Management Area, ca. 10 km southwest of LaRose, La were chosen. The marshes were selected because of the proximity of two deteriorating marsh sites, one bisected by backfilled pipeline canals, to a healthy, well-vegetated marsh site.

Field monitoring was conducted for two years in twenty-four, 4 m x 5 m permanent plots located randomly within the three marsh sites (Figure 4.1): a healthy, well-vegetated marsh dominated by *Sagittaria lancifolia* (hereafter referred to as HB, n=8), a deteriorating marsh bisected by a backfilled pipeline canal and dominated by *Spartina patens* and *S. lancifolia* (hereafter referred to as DB, n=8), and a deteriorating marsh dominated by *S. lancifolia* but not bisected by pipeline canals (hereafter referred to as DC, n=8). Site HB was located north of DB and sites HB and DB were separated from site DC by Grand Bayou Canal, a large straight line canal. All permanent plots were located at random distances perpendicular and parallel to bisecting water bodies within each marsh site. This compensated for any variation that occurred from the marsh edge into the marsh interior along natural water bodies and canals.

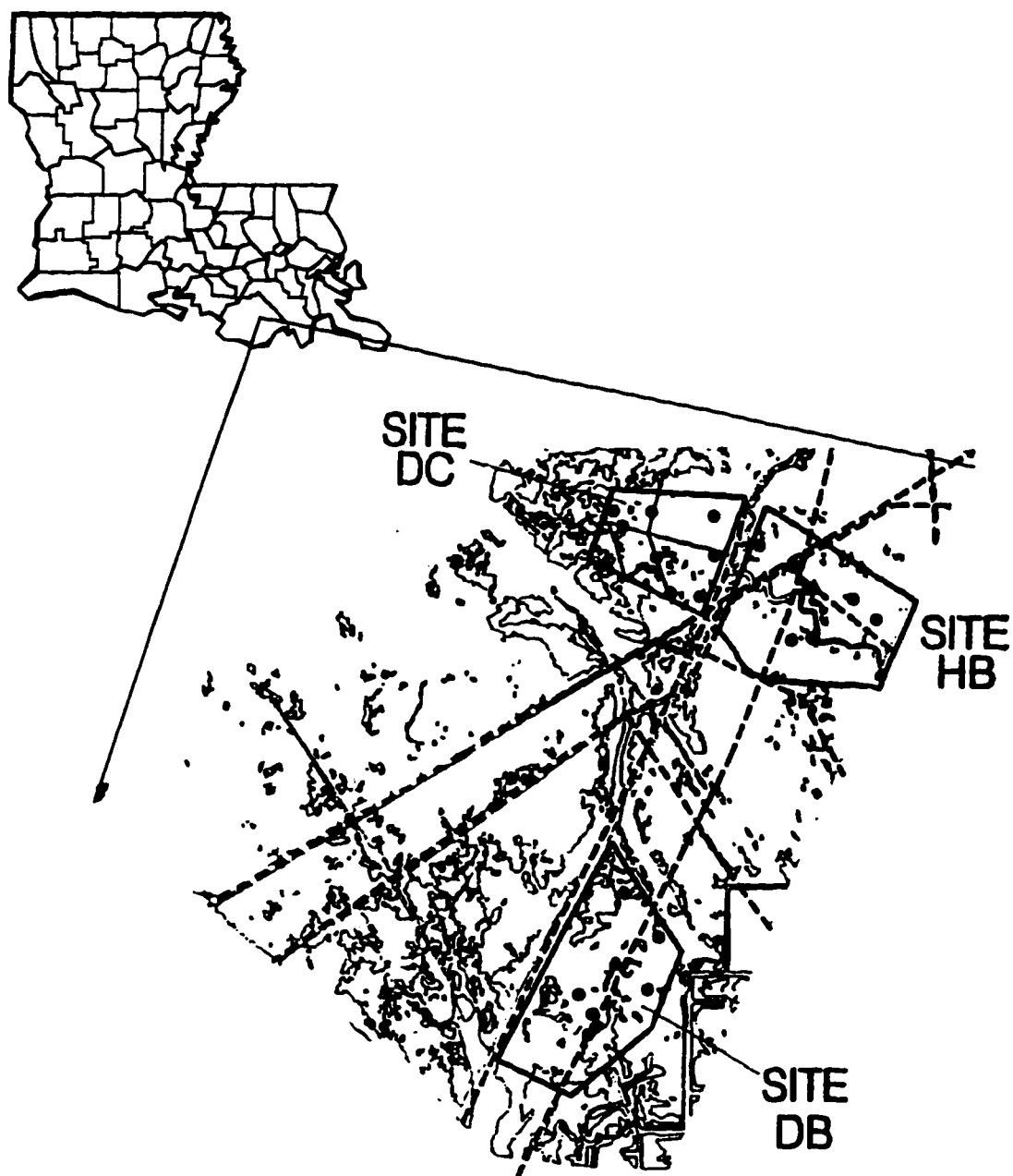


Figure 4.1. Location of reference marsh site (HA, $n=8$), deteriorating marsh site bisected by backfilled pipeline canals (DB, $n=8$), and deteriorating marsh site without backfilled pipeline canals (DC, $n=8$).

Sampling Procedure

Within each 4 m x 5 m permanent plot, specific biotic and abiotic variables were sampled at randomly located 0.50 m² sites. The selected sites were marked after each sampling to insure that no site was sampled twice.

Aboveground biomass was harvested at ground level from a 0.50 m² quadrat. The aboveground biomass was transported to the laboratory where it was separated by plant species into live and dead components, dried to a constant weight at 65^o C for 48 hours, and dry weight determined gravimetrically to the nearest 0.1 g. Within a 1 m² permanent quadrat, located in each 4 m x 5 m permanent plot, ocular estimates of percent plant cover and species composition were determined.

Redox potential measurements were taken in an undisturbed area adjacent to the 0.50 m² clip plot. Redox potential was determined at each plot at two depths (2 cm and 15 cm) with three replications for each depth using bright platinum electrodes, a calomel reference electrode and a pH/mV meter. The potential of the calomel reference electrode against a standard hydrogen electrode (+245 mV) was added to the measured potential in order to determine Eh.

Within the 0.5 m² clip plot, two separate soil cores were taken: one to measure soil bulk density and another for interstitial water analysis. Soil bulk density was determined by collecting a 190.5 cm³ soil core from from the root zone of each plot. Prior to inserting the coring device, the soil was cut with a knife to the same diameter and depth as the coring device and the corer was inserted into the groove. This method minimized compaction of the core. The core was transported to the laboratory, dried at 65^o C to a constant weight and weighed to the nearest 0.1 g.

For interstitial water analysis, a soil core was removed from each plot using an aluminum coring device 15 cm long and 5 cm in diameter. Immediately upon removal,

the core was placed into a 500 ml centrifuge tube, sealed and purged with N₂ gas through a rubber septum for two minutes in order to maintain an anaerobic environment and placed on ice and transported to the laboratory. The soil core was centrifuged at 10,000 g at 4°C to collect interstitial water. An aliquot of unfiltered porewater, after centrifugation, was immediately added to antioxidant buffer and dissolved sulfide concentration was determined (Sulfide electrode, Lazar Research Laboratories, Los Angeles, CA). A second aliquot of unfiltered water was analyzed for pH (Altex Model 3560 digital pH meter) and salinity (Fisher Model 152 salinity/conductivity meter). A third aliquot was filtered through a 0.45 µm Uniflo syringe tip filter (Schleicher and Schull) preserved with nitric acid and analyzed for P concentrations using a Fisher inductively coupled argon plasma emission spectrometer (Atom Comp Series 800). A fourth aliquot was preserved with sulfuric acid (U S EPA, 1974) and analyzed for NH₄-N concentration using the Technicon Industrial Method at the Institute of Ecology, Soil Biology Laboratory, University of Georgia.

Samples were obtained three times within the first year of the study: May 10/11, 1991, July 26/27, 1991 and September 5/6, 1991, and twice within the second year of the study: July 23/24, 1992 and September 23/24, 1992.

Statistical Analysis

All statistical analysis was computed using SAS (GLM Procedure, CORR Procedure, Statistical Analysis Systems, 1985). The study was analyzed as a split plot design with time, treatment and distances from canals, bayous and water bodies as main effects. Differences between healthy and deteriorating marshes as well as differences between the deteriorating marsh with pipeline canals and the deteriorating marsh without pipeline canals were tested using contrast statements. Tests for normality and homogeneity of variance were conducted on all variables. All variables were normal with homogeneous

variances with the exception of interstitial P concentration. Interstitial P concentration was transformed with a reciprocal transformation. Untransformed means are presented.

Results

Prior to the first sampling it was observed that vegetation dieback was occurring within the two deteriorating marsh sites; the vegetation was not continuous, and shallow open ponds were common throughout. As well as vegetation dieback, a difference in community structure within the deteriorating marshes was apparent. *Sagittaria lancifolia* was the dominant plant species at the reference site. In contrast, the deteriorating marshes were dominated by brackish marsh plant species such as *Spartina patens* and *Paspalum vaginatum*. Collectively, the live biomass of all plant species (Figure 4.2) did not differ significantly between the reference marsh and the deteriorating marshes ($P = 0.38$). However, the deteriorating site without pipeline canals had significantly lower live biomass than deteriorating site with pipeline canals ($P < 0.05$). Live biomass was highest in May, 1992 and lowest in October, 1992 ($P < 0.05$). The marsh site x sampling date interaction was not significant ($P = 0.66$).

The greatest species diversity occurred at the reference site. Although multiple plant species were found at all three marsh sites, each marsh had a different suite of plant species (Table 4.1). *Sagittaria lancifolia* was dominant in site HB and *S. lancifolia* and *P. vaginatum* were present in significant proportions at site DC. *Spartina patens* (Figure 4.3) had higher biomass at DB than at DC ($P < 0.01$) and higher than the reference marsh ($P < 0.01$). *Paspalum vaginatum* live biomass was highest in the deteriorating marsh with pipeline canals ($P < 0.05$) and higher in the deteriorating marshes than in the reference marsh ($P < 0.05$). Live plant cover did not differ significantly between marshes on any sampling date ($P = 0.84$).

