THE IMPACT OF THE CAERNARVON DIVERSION ON ABOVE- AND BELOWGROUND MARSH BIOMASS IN THE BRETON SOUND ESTUARY AFTER HURRICANE KATRINA

A Thesis

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ABSTRACT

The purpose of this project was to monitor the recovery of marsh vegetation following Hurricane Katrina in the Breton Sound estuary, which is directly influenced by the Caernarvon freshwater diversion. From March 2006 until October 2007, we measured above- and belowground biomass, soil porewater nutrients (NO_x, NH_4, and PO_4), porewater salinity, porewater sulfide concentrations, and soil redox potential bimonthly at duplicate Near (N), Intermediate (M), and Far (F) sites (with respect to the diversion), as well as at a Reference (C) site. End of season live (EOSL) below and above ground biomass was measured at 26 separate stations where vegetation studies had been done previously. Peak standing crop ranged from 423 $g/m^2/yr$ at site M2 to 1515 at F1. Aboveground biomass was significantly greater at N1 than at N2, M1, or M2. Peak belowground biomass ranged from 8315 $g/m^2/yr$ at C2 to 17890 $g/m^2/yr$ at N1. Multiple regression analysis showed that sulfide levels were related to belowground biomass ($p < 0.05$; $R^2=0.13$). PO_4 concentration was related to aboveground biomass ($p<0.05$; $R^2=0.09$). Short term vertical marsh accretion measurements ranged from 0.49 cm y^{-1} at N2 to 1.25 cm y^{-1} at N1, suggesting that all the study sites are stable or accreting at a rate which offsets RSLR. Analysis of the results of short term accretion compared to regional relative sea level rise (RSLR) indicates that all Caernarvon sites are keeping pace with RSLR. Analysis of porewater parameters showed that stress on plants was low to moderate while nutrient concentrations were greatest at the Near sites.

1. INTRODUCTION

1.1 Background

The Mississippi river delta plain formed as a series of delta complexes during the last 6-7 thousand years (Roberts 1997). Deltaic complexes are made up of one or more delta lobes which are the result of river switching and associated sediment deposition events. Delta lobes consist of subdeltas and crevasse splays resulting from overbank flooding events from the main river channel and numerous distributaries. The amount of time and spatial extent to which flood events occurred are the defining characteristics of delta lobes and crevasse splays. Delta lobes encompass hundreds of square kilometers and are active on a time scale from hundreds to over a thousand years. Crevasse splays cover only a few square kilometers and are active for only a few decades (Roberts 1997).

Switching of delta lobes, overbank flooding, and crevasses of the Mississippi River were major geological processes in the formation and maintenance of the Mississippi delta (Hatton et al. 1983; Kesel 1988; Kesel 1989; Roberts 1997; Davis 2000; Day et al. 2007, Tornqvist et al. 2008). Since the early 1900's, however, most of coastal Louisiana has been hydrologically isolated from the Mississippi River by the flood control levees that have almost completely halted input of river water to the delta plain (Kesel 1988; Kesel 1989; Mossa 1996, Day et al. 2007). In addition, canal dredging and spoil bank construction associated mainly with the oil and gas industry have further altered the natural hydrology of the delta, promoting saltwater intrusion and limiting hydrological exchange (Bass and Turner 1977; Swenson and Turner 1987, Day et al. 2000). These modifications to the coastal landscape have led to a massive loss of wetlands during the 20th century, with about 4500 km^2 lost during the 20th century (Salinas et al. 1986; Boesch et al. 1994; LDNR 1998; Day et al. 2000, 2007, Boesch 2006a). Relative sea-level rise (RSLR), the combined effect of eustatic sea-level rise (1-2 mm $yr^{-1}$for the 20th century;

Gornitz et al. 1982) and coastal subsidence, is in excess of 10 mm yr^{-1} in much of the Mississippi delta (Penland and Ramsey 1990). It is predicted that eustatic sea level will increase in the 21st century by 40 cm to over one meter (IPCC 2001; Rahmstorf 2007). In order to maintain a viable and healthy marsh, rates of vertical marsh accretion must keep up with this rise in sea level.

There is a strong consensus in the scientific and management community that the long-term survival of Louisiana's coastal wetlands depends, as one of the major restoration strategies, on the reintroduction of river flow into the coastal wetlands to stem saltwater intrusion and supply nutrients and sediments for wetland restoration (Templet and Meyer-Arendt 1988; Kesel 1989; Boesch 1996; Day et al. 1997, 2000, 2007; Gosselink 2001, Boesch et al. 2006b). The State of Louisiana and the Federal Government have developed a plan for freshwater diversions that will mimic flooding events of the Mississippi River (Chatry and Chew 1985; LDNR 1998, USCOE 2004, LCPRA report 2007). This study focuses on the second largest diversion structure currently in operation in Louisiana, located southeast of New Orleans at Caernarvon.

On August 29, Hurricane Katrina passed over southeastern Louisiana with winds in excess of 250 kph (155 mph), and a tidal surge southeast of New Orleans of nearly six meters. Levees in the eastern part of the city were overtopped and levees fronting Lake Pontchartrain and along the Mississippi River Gulf Outlet failed (Travis 2005, Boesch et al. 2006, Day et al. 2007). Before impacting New Orleans, the hurricane crossed the lower Mississippi River near its mouth and traversed the Breton Sound estuary. Strong easterly winds north of the eye pushed a six-meter storm surge over the marshes, causing massive disturbance to the extensive wetlands in the estuary (Boesch et al. 2006; Day et al. 2007).

1.2 Purpose of Study

The purpose of the present project was to monitor the recovery of marsh vegetation in the Breton Sound estuary, which is directly influenced by the Caernarvon freshwater diversion. Specifically, we measured above- and belowground biomass, and porewater chemistry bimonthly at Near, Intermediate, and Far sites (with respect to the diversion), as well as at a reference site. Data collection began March 2006 and was completed October of 2007. Study sites were chosen from a larger set of 30 stations where end of season live (EOSL) biomass had been previously measured. In order to assess marsh productivity, we measured a number of environmental parameters at each study site that effect marsh growth including water level, soil redox potential, soil bulk density, accretion, and porewater salinity, sulfide and nutrient (NO_x, NH_4, and PO_4) concentrations.