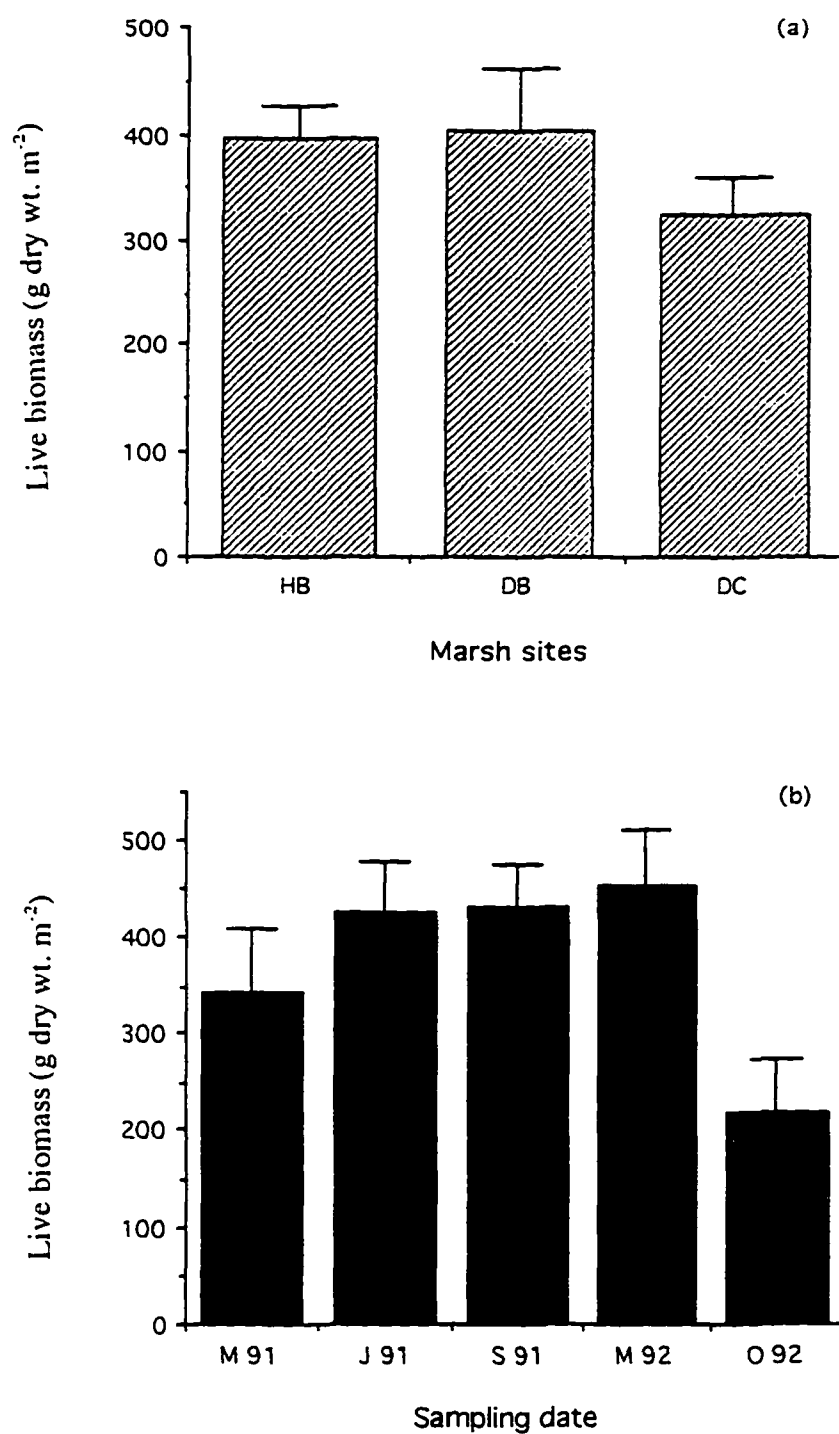


Figure 4.2. Aboveground live biomass by marsh site averaged across all sampling dates (a) and by sampling date averaged across all marsh sites (b). Bars represent mean live biomass \pm 1 standard error.

Table 4.1. Relative plant species composition by marsh site for all plant species with more than one occurrence. Dominance for each plant species given as mean percentage of total biomass; 1 standard error of the mean in parentheses.

Plant Species	HB		DB		DC	
<i>Spartina patens</i>	0	(0)	46.7	(4.0)	0.6	(0.3)
<i>Distichlis spicata</i>	0	(0)	23.7	(3.1)	0	(0)
<i>Lythrum lineare</i>	0.1	(0)	3.4	(0.7)	2.7	(1.5)
<i>Spartina alterniflora</i>	0	(0)	0.1	(0.04)	0	(0)
<i>Kosteletskyia virginica</i>	0.2	(0.05)	0.3	(0.1)	2.0	(0.8)
<i>Sagittaria lancifolia</i>	47.6	(2.6)	9.4	(2.0)	34.1	(3.8)
<i>Eichornia crassipes</i>	0	(0)	0	(0)	3.2	(1.6)
<i>Sacciolepis striata</i>	13.4	(2.0)	0	(0)	8.3	(1.8)
<i>Echinochloa walteri</i>	0	(0)	0	(0)	0.2	(0.1)
<i>Panicum rapens</i>	0	(0)	0	(0)	0.8	(0.4)
<i>A. philoxeroides</i>	22.2	(1.9)	0	(0)	0.3	(0.1)
<i>Paspalum vaginatum</i>	0	(0)	3.5	(1.2)	28.0	(4.2)
<i>Hydrocotyle</i> sp.	0.2	(0.07)	0	(0)	0.1	(0.03)
<i>Eleocharis</i> sp.	0.3	(0.1)	0.1	(0.03)	1.6	(0.5)
<i>Phyla nodiflora</i>	0.2	(0.4)	0	(0)	2.7	(0.6)
<i>Galium tinctorium</i>	0.1	(0.03)	0	(0)	0	(0)
<i>Eleocharis fallax</i>	4.5	(0.8)	6.9	(1.2)	2.4	(0.6)
<i>Leptochloa fascicularis</i>	0.3	(0.1)	0.1	(0.06)	0	(0)
<i>Panicum</i> sp.	0	(0)	0	(0)	0.1	(0.04)
<i>Polygonum sagittata</i>	0	(0)	0	(0)	2.7	(1.6)
<i>Polygonum punctatum</i>	8.4	(1.9)	1.5	(0.6)	3.2	(1.2)
<i>Bacopa monieri</i>	0.1	(0.03)	0	(0)	5.2	(1.7)
<i>Ipomaea sagittata</i>	0	(0)	0.5	(0.3)	0	(0)
<i>Scirpus olneyi</i>	0.5	(0.2)	0.4	(0.2)	0	(0)
<i>Ptilimnium capillaceum</i>	0.2	(0.07)	0	(0)	0	(0)
<i>Cyperus odoratus</i>	0	(0)	2.5	(0.7)	1.7	(0.8)
<i>Vigna luteola</i>	0.1	(0.02)	0.1	(0.05)	0	(0)

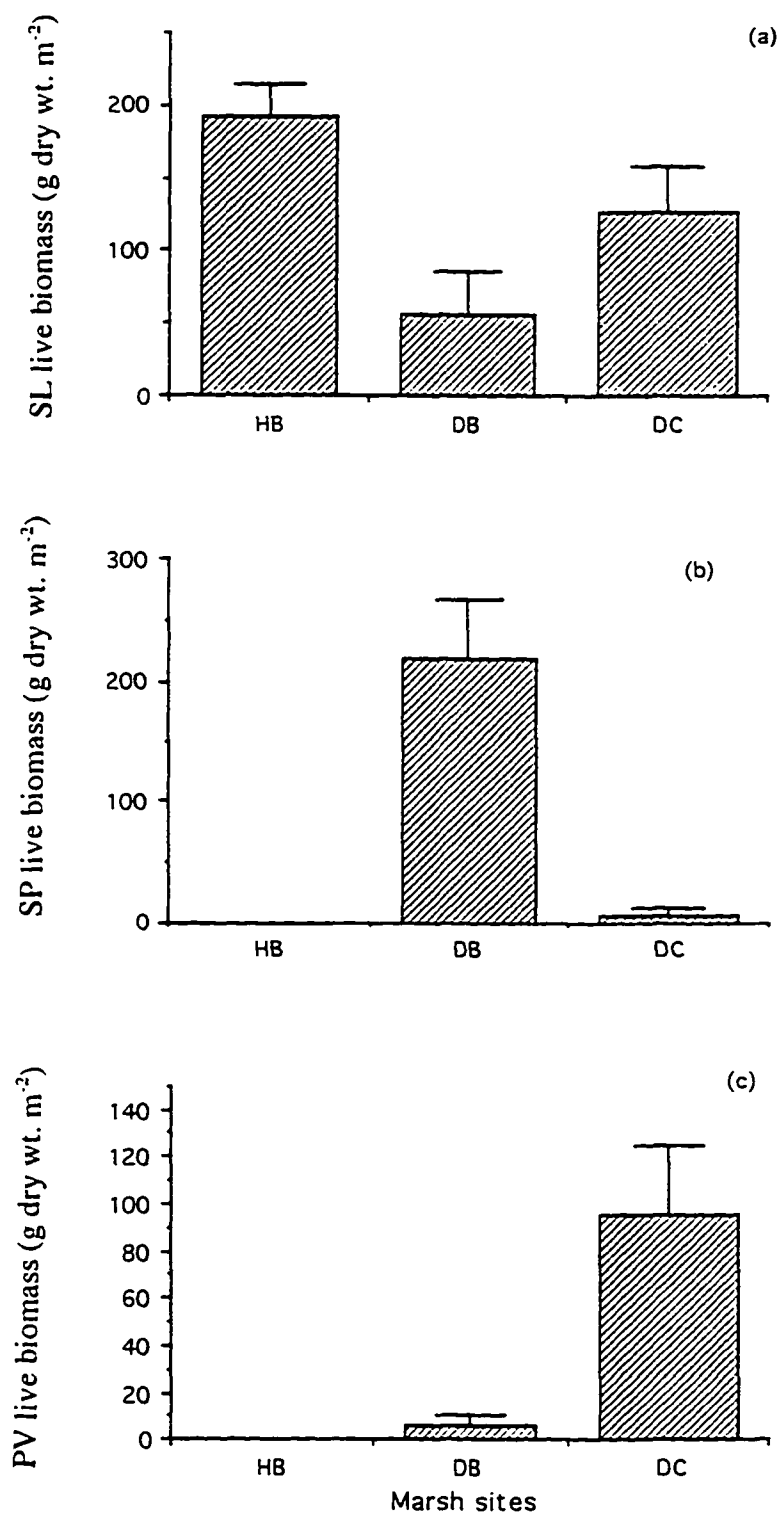


Figure 4.3. Aboveground live biomass by marsh site averaged across all sampling dates for *S. lancifolia* (a), *S. patens* (b), and *P. vaginatum* (c). Bars represent mean live biomass \pm 1 standard error.

Soil redox potential at 2 cm depth (Figure 4.4a) was highly variable. Except for September, 1991, surface soils were less reduced at the reference site [marsh site x sampling date interaction significant ($P < 0.01$)].

Soil Eh at 15 cm depth (Figure 4.4b) was higher at the reference marsh than at the deteriorating marshes on all dates except September, 1991 [marsh location x sampling date interaction significant ($P < 0.01$)]. The deteriorating sites did not differ with respect to soil redox potential at 15 cm depth.

Interstitial sulfide concentration (Figure 4.5) was higher at the deteriorating marshes than at the reference marsh [significant sampling date x marsh site interaction ($P < .05$)] but no significant difference was found between the two deteriorating marsh sites on any sampling date. The highest interstitial sulfide concentration at all three marsh sites occurred on October, 1992.

Interstitial $\text{NH}_4\text{-N}$ concentration (Figure 4.6) did not differ between marsh sites but did differ significantly between sampling dates ($P < 0.05$). The highest $\text{NH}_4\text{-N}$ concentrations occurred in the fall sampling of each year (September, 1991 and October, 1992).

Interstitial P concentration (Figure 4.7) was significantly lower at the reference marsh than at the deteriorating marshes on all sampling dates except May, 1991 [significant sampling date x marsh site interaction ($P < 0.01$)].

Interstitial pH (Figure 4.8) was lower at the reference marsh than at the deteriorating marshes on May, 1991 and May, 1992. Site DC had a lower pH than DB on May, 1991 and May, 1992 [marsh location x sampling date interaction significant ($P < 0.01$)]. Although differences were found between treatments through time, the range of interstitial pH was small.

The reference marsh had lower interstitial salinities (Figure 4.9) than the deteriorating marshes on all sampling dates except September, 1991. Site DB had higher interstitial

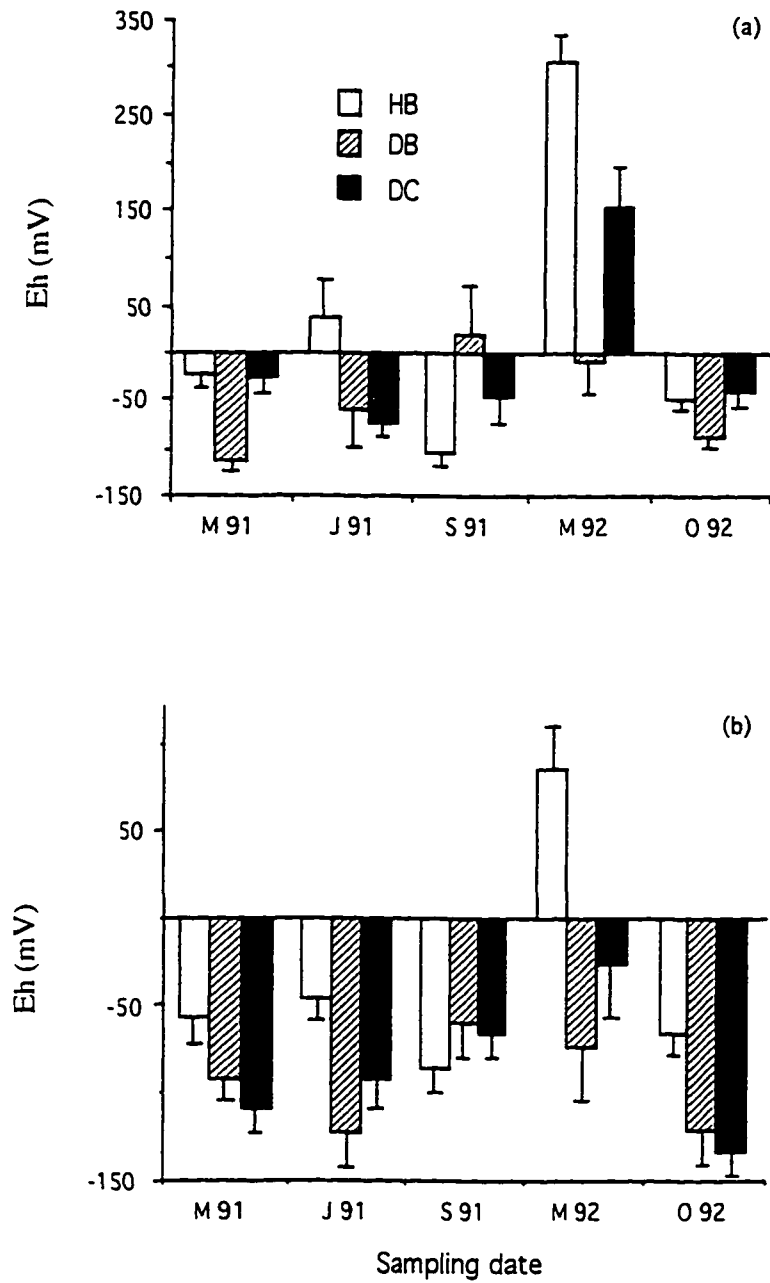


Figure 4.4. Redox potential at 2 cm depth (a) and at 15 cm depth (b) by marsh site for each sampling date. Bars represent mean ± 1 standard error.

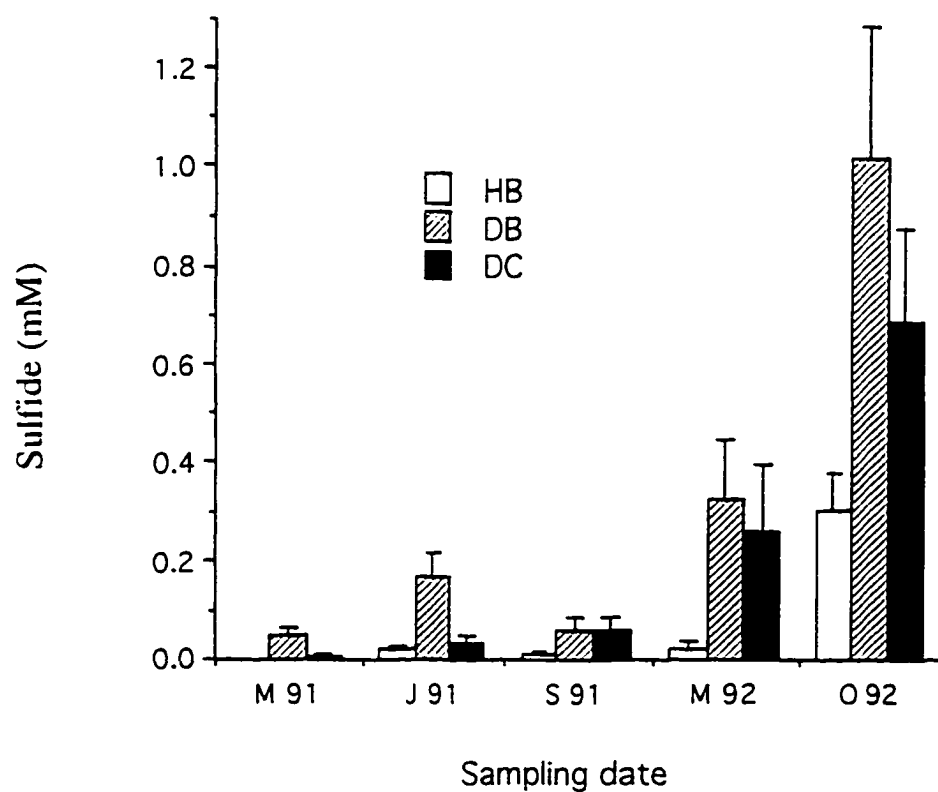


Figure 4.5. Interstitial sulfide concentration by marsh site for each sampling date. Bars represent mean ± 1 standard error.

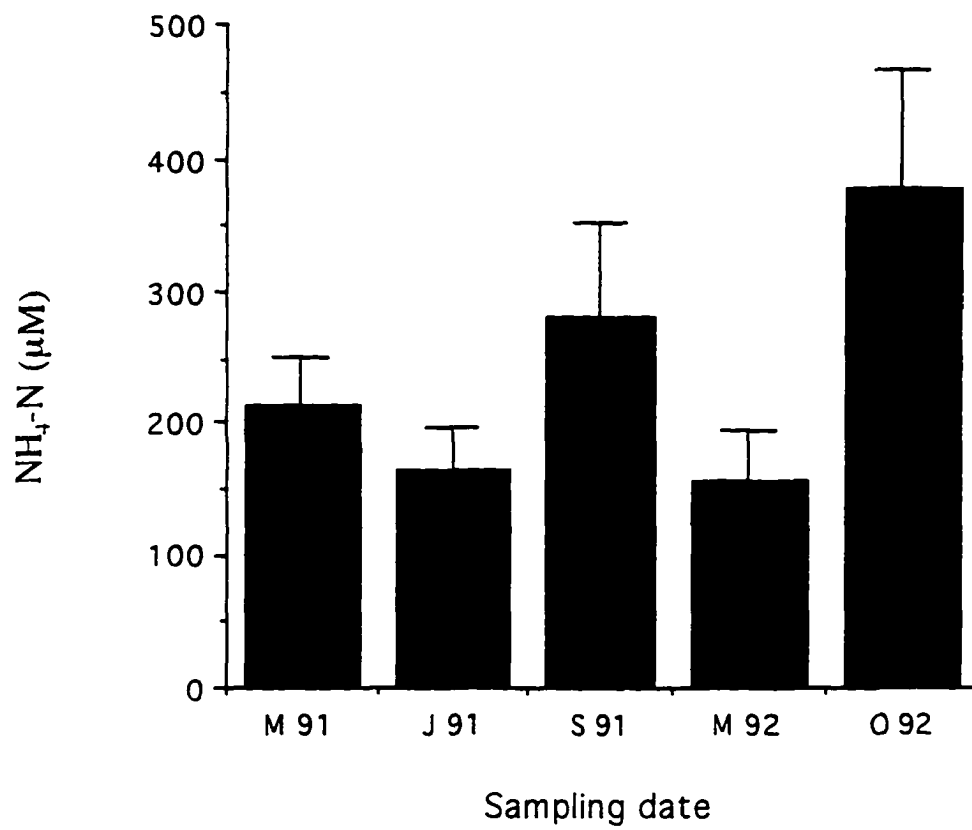


Figure 4.6. Interstitial $\text{NH}_4\text{-N}$ concentration by sampling date averaged across all marsh sites. Bars represent mean ± 1 standard error.

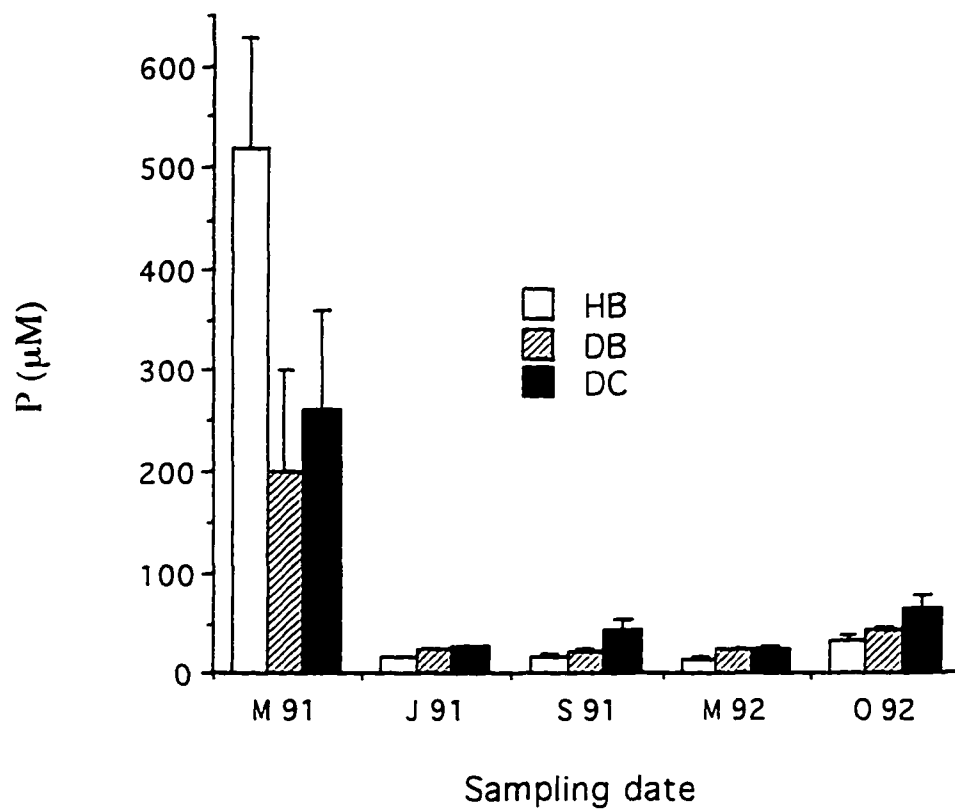


Figure 4.7. Interstitial P concentration by marsh site for each sampling date. Bars represent mean \pm 1 standard error.