1.3 Site Description

The Caernarvon diversion is located at river mile 81.5 on the east bank of the Mississippi River near Caernarvon, Louisiana in St. Bernard Parish. The water control structure has been in operation since August 1991 and consists of a five box culvert with vertical lift gates allowing for the passage of river water with a maximum discharge rate of 226 m^3s^{-1}. However, discharge from August 1991 to January 2000 had an average of 21m^3s^{-1}, much less than the potential maximum. Discharge rates through the diversion were calculated from a rating curve developed by the Louisiana Department of Natural Resources (Lane et al. 2006). From the diversion structure, river water enters the Breton Sound estuary.

The Breton Sound estuary is part of the Plaquemines-St. Bernard delta complex, which was first formed several thousand years ago (Scruton 1960; Roberts 1997). Mississippi river water inputs into the estuary decreased drastically with the leveeing of the Mississippi River that began soon after the colonization of New Orleans by the French in 1719 (Welder 1959, Boesch

1996, Colten 2000). However, there were major freshwater inputs to the area via crevasses, minor distributary flow, and overbank flooding until the early part of the 20th century (Welder 1959). Some Mississippi River water enters Breton Sound south of where artificial levees end along the east bank of the river (Lane et al. 1999).

The Breton Sound estuary consists of about $1100 km^2$ of fresh, brackish, and saline wetlands, which are hydrologically bounded to the north by the natural levees of Bayou La Loutre, to the west by the Mississippi River levee, and to the east by the spoil banks of the Mississippi River Gulf Outlet (MRGO). The southern end of the estuary connects to the Gulf of Mexico. Prior to Hurricane Katrina, diverted river water traveled two major routes before entering the open water environment of Breton Sound and the Gulf of Mexico. After the storm, the western route was blocked, and it now appears that most water flows out to the Gulf of Mexico via Lake Leary and Bayou Terra aux Bouef. Some water also flows to the Gulf via Bayou Gentilly and Little Lake. There is also considerable over-marsh flow depending on structure discharge, winds, and Gulf water levels (Snedden et al. 2006).

2. MATERIALS AND METHODS

2.1 Study Site Locations

Measurements of above and belowground biomass were taken bimonthly at duplicate Near (3.4 km), Mid (12.3 km), and Far (18.1 km) sites (with respect to the diversion) in the Breton Sound estuary (Figure 1). A reference site, located outside of the Caernarvon discharge hydrologic basin, was also sampled bimonthly as part of this study. Sites were chosen based on previous data that showed the diversion's direct hydrologic influence on nutrient concentrations in the estuary extended to the Far site location (Lane et al 1999 and 2007).

Sampling was carried out from March 2006 and to October 2007, with a total of nine sampling trips to the eight intensive sites. Measurements were made approximately 50m from the bayou edge in relatively homogenous vegetation. The two Near sites (N1 & N2) were located south and southwest of Big Mar, the two Mid sites (M1 & M2) were located in the vicinity of Grand and Little Lakes, and the Far sites (F1 & F2) were located in the marshes near Four Horse Lake (Figure 1). The Reference sites (C1 & C2) were located outside the direct influence of the diversion near Reggio Canal. These study sites were selected to coincide with where previous EOSL measurements were taken (Lane, personal communication). These earlier EOSL studies showed that significant changes in biomass occurred with distance from the diversion structure. The degree of hurricane disturbance that occurred at each sampling site ranged from severe (N2) to moderate (F2).

2.2 Belowground Biomass

Belowground marsh biomass was measured at the same locations and times as aboveground biomass using methods outlined by Gross et al. (1991). Triplicate cores were collected at each study site with 7.5 cm x 30 cm thin-walled stainless steel tubes with sharpened ends. The cores

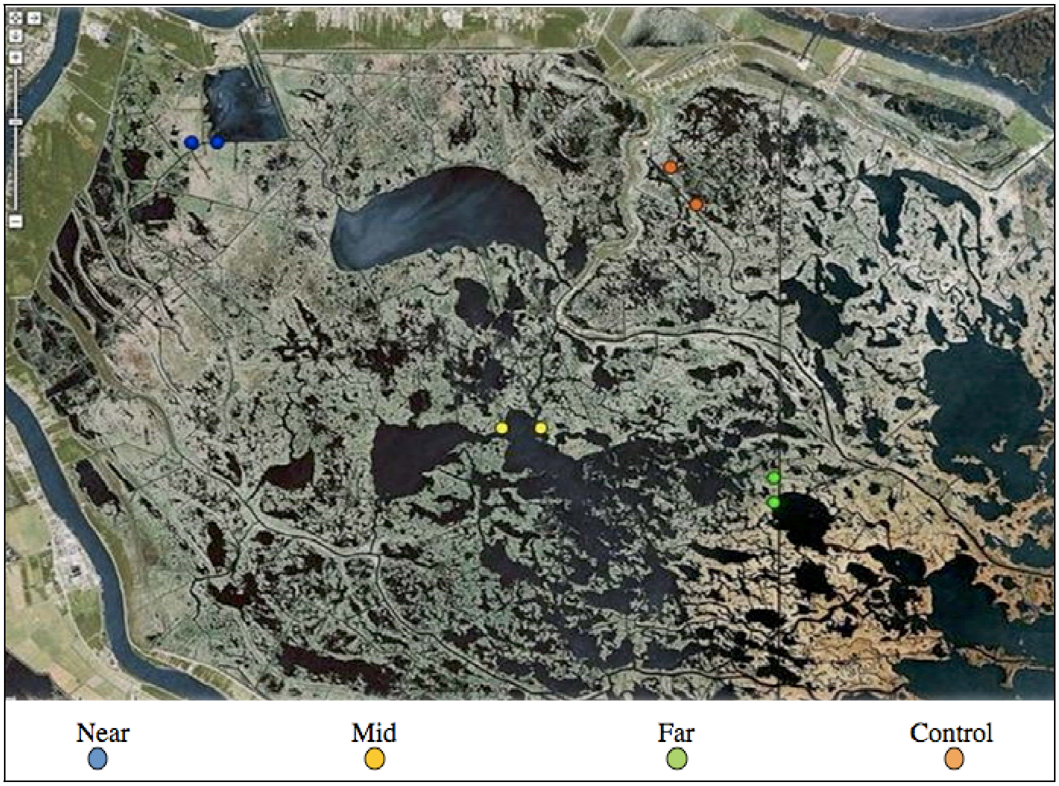

Figure 1. Location of duplicate, Intensive Sampling Sites sampled bi-monthly.

were taken immediately after aboveground biomass was collected, and labeled so that each core was paired with its respective aboveground biomass sample. Cores were cut in the laboratory into 5 cm sections to the bottom of the live root zone. Soil and detritus was removed from the core sections by washing the cores over a 2 mm-mesh sieve. Belowground materials retained by the mesh were separated into live and dead fractions and dried at room temperature. The sieved biomass was placed on a light table and the live biomass was sorted from the dead. Biomass was considered live if it was white and turgid (Fitter 2002; Darby 2008).