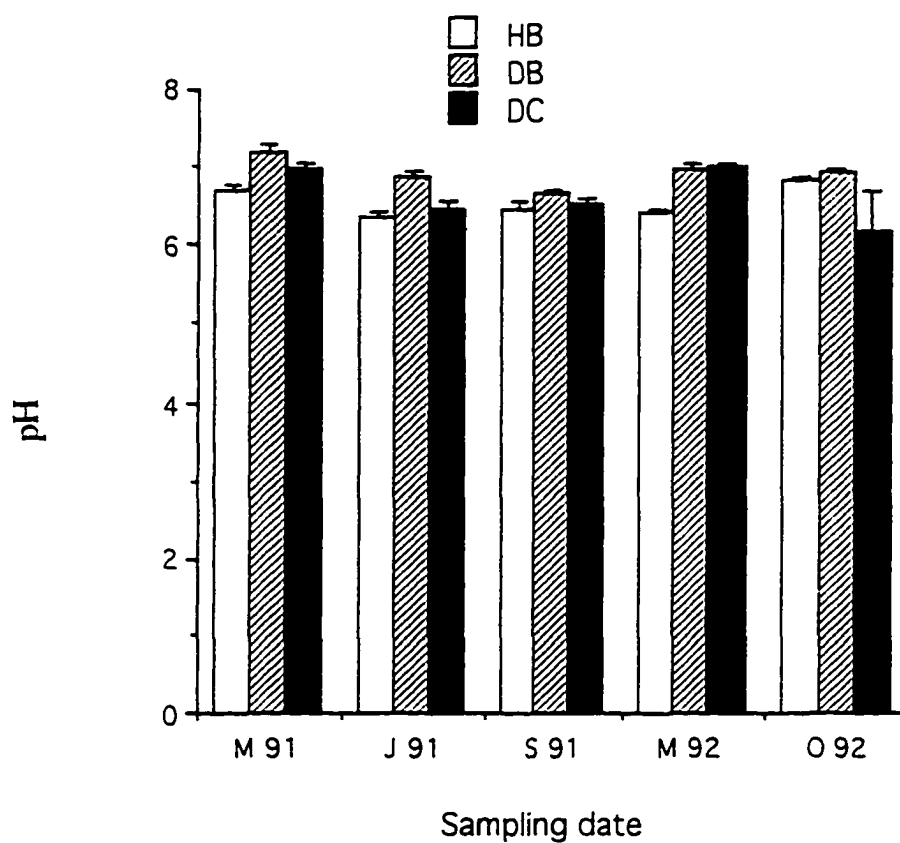


Figure 4.8. Interstitial pH by marsh site for each sampling date. Bars represent mean ± 1 standard error.

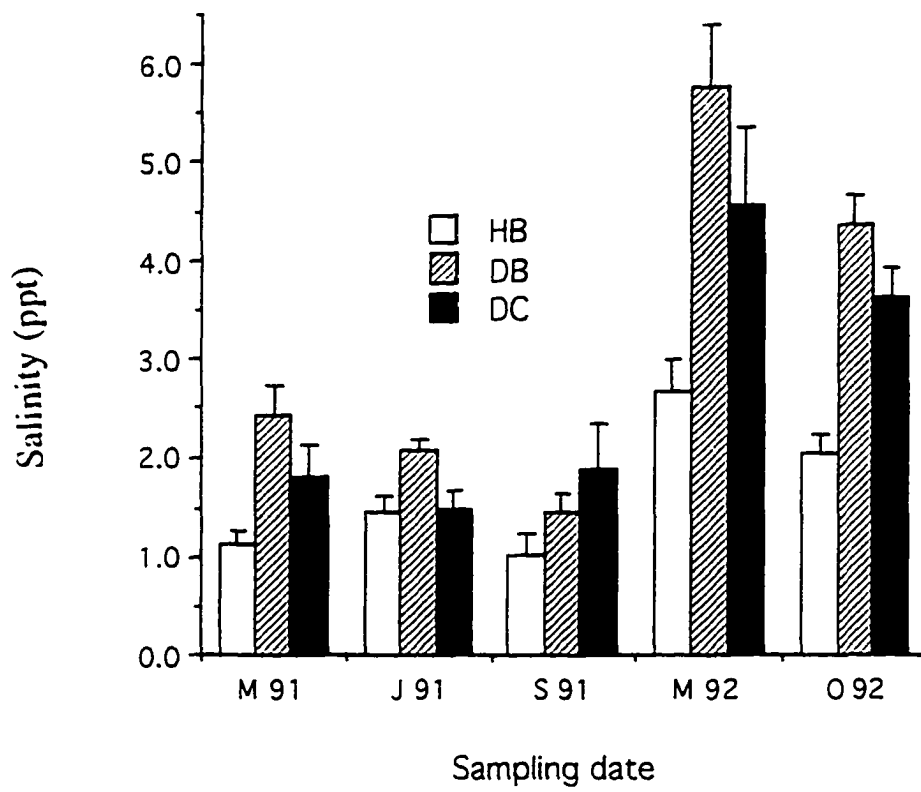


Figure 4.9. Interstitial salinity by marsh site for each sampling date. Bars represent mean ± 1 standard error.

salinities than site DC on July, 1991 [marsh location x sampling date interaction significant ($P < 0.01$)]. Interstitial salinities were highest on May, 1992 and October, 1992 ($P < 0.01$).

The soil bulk density (Figure 4.10) of the reference marsh was significantly lower than the deteriorating marshes ($P < 0.01$) but soil bulk density did not differ between the two deteriorating sites. Site DB had the highest average soil bulk density on all sampling dates (marsh site x sampling date interaction significant [$P < 0.01$]).

Discussion

Reference Marsh vs. Deteriorating Marshes

Before the study was initiated, differences in the vegetative health of the reference and deteriorating marshes were observed but these differences were not apparent from the live biomass and live plant cover results. Instead we found a dieback of the *S. lancifolia* dominated marsh vegetation and a replacement of this association by a *S. patens* dominated, brackish marsh vegetation. Although this supports our initial observation that the dieback of the *S. lancifolia* dominated marsh community was occurring in sites DB and DC we hypothesize that oligohaline marsh vegetation does not always convert directly to open water but instead a transition in community structure may occur first. This is supported by Chabreck and Linscombe (1982) who have found a northward movement of more saline marsh types since 1949 in coastal Louisiana.

It is generally accepted that *S. patens* is an indicator plant species of brackish marsh habitat types (Chabreck and Linscombe, 1982). The presence of *S. patens* within the deteriorating sites would be indicative of changes in vegetational structure caused by saltwater intrusion; this change in plant community structure corresponded to the higher salinities found within the deteriorating marsh sites. Wang (1988) has shown that straight line canals cause increased penetration of saltwater into marshes; the deteriorating sites were farther south than the reference site and were located adjacent to

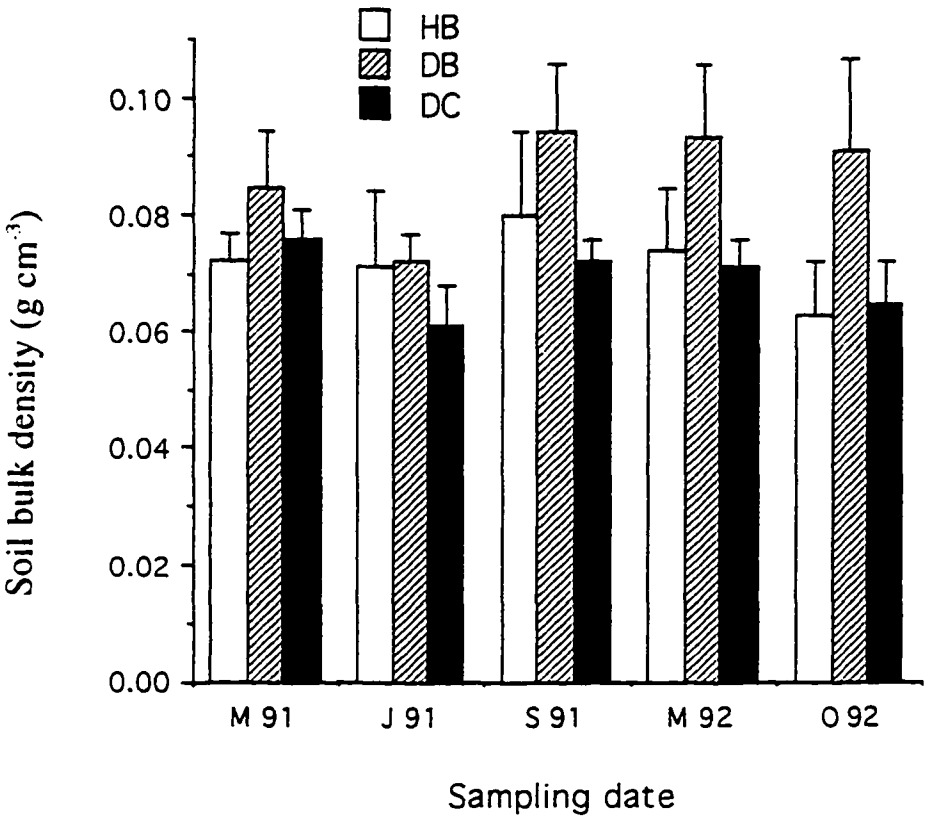


Figure 4.10. Soil bulk density by marsh site for each sampling date. Bars represent mean \pm 1 standard error.

a large navigation canal (Figure 4.1) making the deteriorating sites closer to a source of saltwater. HB is separated hydrologically from straight canals by a meandering bayou (Bayou Blue), which likely slows the penetration of saltwater. This explains the lower salinities that occurred throughout the study period at the reference marsh, even on the sampling date (October 1992) following the passage of Hurricane Andrew (August 1992). In Louisiana no consistent trend of increasing salinities has been found in long-term salinity studies (Fuller et al., 1990). After 30 to 40 days of exposure at salinities of 3 ppt, net carbon assimilation in *S. lancifolia* was reduced 45% (Pezeshki et al., 1987a) and salinities as low as 4.8 ppt have been shown to cause the reduced growth of *S. lancifolia* (McKee and Mendelssohn, 1989). Salinities between 5-7 ppt decreased the stomatal conductance and photosynthesis of *P. hemitomon*, a dominant freshwater marsh plant species in Louisiana (Pezeshki et al., 1987b). In this study the salinity differences between marshes corresponded to the differences in vegetation composition between marshes. The *S. patens*/*P. vaginatum* dominated community was found to be replacing the *S. lancifolia* dominated community in the higher salinity, deteriorating marshes, but not in the reference marsh where lower salinities persisted. This is evidence of saltwater intrusion causing the dieback of low-salinity marsh plant species in coastal Louisiana.

Saltwater intrusion can cause, in part, the formation of sulfides to occur concurrently with increased salinities, compounding the stresses on wetland vegetation. When seawater influence increases, an increase in sulfate also occurs (Riley and Skirrow, 1975) and under more reduced soil conditions, sulfate will be reduced to sulfides (Gambrell and Patrick, 1978). Sulfides are toxic to both freshwater marsh and salt marsh plant species (Goodman and Williams, 1961; King et al., 1982; Havill et al., 1985; Koch and Mendelssohn, 1989) but likely cause more reduced growth in freshwater marsh plant species than in salt marsh plant species (Koch and Mendelssohn,

1989). The deteriorating marshes experienced saltwater intrusion, more reduced soil conditions at 15 cm depth, and higher sulfide concentrations. It is likely that increased sulfides as well as increased salinities are responsible for the dieback of the *S. lancifolia* dominated marsh vegetation in the deteriorating sites.

More reduced soil conditions and increased sulfides have been shown to inhibit the uptake of $\text{NH}_4\text{-N}$ by several wetland plant species (Koch et al., 1990; Bradley and Morris, 1990), including the freshwater marsh plant *P. hemitomon*. If an inhibition of $\text{NH}_4\text{-N}$ uptake occurs, increased interstitial $\text{NH}_4\text{-N}$ concentrations would be expected. Concentrations of $\text{NH}_4\text{-N}$ did not differ between marshes. Since more salt tolerant plant species (*S. patens* and *P. vaginatum*) were found in the deteriorating sites, these plants may not be as stressed by higher salinities and sulfides as the *S. lancifolia* dominated marsh vegetation. *Spartina alterniflora*, a salt marsh plant species, has been shown to be more tolerant of higher sulfide concentrations than the freshwater marsh plant *P. hemitomon* (Koch and Mendelssohn, 1989; Koch et al., 1990), therefore the uptake of $\text{NH}_4\text{-N}$ by more salt tolerant plant species such as *S. patens* may not be inhibited by higher sulfide concentrations found within the deteriorating marshes.

Lower salinity marshes typically have lower soil bulk densities since organic matter makes up a greater percentage of the soil volume (Nyman et al., 1990). Higher soil bulk density within the deteriorating marshes corresponded to the higher salinities and more salt tolerant plant species within the deteriorating marshes. The higher soil bulk density is another indicator of the permanent changes to the deteriorating marshes caused by the increased salinity and sulfide concentration. Collectively these changes indicate that the dieback of *S. lancifolia* dominated marsh vegetation does not always lead to open water formation and wetland loss, but instead can cause a change to a more salt tolerant plant community structure.

Pipeline Canal Effects

Approximately 8% of Louisiana's coastal marshes have been converted into open water canals and associated spoil banks (Neill and Turner, 1987). Most of these canals were created for servicing the oil and gas industry (Adams et al., 1976). Backfilling of canals has become a method of mitigating effects from canal construction, yet canals typically remain unvegetated after backfilling (Neill and Turner, 1987). Although the majority of marsh vegetation in coastal Louisiana grows between 15 cm below to 15 cm above mean water level (Sasser, 1977) and backfilled canals have water depths less than 50 cm deep, less than 10% vegetative cover is typical of an average canal five years after backfilling (Neill and Turner, 1987). In this study, the effect of backfilled pipeline canals on marsh deterioration can only be determined through correlative data. Marsh site DB, bisected by two backfilled pipeline canals was dominated by the brackish marsh plant *S. patens*. Although site DB had the highest live biomass, it had the lowest *S. lancifolia* biomass. This indicates that DB experienced the greatest dieback of low-salinity marsh vegetation of the two deteriorating sites. Also, the backfilled pipeline canals may accelerate the movement of more saline waters into this marsh thereby causing the dieback of *S. lancifolia* dominated marsh vegetation, but confounding effects are present at site DB. Site DB is farthest south of the three sites placing it closer to a source of saline waters, and site DB is located adjacent to Grand Bayou Canal, a large, straight-line canal. Although these factors are important to the movement of saline waters into site DB, the presence of backfilled pipeline canals may exacerbate the problem of saltwater intrusion since pipeline canals allow for a direct hydrologic connection to more saline waters (Wang, 1988).

Conclusion

Increased salinities and sulfides were likely responsible for the dieback of the *S. lancifolia* dominated marsh community. Saltwater intrusion is the direct cause of the

increased salinities and is also responsible for the increase in toxic sulfides. The dieback of low-salinity marsh vegetation was not followed by a conversion to open water. Instead, the *S. lancifolia* dominated marsh community was replaced by a brackish marsh community dominated by *S. patens*. Backfilled pipeline canals may accelerate the movement of more saline waters into the low salinity marsh environment causing increased damage to the *S. lancifolia* dominated marsh vegetation. Only restoration techniques that involve both decreasing saltwater intrusion and increasing marsh elevations (Chapter 5) will be successful in combatting the complex problem of vegetation dieback in Louisiana's oligohaline marshes.

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CHAPTER 5

FACTORS AFFECTING VEGETATION DIEBACK OF AN OLIGOHALINE MARSH IN COASTAL LOUISIANA: FIELD MANIPULATION OF SALINITY AND SUBMERGENCE*

Introduction

Louisiana's coastal marshes were formed during the last 7000 y as the Mississippi River switched course forming a series of overlapping, delta lobes. The process of delta creation and abandonment has formed approximately 1.18 million ha of coastal wetlands of which seventy percent are oligohaline marshes (Turner, 1990). Due to subsidence and the lack of sediments available to coastal marshes, the relative sea level rise rate for the deltaic plain is as high as 1.19 cm y⁻¹ (Penland and Ramsey, 1990).

Wetland loss rates in coastal Louisiana are estimated at 65.6 km² y⁻¹ (Dunbar, Britsch and Kemp, 1992) and are related to subsidence of the deltaic plain, decreased sediment availability and increased oceanic influences (Morgan 1977; Kesel, 1987; Turner and Cahoon, 1987; Penland and Ramsey, 1990). An important component of the wetland loss process in Louisiana is vegetation dieback, which primarily occurs within the marsh interior (May and Britsch, 1987; Turner and Cahoon, 1987). During this process, vegetation dies and vegetated marsh converts to shallow ponds that coalesce to form larger lakes, resulting in wetland loss (Turner and Rao, 1990). Although the general mechanisms of wetland loss have been identified (Turner and Cahoon, 1987), the vegetation dieback process is not as well understood, especially for oligohaline coastal marshes.

The processes that can affect the dieback of vegetation within low-salinity tidal marshes are complex. Stresses associated with excessive plant submergence, due to marsh subsidence, caused decreased plant growth in *Panicum hemitomon* Schultes., *Leersia oryzoides* L. (Swartz.) and *Sagittaria lancifolia* L., three dominant freshwater

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marsh species of coastal Louisiana (McKee and Mendelssohn, 1989). Also, flooding can reduce the germination of many freshwater marsh plant species (van der Valk, 1981; van der Valk and Davis, 1978; Galinato and van der Valk 1986; Baldwin, McKee and Mendelssohn, 1996), although some species, e.g., *S. lancifolia*, are still able to germinate when flooded (Baldwin, McKee and Mendelssohn, 1996).

In addition to submergence, saltwater intrusion is often cited as a cause of vegetation dieback in coastal Louisiana, although few studies have documented this process in oligohaline marshes. Using historical maps of dominant marsh vegetation types in Louisiana, Chabreck and Linscombe (1982) have reported the movement of more saline marsh types in a northward direction since 1949. Also, under greenhouse conditions, salinities as low as 4.8 ppt decreased the growth of *S. lancifolia*, although other freshwater and oligohaline marsh plant species such as *P. hemitomon* and *L. oryzoides* tolerated salinities as high as 9.4 ppt (McKee and Mendelssohn, 1989). In the field, McKee and Mendelssohn (1989) transplanted fresh marsh sods containing *S. lancifolia*, *P. hemitomon*, and *L. oryzoides* to a brackish marsh in order to identify the effects of saltwater intrusion on freshwater marsh vegetation. A sudden and unexpected salinity increase to 15 ppt rapidly killed these species. However, the effects of a nonlethal increase in salinity on tidal freshwater and oligohaline marsh vegetation have not been tested in the field.

Even less well understood are the combined effects of increased plant submergence and saltwater intrusion. Increased soil waterlogging, a direct effect of plant submergence, decreases the diffusion rate of oxygen through waterlogged sediments (Greenwood, 1961) and causes more reduced soil conditions (Gambrell and Patrick, 1978). Strongly reduced soils coupled with high, potentially toxic, sulfide concentrations, resulting from the biochemical reduction of seawater-derived sulfate (Riley and Skirrow 1975; Gambrell and Patrick, 1978), can reduce the growth of

freshwater marsh plant species, such as *P. hemitomon* (Koch and Mendelssohn, 1989). Thus, the interaction between increased plant submergence and saltwater intrusion can directly effect the growth and vigor of tidal freshwater marsh vegetation.

In order to determine the effects of plant submergence, salinity, and their interaction on vegetation dieback in an oligohaline marsh in coastal Louisiana, a manipulative field experiment was conducted where oligohaline marsh vegetation growing within a low-salinity environment was exposed to increased submergence, increased salinity and their combination.

Methods

Study Area

This study was conducted along a salinity gradient in an oligohaline marsh located along Grand Bayou Canal (29° 30' N, 90° 27' W), ≈ 4 km west of LaRose, LA. The lower salinity component of the ecocline was dominated by *S. lancifolia* and had salinities ranging between 0.5 and 1.5 ppt during the 2 yr prior to this study (E. C. Webb, unpublished data). The higher salinity component of the gradient was dominated by *Spartina patens* (Aiton) Muhl. and had salinities ranging between 1.8 and 5.8 ppt during the 2 yr previous to this study (E. C. Webb, unpublished data). The lower salinity component of the ecocline was selected as the donor marsh site and the higher salinity component was selected as the recipient marsh site (see Experimental Design). The recipient marsh site was located approximately 0.5 km from the donor marsh site and both sites were located adjacent to the same tranasse (a narrow ditch created by trappers to traverse the marsh). Water levels were assumed to be constant over the short distance between sites. The soils of both the donor and recipient marsh are highly organic and are classified as Allemands muck (USDA, 1984).

Experimental Design

Sections of marsh, dominated by *S. lancifolia*, were exhumed from the lower salinity donor marsh, surrounded with landscape cloth to maintain the soil structure, and placed in one of four locations depending on the treatment: (1) increased submergence - sods were lowered 15 cm below the marsh surface within the donor marsh, (2) increased salinity - sods were moved to the higher salinity marsh and adjacent dieback pond, (3) increased salinity and submergence - sods were moved to the higher salinity marsh and dieback pond but placed at an elevation equivalent to 15 cm below the marsh surface, and (4) control - sods were exhumed and replaced at the ambient elevation of the donor marsh. There were five replicates for each treatment.