Three estimates of annual net belowground primary productivity (NPP) were determined at the eight intensive study sites for the period from July 2006 through June 2007. NAPP estimates were derived by subtracting the minimum biomass measurement from the maximum biomass measurement, the Milner-Hughes method (1968), and the peak standing crop method (Hopkinson et al. 1978).

2.3 Aboveground Biomass

Three 0.25 m^2 clip plots were taken bimonthly at each location using randomly placed quadrats. Vegetation within the quadrat was cut as close to the marsh surface as possible, stored in labeled paper bags, brought back to the laboratory, and refrigerated until processing. Live vegetation was separated from dead vegetation, dried at 60 degrees C, and weighed. Dead vegetation was discarded. Annual NPP for aboveground biomass at the eight intensive study sites was determined by subtracting the minimum biomass measurement from the maximum biomass measurement from the July 2006 through June 2007 sampling period. The Milner-Hughes method and peak standing crop method were also used to estimate annual aboveground productivity at the eight sampling sites. Species composition of vegetation at each of the sampling sites was assessed during fall 2006 using the Braun-Blanquet method (Poore 1955).

Live above and belowground biomass at the 26 EOSL sampling sites was sampled during October 2006. Soil samples from each EOSL belowground core were analyzed to determine soil bulk density.

2.4 Soil Redox Potential

Soil redox potential was measured at 15cm depth in each of the core samples using brightened platinum electrodes. The electrodes were placed in their respective cores and allowed to equilibrate in situ for at least one hour prior to measurement (Faulkner et al. 1989). Each electrode was checked for accuracy with quinhydrone in pH 4 and pH 7 buffers (which give 218 mV and 40.8 mV readings at 25˚C, respectively). Readings were taken with a portable pH/mV digital meter (model 5985-80; Cole-Parmer Instrument Co., Chicago, Illinois). The potential of a calomel reference electrode (+244 mV) was added to each value to calculate Eh.

2.5 Water Level

Daily water levels at the Control, Far, and Mid sites were recorded from April 12 to November 27, 2007. Water level data at the Near site was not recorded due to equipment malfunction. At one of each pair of study sites, a 4’x4” PVC piezometer was installed with a YSI-6000 sonde located at the bottom which recorded water level every 15 minutes. Data collected in the YSI data logger was downloaded onto a handheld data logger during each site visit.

2.6 Porewater Sampling

A narrow diameter plastic tube connected to a 50 ml syringe was used to extract interstitial water from the soil (McKee et al. 1988). The rigid plastic tube (3 mm diameter and sealed at the lower end) was inserted into the soil to a 15 cm depth. The length of the inserted section (beginning 5 cm below the soil surface) was perforated by five small holes. Suction was applied to the tube with a 50 ml-capacity syringe. A 3-way valve inserted between the syringe and collection tube allowed the expulsion of air and debris. This apparatus allowed for the collection of relatively clear interstitial water from the wetland substrate within a few seconds and without significant exposure to the atmosphere. The first 5 ml of each sample was discarded. Salinity was determined on an aliquot (5 ml) of clear interstitial water collected in a separate container using a YSI 85 salinity meter. Another aliquot (5 ml) of the interstitial water was transferred through the three-way valve to a graduated cylinder containing an equal volume of antioxidant buffer (Lazar operating instructions for Model IS-146 Sulfide Electrode). The antioxidant buffer prevented the oxidation of sulfide in the sample and converted H_2S and HS^- forms of sulfur to S^{-2}. The samples were returned to the laboratory and analyzed for total sulfide with a Lazar (Model IS-146, Lazar Research Laboratories, Los Angeles, CA) sulfide electrode. A standard curve was constructed with a series of solutions of Na_2S prepared with the antioxidant buffer. An additional 60-80 ml of interstitial water was collected at each site in a separate container for nutrient analysis, and immediately stored on ice. Within 24 hours, nutrient interstitial water samples were filtered through pre-rinsed 25 mm 0.7 um Whatman GF/F glass fiber filters into acid-washed bottles and frozen until nutrient analysis. Nitrate+nitrite (NO_x) was determined using the automated cadmium reduction method with an Alpkem© autoanalyzer (Greenberg et al. 1985). Ammonia+ammonium (NH_x) was determined by the automated phenate method, and phosphate (PO_4-P) by the automated ascorbic acid reduction method (Greenberg et al. 1985). Porewater nutrient concentrations are reported as the average of three replicate samples for each site during each sampling period.

2.7 Accretion

Three feldspar marker horizons were established at each of the eight intensive study sites during the first sampling trip in March 2006. Powdered feldspar clay was laid on the marsh surface 1 cm thick in three randomly placed $0.25m^2$ plots at each of the study sites. The thickness of the material deposited on top of the feldspar marker was measured destructively by taking three 20cm x 20cm plugs using a shovel, cleanly slicing the core into several sections to reveal the horizon, then measuring the thickness of the material above the horizon at three different locations (Lane et al. 2006). These measurements were averaged to provide an estimate of vertical accretion at each marker and averaged with the other site markers to arrive at an estimate of vertical accretion at each site. The rate of vertical accretion at each site was calculated by dividing the thickness of material above the surface of the horizon by the amount of time (21 months) the horizon had been in the sediment.

2.8 Statistical Analysis

Above- and belowground biomass, redox potential, porewater salinity, and porewater nutrient values at each site were averaged and the standard error was calculated by site for each sampling trip. A multiple regression analysis of belowground biomass and aboveground biomass was carried out to determine which of the edaphic factors measured in the study were most influencing growth. Above- and belowground biomass, porewater redox, porewater salinity, porewater nutrient results and short-term marsh accretion were analyzed using a one-way ANOVA statistical test to determine the variation among sites. A comparison of means for all pairs was carried out using the Tukey Kramer Highly Significant Difference (HSD) test. Soil bulk density at the 26 EOSL sites was compared with distance from the freshwater diversion using general linear regression statistics. Above- and belowground biomass was compared to soil bulk density at each EOSL site using linear regression statistics to determine if there was a relationship between biomass and soil bulk density. All analyses were computed using JMP statistical software produced by SAS Institute, Inc. (Sall et al. 2005).