The depth to which the sods were placed (-15 cm) in the increased submergence treatments was equivalent to the elevation of the soil surface of nearby dieback ponds. The experimental treatments (increased salinity and increased salinity and submergence) were duplicated at the recipient marsh and at the pond location. Transplantation into both the recipient marsh and pond sites was done in order to guard against possible failure of the experiment due to uncontrollable, environmental circumstances such as shading from surrounding vegetation or herbivory of the experimental units. Since nothing extraordinary occurred and all treatments were successful, we report the data from both recipient locations.

In order to equalize the initial biomass and species composition within the marsh sections of all treatments, each section of marsh was of the same size (diameter = 30 cm, height = 25 cm) and was selected so that each marsh section contained three swards of *S. lancifolia*. In addition to *S. lancifolia*, the marsh sections contained other species commonly found within the oligohaline donor marsh site. These included *Sacciolepis striata* (L.) Nash., *Alternanthera philoxeroides* (Martius) Grisebach, *Lippia nodiflora*

(L.) Michaux. and *Eleocharis fallax*. Weatherby. The experiment was initiated on 30 April 1993 and was terminated at the end of the growing season on 1 October 1993.

Variables Sampled

Prior to plant harvest, redox potential was determined in each sod at two depths: 2 cm and 15 cm. Redox potential was measured using two brightened platinum electrodes and a calomel reference electrode at each depth. The electrodes were allowed to stabilize for 30 min prior to measurements. The potential of the calomel reference electrode against a standard hydrogen electrode (+244 mV) was added to the measured potential in order to determine Eh.

Aboveground vegetation was harvested at the soil surface from each treatment. The plants were brought to the laboratory, sorted by species into their live and dead components, dried at 65° C for 48 h, and weighed to the nearest 0.1 g for biomass determination.

After the vegetation was harvested, soil cores, 5 cm in diameter and 15 cm long, were extracted from the root zone of each sod using an aluminum coring device. The soil cores were immediately placed in 500 mL polypropylene centrifuge tubes. The centrifuge tubes were sealed and purged with nitrogen gas for 2 min in order to maintain an anaerobic environment, placed on ice, and brought to the laboratory for analysis. The soil samples were centrifuged at 10,000 g at 4° C for 15 min in a refrigerated centrifuge. An aliquot of the interstitial water was immediately placed in an antioxidant buffer and analyzed for hydrogen sulfide (Sulfide electrode, Lazer Research Laboratory, Los Angeles, CA). The remaining interstitial water was divided and separate aliquots were analyzed for pH (Altex Model 3560 digital pH meter) and salinity (Fisher Model 152 salinity/conductivity meter), filtered through a 0.45 µm Uniflo syringe tip filter (Schleicher and Schull, Keene, NH), preserved with nitric acid (US EPA, 1974) and analyzed for P, Mg, K, Ca, and Na concentrations with a Fisher inductively coupled

argon plasma emission spectrometer (Atom Comp Series 800), or filtered through a 0.45 μm filter (same as above), preserved with sulfuric acid (US EPA, 1974) and analyzed for $\text{NH}_4\text{-N}$ concentration using the Technicon Industrial Method at the Institute of Ecology, Soil Biology Laboratory, The University of Georgia.

Statistical Analysis

Analysis of variance was used to analyze the data (GLM Procedure, SAS Institute, Cary, N. C., 1985). Six treatments (two marsh elevations at three marsh locations) were analyzed as a modified block design ($n=5$) with treatments blocked on the distance from a tranasse (a narrow ditch created by trappers to traverse the marsh) that bisects the study area. All variables were tested for normality and homogeneity of variance in order to meet the assumptions of analysis of variance. A square root transformation was used to transform the variables live biomass, interstitial salinity and K. A log transformation was used to transform interstitial $\text{NH}_4\text{-N}$. Other variables did not require transformations to meet ANOVA assumptions. Untransformed means are presented.

Results

Sagittaria lancifolia was the dominant plant species in the treatment sods at all three marsh locations making up 70% of the aboveground live biomass at the donor marsh, 67% of the aboveground live biomass at the recipient marsh and 68% of the aboveground live biomass at the recipient pond (data not shown). Species composition was consistent across all treatments. Aboveground live biomass (Figure 5.1) was highest at the pond location and lowest at the donor marsh location (main effect of marsh location significant [$p < 0.01$]). Also, live biomass was significantly reduced at all three locations when the marsh surface was lowered 15 cm (main effect of marsh elevation significant [$p < 0.01$]). The effect of marsh elevation was greatest at the recipient sites compared to the donor site (significant marsh location x elevation interaction [$p < 0.05$]).

Interstitial salinity and the major cations of seawater (Table 5.1) were significantly higher in the recipient sites than at the donor marsh site ($p < 0.01$) but did not differ significantly ($p = 0.69$) with elevation. Interstitial pH (Table 5.1) also varied significantly ($p < 0.01$) with marsh location and was highest at the pond location and lowest at the donor marsh location, however, the range in mean pH was small. Elevation did not have a significant effect on interstitial pH ($p = 0.07$).

At the 2 cm depth (Figure 5.2a), soils were the most reduced at the recipient marsh location and least reduced at the recipient pond location [main effect of marsh location significant ($p < 0.01$)]. The soil redox potential at the donor marsh was intermediate between the two recipient sites. No elevational effect for surface redox potential was found. Redox potential at 15 cm depth (Figure 5.2b) was higher at the recipient pond site than at the recipient marsh site (main effect of marsh location significant [$p < 0.01$]). Soils were more reduced at the lower elevation at all three marsh locations (main effect of elevation significant [$p < 0.01$]).

Overall, interstitial sulfide concentrations (Figure 5.3) were higher in the marsh sections transplanted to the recipient marsh and pond locations [main effect of marsh location significant [$p < 0.01$]]. The highest interstitial sulfide concentrations occurred within the marsh sections at the -15 cm elevation within the recipient sites [significant marsh location x elevation interaction [$p < 0.01$]]. Interstitial sulfide concentrations did not differ between the two elevational treatments at the donor marsh location.

Interstitial $\text{NH}_4\text{-N}$ (Figure 5.4a) and P (Figure 5.4b) concentrations were higher at the recipient sites than at the donor marsh [main effect of marsh location significant [$p < 0.01$]]. Interstitial $\text{NH}_4\text{-N}$ and P were highest when the marsh elevation was decreased 15 cm within the two recipient sites [significant marsh location x elevation interaction [$p < 0.05$]]

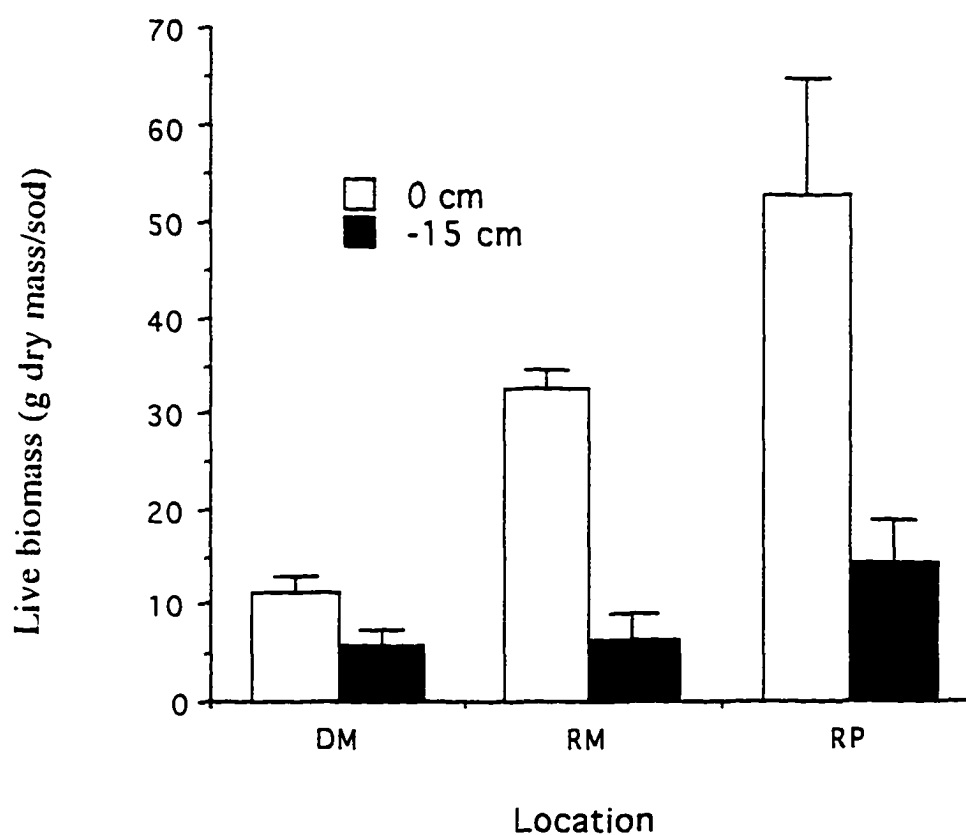


Figure 5.1. Live biomass by elevation for each marsh location. DM=Donor marsh location, RM=Recipient marsh location and RP=Recipient pond location. Bars represent mean \pm 1 standard error.

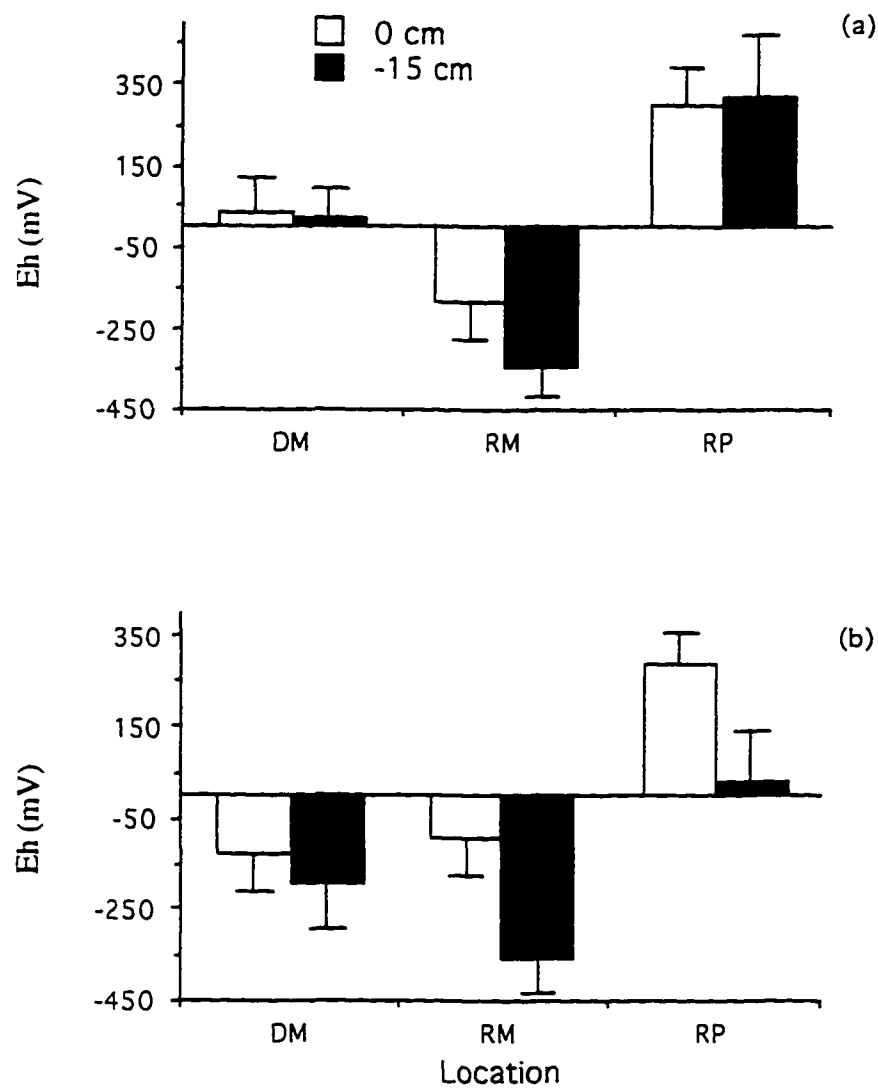


Figure 5.2. Redox potential (mV) at 2 cm depth (a) and at 15 cm depth (b) by elevation type for each marsh location. DM=Donor marsh location, RM=Recipient marsh location and RP=Recipient pond location. Bars represent mean \pm 1 standard error.

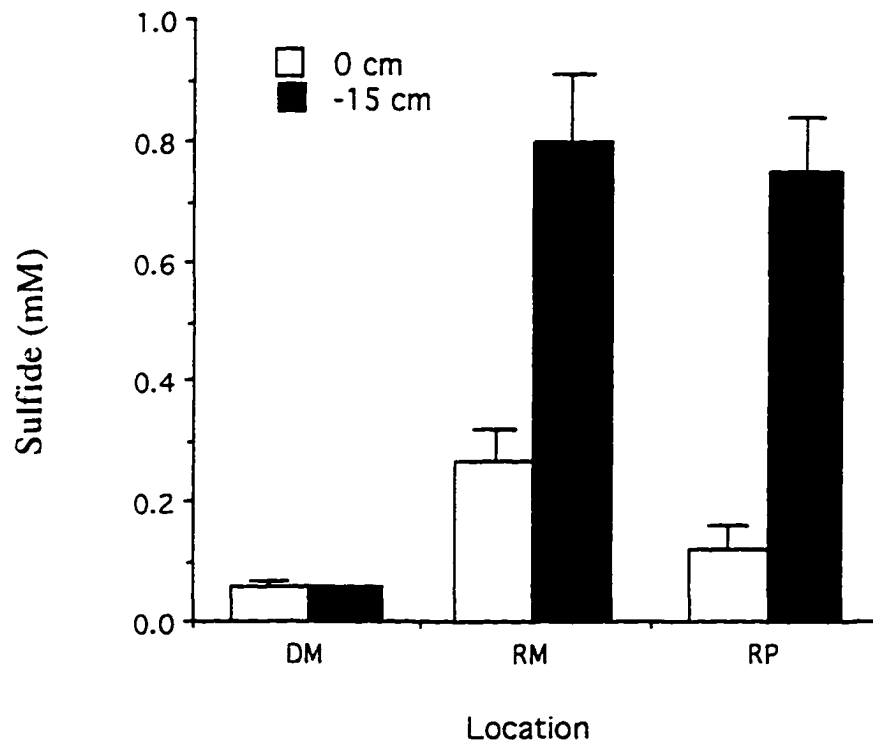


Figure 5.3. Interstitial sulfide concentration (mM) by elevation type for each marsh location. DM=Donor marsh location, RM=Recipient marsh location and RP=Recipient pond location. Bars represent mean \pm 1 standard error.

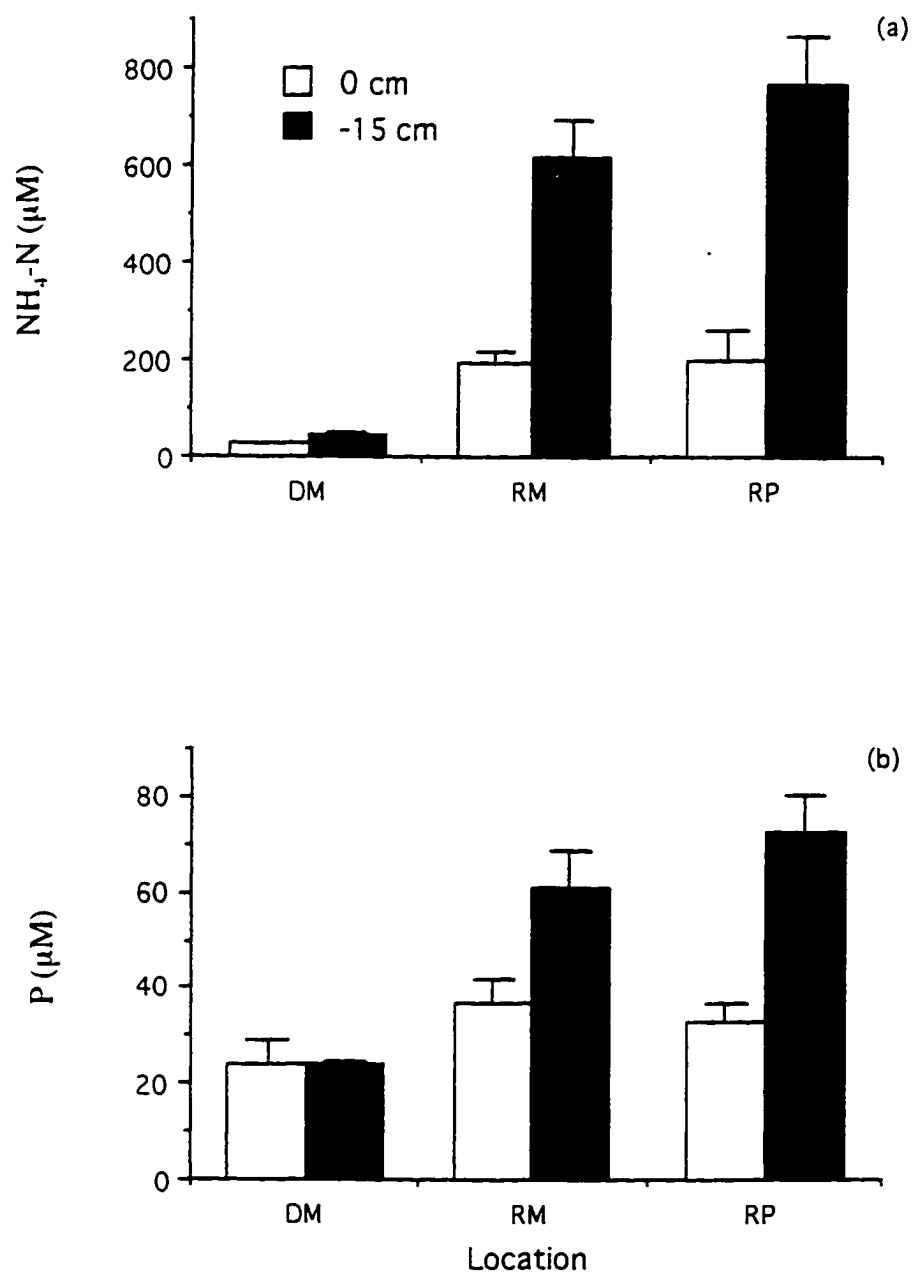


Figure 5.4. Interstitial $\text{NH}_4\text{-N}$ (a) and P (b) concentration (μM) by elevation type for each marsh location. DM=Donor marsh location, RM=Recipient marsh location and RP=Recipient pond location. Bars represent mean ± 1 standard error.

Table 5.1. Interstitial, pH, salinity (ppt) and interstitial soil cation concentration (mM). Mean values given for each marsh location (DM=donor marsh, RM=recipient marsh, RP=recipient pond); parentheses indicate 1 standard error. Different letters following values indicate a significant difference at the 0.05 probability level.

Variable	DM		RM		RP	
pH	6.7 a	(0.1)	6.7 a	(0.1)	7.2 b	(0.1)
Salinity	2.3 a	(0.1)	4.6 b	(0.1)	4.4 b	(0.1)
Na	35.6 a	(0.9)	64.0 b	(1.0)	59.9 c	(1.1)
K	1.8 a	(0.2)	4.0 b	(0.2)	4.0 b	(0.1)
Ca	1.4 a	(0.0)	1.9 b	(0.0)	2.1 c	(0.0)
Mg	3.7 a	(0.1)	7.2 b	(0.1)	7.0 b	(0.1)

Discussion

The increase in submergence experienced by the 15 cm decrease in elevation, which was equivalent to the elevation of the pond bottom, caused decreased plant growth of *S. lancifolia*, a dominant, oligohaline marsh species. Increasing the salinity exposure of oligohaline marsh vegetation, either by transplanting them into a higher salinity marsh or pond, resulted in greater biomass. Though salinity and the major cations of seawater were significantly higher at the recipient marsh and pond locations than at the donor marsh, salinities of 4-5 ppt were not detrimental to the *S. lancifolia* dominated marsh vegetation.

Natural salinity increases that occur as oceanic influence becomes greater can cause changes in vegetation structure (Chabreck and Linscombe, 1982) as well as cause vegetation dieback. In this study, when oligohaline marsh vegetation experienced increased salinity, there was no decrease in biomass. McKee and Mendelssohn (1989) found tissue damage and decreased stem elongation in *S. lancifolia* at salinities of 4.8 ppt, in a 35-d greenhouse experiment. Also, Pezeshki et al. (1987) found that salinities

between 5-7 ppt caused decreased stomatal conductance and photosynthesis in *P. hemitomon* during a 7-d exposure. We did not find a decrease in biomass at similar salinities within this study. The short-term effects of increased salinity may cause initial tissue death in *S. lancifolia* and may be responsible for the negative effects of salinity on *S. lancifolia* found in other studies. However, the initial dieback of *S. lancifolia* due to salinity increases is often followed by plant regrowth with time (Rebecca Howard, National Biological Service, personal communication). This experiment examined the effects of salinity on a *S. lancifolia* dominated marsh community over 5 mo. Although no initial tissue dieback was observed, the vegetation in our treatments may have experienced some initial shock to the increase in salinity. By allowing the experiment to proceed for an entire growing season, the plants were able to recover from the initial shock of slightly elevated salinities and we were better able to determine the longer term effects of salinity increases on a *S. lancifolia* dominated marsh community.

Unlike the effect of increased salinity, increased plant submergence caused a decrease in plant growth in this *S. lancifolia* dominated community. Soil redox potential at -15 cm in the decreased elevation treatment at the donor marsh was lower than at the ambient elevation at the donor marsh, indicating that more reduced soil conditions can cause decreased plant growth. McKee and Mendelssohn (1989) also found that increased submergence under field conditions reduced the biomass accumulation of a freshwater marsh community dominated by *S. lancifolia* and *P. hemitomon*.