3. RESULTS

3.1 Diversion Discharge

Mean monthly discharge from the Caernarvon freshwater diversion during the study period ranged from a high of 139 m^3/s in March 2007 to a low of 11 m^3/s in August 2006. Discharge increased from January 2006 (41 m^3/s) to April 2006 (103 m^3/s) before decreasing steadily to a low of 11 m^3/s in August 2006. Discharge increased again for three months until November 2006 (67 m^3/s), decreased to 27 m^3/s in December 2006, and increased again until discharge peaked in March 2007. Discharge decreased steadily until June 2007 (31 m^3/s) before increasing to 66 m^3/s in August 2007 and declining to 14 m^3/s in October 2007. The two greatest periods of discharge during the study period occurred between February and May during both years (Figure 2).

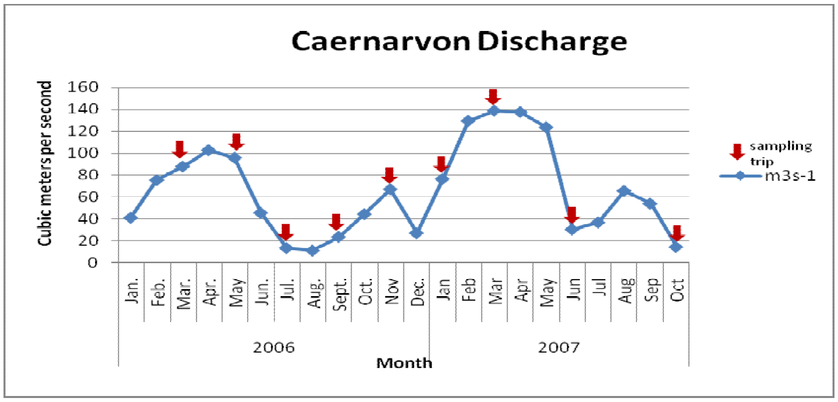

Figure 2. Mean monthly discharge from the Caernarvon diversion structure.

3.2 Belowground Biomass

Live belowground biomass from May 2006-June 2007 ranged from 17890 g/m^2 at N1 in April 2007 to 1920 g/m^2 at N2 in May 2006 (Figure 3). Peak belowground biomass occurred during April 2007 at C1 (9520 g/m^2), C2 (8315 g/m^2), F1 (8588 g/m^2), and F2 (15292 g/m^2) and at M1 (10200 g/m^2), M2 (11217 g/m^2), and N2 (8856 g/m^2) in June 2007. Belowground biomass increased at each study site from May 2006 until June 2007. The largest net increase (12825 g/m^2) occurred at site N1, from 3206 g/m^2 in May 2006 to 16031 g/m^2 in June 2007. The smallest net increase (2797 g/m^2) occurred at site F1, from 3386 g/m^2 in May 2006 to 6183 g/m^2 in June 2007 (Table 1). The results of the Tukey Kramer HSD test showed that belowground biomass at site N1 was significantly greater than all of the other sites with the exception of site F2. Results of the multiple regression analysis showed that sulfide concentration was the only factor significantly effecting belowground biomass (p-value <0.05, $R^2 = 0.13$).

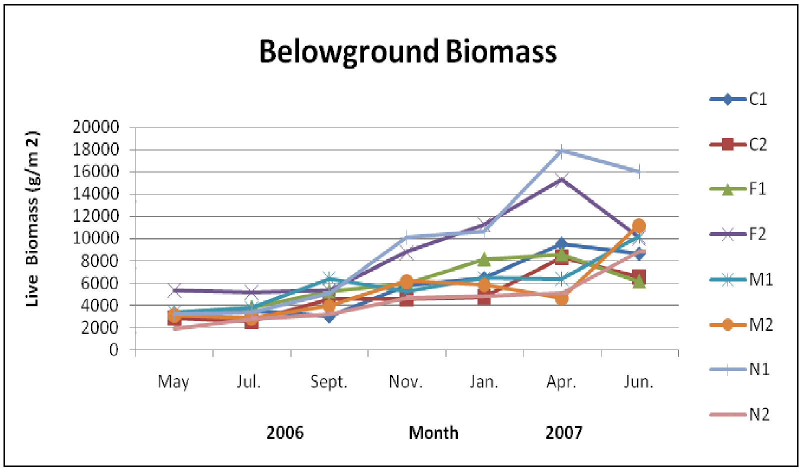

Figure 3. Mean belowground biomass at the eight intensive study sites during the May 2006 – June 2007 study period.

Table 1. Mean belowground biomass (g/m^2 ±standard error).

	C1	C2	F1	F2	M1	M2	N1	N2
May '06	3127±984	2819±382	3386±684	5368±1026	3366±768	3120±1157	3206±1633	1920±1633
July '06	3565±821	2539±132	3812±511	5161±617	3764±133	2854±190	3405 ±595	2828±595
Sept. '06	3014±291	4531±566	5283±661	5316±1195	6433±113	3961±493	5149 ±732	3231±732
Nov. '06	5772±1718	4569±549	5983±1226	8762±1275	5244±533	6186±659	10094±1718	4719±1718
Jan. '07	6455±532	4747±845	8165±2276	11253±791	6438±753	5869±662	10620 ±719	4849±719
Apr. '07	9520±2678	8315±1594	8588±1190	15292±1881	6409±959	4654±699	17890±1625	5093±1625
Jun. '07	8625±1570	6578±566	6183±1264	10065±1795	10200±231	11217±4318	16031±3957	8856±3957

Annual belowground NPP using the max-min. method ranged from a high of 14485 g/m^2/yr to a low of 4776 g/m^2/yr (Table 2). NPP estimates at the near site were 14485 g/m^2/yr at site N1 and 6028 g/m^2/yr at site N2. The mid sites had annual NPP of 7453 g/m^2/yr at site M1 and 7346 g/m^2/yr at site M2. At the far sites, annual NPP was 4476 g/m^2/yr and 10131 g/m^2/yr at sites F1 and F2, respectively. Annual NPP at the control sites were 6506 and 5776 g/m^2/yr at sites C1 and C2, respectively. The results of annual NPP using the Milner-Hughes and Peak Standing Crop methods are reported below (Table 2).

When belowground biomass was averaged between replicate sites to estimate the amount of biomass with depth, the Far and Near sites had greater mean belowground biomass than the

Table 2. Annual NPP of belowground biomass (g/m^2/yr) for July 2006 - June 2007.

Site	Max-Min.	Milner-Hughes	Peak Standing Crop
C1	6506	6506	9520
C2	5776	5776	8315
F1	4776	4776	8588
F2	10131	10131	15292
M1	4956	7654	10200
M2	7256	10212	11217
N1	14485	14485	17890
N2	6028	6028	8856

Control and Mid sites (Figure 4). The Far site (F) had the greatest mean belowground biomass at 7330 g/m^2, while mean belowground biomass at the Near site (N) was 7060 g/m^2. Mean belowground biomass at the Mid site (M) was 5693 g/m^2, and 5299 g/m^2 at the Control site (C). The highest mean belowground biomass in the soil column was measured at the Near site at a depth of 1-5cm (1740 g/m^2). The lowest mean belowground biomass value in the soil column (461 g/m^2) occurred at a depth of 26-30cm at the Control site. Live belowground biomass growth above 20cm depth accounted for 78% of total live belowground biomass at the Near sites, 74% at the Mid sites, 67% at the Far sites, and 76% at the Control sites (Figure 4).

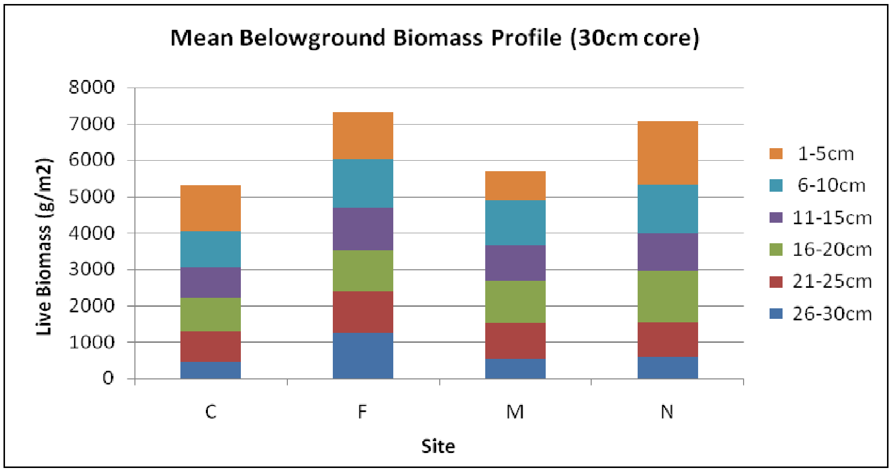

Figure 4. Mean belowground live biomass at 5cm depth intervals.

3.3 Aboveground Biomass

Live aboveground biomass exhibited typical values for salt marshes. Biomass at all sites generally tended to be highest during the growing season (May to November) and lowest in the winter. Overall, live aboveground biomass ranged from a high of 1581 g/m^2 at N1 in June 2007 to a low of 0 g/m^2 at N2 during January and April 2007 (Figure 5).

Live aboveground biomass at the near sites ranged from 1581 g/m^2 to 595 g/m^2 at N1 and 1208 g/m^2 to 0 g/m^2 at N2 (Table 3). Live aboveground biomass at M1 ranged from 1289 g/m^2 to 53 g/m^2 while M2 had a lower range of 440 g/m^2 to 48 g/m^2. Live aboveground biomass at the far sites ranged from 1515 g/m^2 to 392 g/m^2 at F1 and from 1086 g/m^2 to 408 g/m^2 at F2. Ranges at the control sites were from 1383 g/m^2 to 309 g/m^2 at C1 and 1397 g/m^2 to 239 g/m^2 at C2.

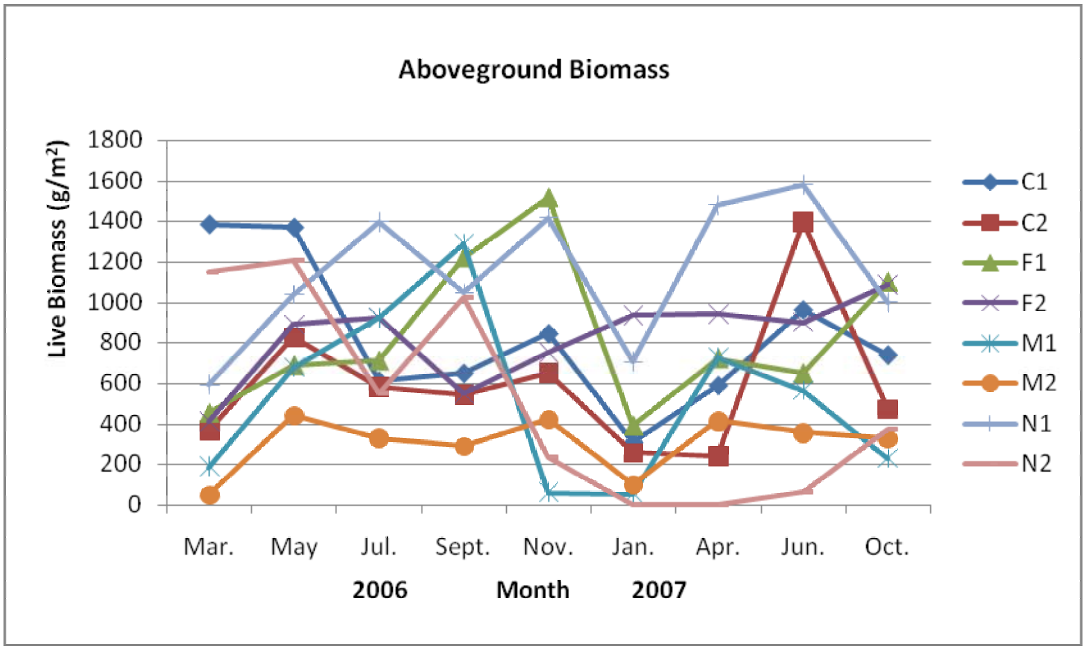

Figure 5. Mean aboveground biomass at the eight intensive study sites during the study period from March 2006 – Sept. 2007.

The results of the Tukey Kramer HSD test showed that aboveground biomass at N1 was significantly greater than at N2, C2, M1, and M2. Aboveground biomass at M2 was significantly less than at N1, C1, F1, and F2. When the duplicate sites were analyzed as single sites (i.e. Near, Mid, Far, and Control) the results of the Tukey Kramer HSD test showed that the Mid sites had significantly lower aboveground biomass than the three other study sites. Results of the multiple regression statistical analysis showed that phosphate concentration was the only significant factor (p-value <0.05, $R^2 = 0.09$) effecting aboveground biomass.