The growth of this oligohaline marsh vegetation was most reduced when exposed to the combination of saltwater intrusion and increased plant submergence. An edaphic characteristic that is influenced by both increases in seawater and more reduced soil conditions is interstitial sulfide. As seawater influence increases, an increase in available sulfate also occurs (Riley and Skirrow, 1975). When soils become more reduced and sulfate is available, sulfate can be reduced to sulfide (Gambrell and Patrick, 1978).

High sulfide concentrations have been shown to reduce wetland plant growth (Koch and Mendelssohn, 1989; Havill, Ingold, and Pearson, 1985; King et al., 1982). The elevated sulfide concentration at the -15 cm elevation at the recipient sites is a result of the interaction of increased sulfate availability due to saltwater intrusion and more reduced soil conditions. Interstitial sulfide concentrations of 1.0 mM and higher caused decreased growth of the freshwater marsh grass *P. hemitomon* (Koch and Mendelssohn, 1989; Koch, Mendelssohn, and McKee, 1990) and increased sulfide concentrations, even as low as 1.0 mM, are responsible for decreased growth of the salt marsh grass *Spartina alterniflora* (Koch, Mendelssohn, and McKee, 1990; Mendelssohn and McKee, 1988; Havill, Ingold, and Pearson, 1985; King et al., 1982). In this study, a sulfide concentration of 0.78 mM, which occurred at the lower elevation recipient sites, caused a reduction in vegetative growth of the oligohaline marsh vegetation dominated by *S. lancifolia*.

Reduced soil conditions and elevated sulfide concentrations inhibit $\text{NH}_4\text{-N}$ uptake in some wetland plants (Koch, Mendelssohn, and McKee, 1990; Bradley and Morris, 1990). It is likely that the uptake of $\text{NH}_4\text{-N}$ by the vegetation in these treatments was inhibited by elevated sulfide concentrations because interstitial $\text{NH}_4\text{-N}$ concentration increased at the -15 cm elevation of the recipient sites. The greatest reduction in live biomass of the *S. lancifolia*-dominated, oligohaline marsh vegetation occurred when the marsh sections were exposed to the combination of increased submergence and saltwater intrusion. This treatment yielded more reduced soil conditions and higher sulfide concentrations, which likely caused an inhibition of the uptake of $\text{NH}_4\text{-N}$ by the vegetation. Collectively, these factors were responsible for the greatest decrease in plant growth of this *S. lancifolia*-dominated, oligohaline marsh vegetation.

Changes in soil reduction can indirectly influence the availability of P. Phosphorus is released from insoluble complexes as soils become more reduced, such as when ferric

iron reduction begins (Gambrell and Patrick, 1978). In the more reduced soil conditions of the marsh sections at the -15 cm elevation within the two recipient sites, interstitial P concentration was higher. The increase in interstitial P concentration within these treatments is directly related to the more reduced soil conditions.

It is unlikely that differences in P concentration affected plant growth since the treatments with the highest interstitial P concentration had the poorest vegetative growth. This indicates that P did not limit plant growth.

In summary, increased interstitial salinity, at least to 4.6 ppt, was not responsible for the dieback of *S. lancifolia* dominated, oligohaline marsh vegetation. Instead, increased plant submergence, especially in combination with increased saltwater intrusion, caused the greatest reduction in growth of this oligohaline marsh community. The combination of plant submergence and increased saltwater intrusion resulted in high interstitial sulfide concentrations and a likely inhibition of $\text{NH}_4\text{-N}$ uptake by the vegetation. The interaction between saltwater intrusion and soil submergence may be the most important cause of vegetation dieback in low-salinity marshes of coastal Louisiana, which are experiencing one of the highest rates of relative sea level rise in the world (Penland and Ramsey, 1990).

This study illustrates that dieback of *S. lancifolia*-dominated, oligohaline marsh vegetation can be alleviated by decreasing plant submergence, even at somewhat elevated salinities. In addition to preventing further vegetation dieback by increasing marsh elevation, marsh areas that have already experienced high rates of vegetation dieback can likely be revegetated. When marsh vegetation was transplanted into the dieback pond at 15 cm above the ambient pond elevation and within the brackish marsh at the ambient marsh elevation, plant growth was as high or higher than at the control site. This indicates that with the proper increase in the surface elevation of dieback zones, possible via sediment addition, marsh vegetation can be reestablished, even

within areas experiencing the effects of saltwater intrusion, as long as the salinity level does not surpass the tolerance limits of the vegetation.

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CHAPTER 6

CONCLUSION

Vegetation dieback in salt marshes and low-salinity marshes in the Mississippi River Deltaic Plain is likely caused by increased plant submergence due to high rates of relative sea level rise. At both the salt marsh site and the oligohaline marsh site, increased plant submergence resulted in more reduced soil conditions. Many studies have shown the deleterious effects that more reduced soil conditions have on plant growth (Mendelssohn and Seneca, 1980; Howes et al., 1981; Mendelssohn and McKee, 1988) and in this study, the growth of marsh vegetation decreased when soils were more reduced.

Saltwater intrusion is often cited as an important cause of wetland loss in coastal Louisiana, but little empirical data is available to support this. In the 34 years between 1956-1990, between 70 and 80 percent of the wetland loss in the Terrebonne basin cannot be attributed to saltwater intrusion (van Heerden et al., 1993). At Bayou Chitigue, the ambient salinities were not high enough to affect the health of salt marsh vegetation. Saltwater intrusion was at least partially responsible for a change in community structure in the low-salinity marsh dominated by *S. lancifolia*, but when tested experimentally, saltwater intrusion did not cause dieback to *S. lancifolia* dominated marsh vegetation.

Although saltwater intrusion did not directly cause vegetation dieback, saltwater intrusion combined with increased submergence was found to be the most damaging to oligohaline marsh vegetation. The combination of more reduced soil conditions and saltwater intrusion (with accompanying increased sulfate) drives the formation of sulfides (Riley and Skirrow, 1975; Gambrell and Patrick, 1978) which have been shown to be detrimental to many wetland plant species including *S. alterniflora* and *S. lancifolia* (King et al., 1982; Havill et al., 1985; Pezeski et al., 1987; Koch and Mendelssohn, 1989). In this study, increased sulfides were detrimental to both salt

marsh vegetation and low-salinity marsh vegetation. In addition to the toxic effect of sulfides on wetland vegetation, increased sulfides and more reduced soil conditions likely inhibited the uptake of $\text{NH}_4\text{-N}$ by vegetation in both the salt marsh and the oligohaline marsh. High sulfides and greatly reduced soil conditions caused by increased marsh flooding are the likely causes for vegetation dieback in many of Louisiana's coastal marshes.

The data from this study supports the model of soil formation processes during submergence presented by Nyman et al. (1993). We have found that increased plant submergence and saltwater intrusion leads to more reduced soil conditions and the formation of toxic sulfides which causes decreased plant growth. As plant growth decreases, there is a decrease in organic matter contribution to soil formation.

Organic matter constitutes a large portion of the total volume of Louisiana coastal marsh soils (Nyman et al., 1990), especially in low-salinity marshes. A decrease in organic matter production due to plant stress can negatively effect vertical marsh accretion. As marsh accretion decreases, plant submergence increases and vegetation becomes further stressed. This relationship between organic matter production and soil formation forms a positive feedback loop (Nyman et al., 1993) that continues until vegetation dieback occurs.

This positive feedback loop may be most pronounced in low-salinity marshes of coastal Louisiana. Organic matter makes up a greater volume of the soil of low-salinity marshes (Nyman et al., 1990) therefore organic matter contribution to soil formation is more important in low-salinity marshes than in saline marshes. Additionally, low-salinity marshes have some seawater influence which includes the addition of significant amounts of sulfate to oligohaline marshes (Riley and Skirrow, 1975). As plant submergence increases in low-salinity marshes, sulfide concentrations increase which is detrimental to the health of oligohaline marsh vegetation (Chapter 4, Chapter 5). Since

plant production is so important to soil formation in low-salinity marshes, soil formation is likely effected by stresses associated with plant submergence.

In contrast to low-salinity marshes, soil formation in freshwater marshes and salt marshes may be controlled by different mechanisms. Sulfide production is very low in tidal freshwater marshes even when marshes are severely flooded due to the lack of available sulfate (Riley and Skirrow, 1975). Since little sulfide is produced in freshwater marshes, increased plant submergence may not effect vegetative health as dramatically as low-salinity marshes (Chapter 5) allowing adequate organic matter contribution to soils. Mineral matter makes up a greater volume of salt marsh soils, therefore organic matter production is not as important to soil formation in salt marshes. These differences in soil formation and edaphic changes associated with increased marsh flooding may explain the rapid and widespread vegetation dieback in low-salinity marshes of coastal Louisiana (Leibowitz and Hill, 1987).

Many management projects in place or scheduled for implementation in coastal Louisiana are based on the management of water levels and salinity. This study indicates that wetland vegetation in salt marshes and low-salinity marshes dominated by *S. lancifolia* may be more susceptible to increased flooding than to increased salinities. The experiments conducted in this study show that by increasing marsh elevation and consequently removing stresses associated with increased marsh flooding, vegetation health can be improved. Additionally, slightly elevated salinities may not be detrimental to many coastal marsh plant species if marsh elevation is adequate. Future management and restoration projects should place an increased emphasis on reducing plant submergence by increasing marsh elevation in order to maintain healthy coastal marshes.

Further studies need to be conducted to determine the effects of salinity, submergence and the combination of saltwater intrusion and submergence on other dominant wetland plant species of coastal Louisiana. Additionally, increased salinities

have caused wetland changes (Chabreck and Linscombe, 1982) but salinity changes and its effect on vegetation dieback and wetland loss is not well understood. More data is necessary on coastwide salinity changes and the direct effects of increased salinities on coastal wetlands.

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APPENDIXES
LETTERS OF PERMISSION



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Eric was born in Columbus, Ohio, on March 21, 1966. He graduated from Gahanna Lincoln High School, Gahanna, Ohio, in 1984. He then attended Ohio Dominican College, Columbus, Ohio, where he received a bachelor of science degree in Biology in 1988. After working in a clinical immunology laboratory for a year, he entered graduate school and received a master of science degree in Biology with an emphasis in wetland ecology and aquatic ecology from Morehead State University in Morehead, Kentucky in 1991. Eric entered graduate school in the Department of Oceanography and Coastal Sciences at Louisiana State University in 1991 and worked as a Research Assistant and a Research Associate at the Wetland Biogeochemistry Institute at Louisiana State University. Eric is currently employed at the Louisiana Department of Natural Resources/Coastal Restoration Division as a Natural Resources Geoscience Supervisor. Eric will graduate with a doctor of philosophy degree in Oceanography and Coastal Sciences from Louisiana State University in 1997.


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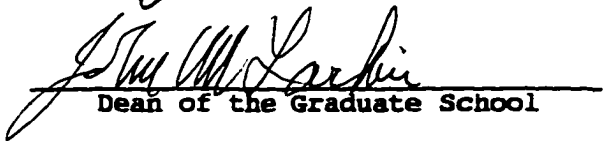
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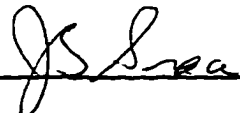
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