Table 3. Mean aboveground biomass (g/m^2±standard error).

	C1	C2	F1	F2	M1	M2	N1	N2
Mar.2006	1383±244	367±85	455±73	408±40	187±53	48±9	595±22	1149±32
May'06	1368±386	825±253	691±94	887±77	677±84	440±68	1040±219	1208±208
July '06	611±65	581±81	712±11	923±169	921±56	329±19	1397±322	551±81
Sept.'06	649±179	545±65	1221±97	547±264	1289±873	289±83	1048±221	1027±571
Nov.'06	844±64	652±150	1515±185	751±67	59±23	423±204	1419±255	237±88
Jan.2007	309±41	256±44	392±142	932±258	53±19	97±58	709±80	0
Apr. '07	589±49	239±35	723±160	939±277	727±76	413±42	1480±216	0
Jun. '07	961±52	1397±314	651±103	899±136	563±31	355±33	1581±236	63±7
Oct. '07	739±59	471±70	1100±323	1086±175	227±49	326±18	1001±84	374±74

Aboveground NPP from July 2006 – June 2007, using the max-min. method, ranged from a high of 1236 $g/m^2/yr$ at M1 to a low of 326 $g/m^2/yr$ at M2 (Table 4). Annual NPP at N1 was 872 $g/m^2/yr$ while NPP at N2 was slightly greater at 964 $g/m^2/yr$. The Mid sites had annual NPP of 1236 $g/m^2/yr$ at M1 and 326 $g/m^2/yr$ at M2. NPP estimates at the Mid sites were similar to NPP at the Far sites. Annual NPP at F1 was 1123 $g/m^2/yr$ while F2 had a much lower value of 392 $g/m^2/yr$. Control site estimates were 652 $g/m^2/yr$ at C1 and 1158 $g/m^2/yr$ at site C2. The result of annual NPP using the Milner-Hughes and Peak Standing Crop methods are reported below (Table 4).

Table 4. NAPP of Aboveground Biomass (g/m^2/yr) for July 2006 - June 2007.

Site	Max-Min.	Milner-Hughes	Peak Standing Crop
C1	652	885	961
C2	1158	1265	1397
F1	1123	1134	1515
F2	392	392	939
M1	1236	1042	1289
M2	326	450	423
N1	872	1243	1581
N2	964	539	1027

3.4 EOSL Results

Live aboveground biomass at the EOSL sampling sites ranged from 3620 g/m^2 at site L12 to 193.3 g/m^2 at site L10 (Table 6). Aboveground biomass at site L12 was more than two times higher than any recorded live aboveground biomass value at the eight, bi-monthly intensive study sites. Live belowground biomass at the 26 EOSL sampling sites ranged from 5444 to 1428 g/m^2. The results of linear regression statistics were not significant when examining EOSL above- and belowground biomass ($R^2 = 0.05$; $R^2 = 0.17$ respectively) with relation to distance from the diversion.

Bulk density measurements ranged from 0.07 g/cm^3 to 0.37 g/cm^3 (Table 6). Among the 26 end of season live (EOSL) sampling sites, the lowest recorded bulk density was measured at a distance of 25.5 kilometers from the diversion while the highest recorded bulk density was 4.9 km from the diversion. There was not a strong relationship found between EOSL bulk density and distance from the diversion (R^2=0.1561). Bulk density did not correlate strongly with aboveground biomass ($R^2 = 0.0034$) or belowground biomass production ($R^2 = 0.0024$) at the sampling sites.

3.5 Species Composition

Site N1 had the greatest variety of species. *Panicum virgatum, Polygonum sp.,*

Table 5. EOSL sampling site measurements including distance from the Caernarvon diversion, soil bulk density, and above- and belowground biomass.

Site	Distance (km)	Bulk Density (g/cm^3)	Aboveground Biomass (g/m^2)	Belowground Biomass (g/m^2)
L1	41.8	0.277	764.0	5444.0
L2	1.6	0.277	590.7	4620.1
L3	1.6	0.199	1,064.0	5189.6
L4	43.1	0.296	606.7	3642.6
L5	40.6	0.206	877.3	4412.7
L6	25.5	0.068	672.0	3471.6
L7	19.3	0.164	628.0	2368.7
L8	22.2	0.231	489.3	2676.0
L9	18.2	0.129	988.0	4366.7
L10	14.9	0.162	193.3	3882.4
L11	15.1	0.142	1,114.7	3836.5
L12	9.0	0.148	3,620.0	5009.0
L13	14.3	0.136	1,744.0	4760.2
L14	19.1	0.154	n/a	1427.6
L15	10.6	0.118	n/a	3142.5
L16	16.6	0.258	704.0	1534.0
L17	14.7	0.125	n/a	3350.7
L18	9.0	0.143	564.0	5249.5
L19	13.7	0.143	n/a	3628.7
L20	8.3	0.126	1,396.0	3699.8
L21	6.7	0.172	980.0	2308.2
L22	4.9	0.366	808.0	3716.1
L23	5.3	0.333	1,680.0	5142.5
L24	7.4	0.203	1,714.0	3558.4
L25	11.9	0.123	568.0	2706.9
L26	1.6	0.149	616.0	3174.2

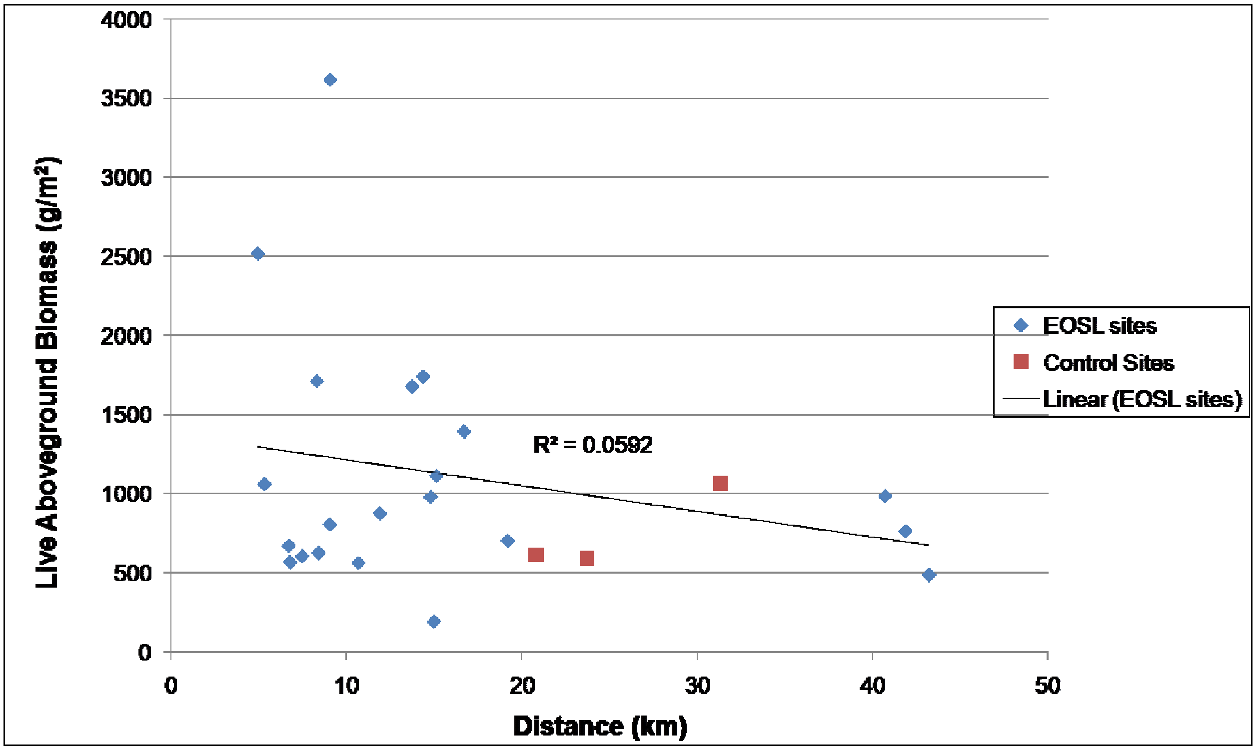

Figure 6. EOSL site live aboveground biomass in relation to distance from the diversion. Simple linear regression statistics did not show a significant relationship between live aboveground biomass and distance.

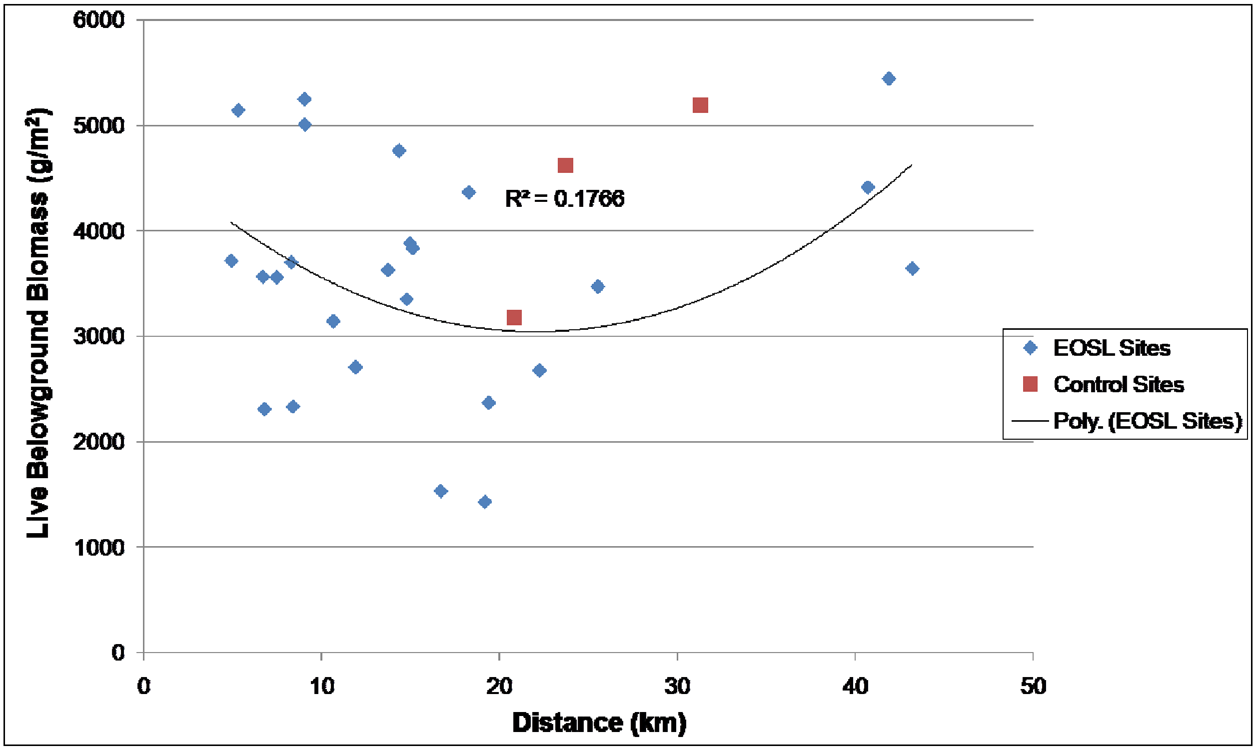

Figure 7. EOSL site live belowground biomass in relation to distance from the diversion. Polynomial regression statistics did not show a strong significant relationship between live belowground biomass and distance.

and *Alternanthera philoxeroides* were the dominant species, while *Spartina patens* was present in minor abundance. Species composition indicates that N1 was a fresh/intermediate marsh strongly influenced by the freshwater diversion. N1 was located on the northern side of the channel immediately south of Big Mar. Site N2 was folded laterally from the effects of Hurricane Katrina, and was hydrologically isolated due to the increased elevation resulting from the folding. N2 was located approximately one kilometer east of site N1 on the southern side of the channel that forms the southern boundary of Big Mar. Also of note was its proximity to the adjacent spoil bank lining the southern edge of the nearby channel. The spoil bank was a barrier to freshwater flow, unlike N1. Standing water was never observed at N2, which was dominated by a thick mat of *Vigna luteola* and *Ipomoea sp.* All other intensive sites had some standing water during at least one of the sampling trips. The Mid site marshes were dominated by *Spartina patens* and *Scirpus americanus,* which are intermediate marsh species and are the result of freshwater from the diversion mixing with the saltwater from the Gulf of Mexico. The Far sites were dominated by *Spartina alterniflora, Spartina patens,* and *Juncus roemerianus.* The presence of the saline marsh species *Spartina alterniflora* indicates that greater levels of saltwater are influencing far site species composition than at either the Near or Mid sites. *Scirpus americanus* and *Spartina patens* were co-dominant at the Control site, with fewer occurrences of *Distichlis spicata*. The intermediate marsh species present at the Control site indicate that saltwater is present in quantities that limit species diversity more than at the Near site. Species present at the Control site suggest that it is an intermediate marsh comparable to the vegetative makeup of the Mid sites.

3.6 Soil Redox Potential

Mean redox potential decreased at the Control, Far, Mid, and Near sites during the four sampling periods beginning in March 2006 and ending in September 2006. Redox potential at the Control, Mid, and Near sites increased during the January 2007 sampling period, while the far site continued to decline slightly before increasing in April 2007 (Figure 6). The Control and Mid sites had higher redox potentials than the Near and Far sites in October 2007.

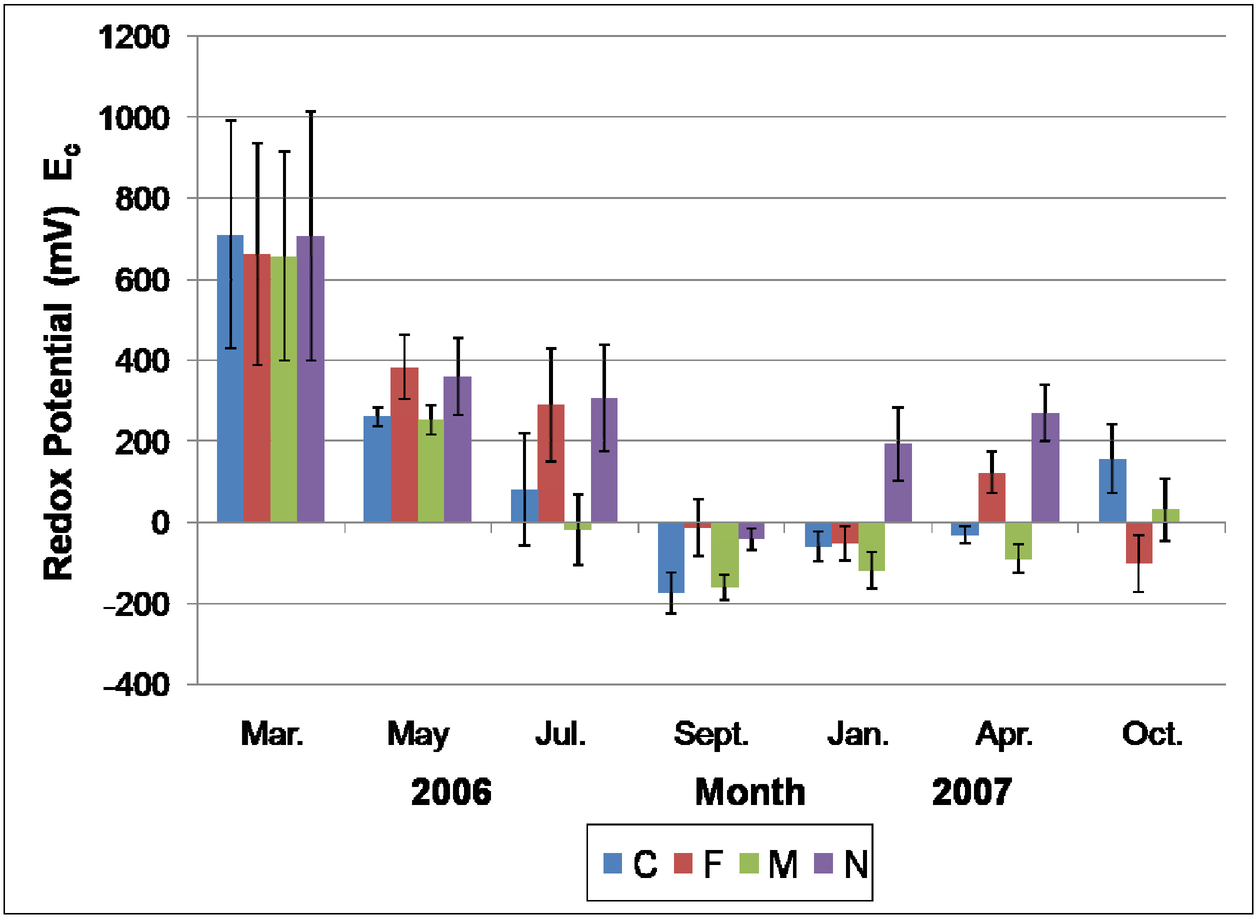

Figure 8. Mean soil redox potentials at the intensive sites from Mar. 2006 – Oct. 2007.

Mean redox potentials were highest in all four regions of the estuary during March 2006. Mean redox potentials at the Near site ranged from +706.7 mV to -40.0 mV (Table 5). Redox measurements at the Mid site were lower than all other sites for every month except September

2006. Redox measurements at the Mid site ranged from +658.3 mV to -159.2 mV. Mean redox potentials at the Far site ranged from +663.3 mV to -11.7 mV. The largest range in redox potentials occurred at the Control site, which ranged from +711.7 mV to -173.3 mV. The results of the Tukey Kramer HSD test comparing means amongst sites showed that redox potential at the Near site were significantly higher than at the Mid site.

Table 6. Mean Redox Potential (Mv± standard error).

		Mar. '06	May '06	Jul. '06	Sept. '06	Jan. '07	Apr. '07	Oct. '07
C	E_c=	711.7±280.7	261.7 ±23.2	82.5 ±138.7	-173.3 ±49.3	-59.2 ±36.2	-30.0 ±21.4	157.5±86.3
F		663.3±273.7	384.7 ±78.9	290.8±138.8	-11.7 ±69.7	-50.8 ±41.5	124.2 ±53.0	-100.8±69.0
M		658.3±257.4	254.7 ±35.5	-17.5 ±86.8	-159.2 ±30.4	-117.5 ±45.5	-88.3 ±35.8	32.5±77.1
N		706.7±308.1	361.7 ±95.1	307.5±132.6	-40.0 ±27.0	195.0 ±91.3	270.0 ±69.5	n/a

3.7 Water Level

Water levels at the Mid site were higher than water levels at both the Control and Far sites from early May until later October (Figure 7). The highest water levels occurred at all sites in late May and early June. After that time water levels at all sites gradually decreased for the remainder of the study period.

Water levels at the Control site were highest in late May. This was one of the few times during the study period when water levels were above the marsh surface. Water depth was <20cm at the site during this approximately one-week inundation period. Water levels at the Far site were also above the marsh surface (<20cm depth) for a brief time in late May and early June. The Far site had slightly higher water levels than the Control site throughout the study period. The Mid site was inundated for the majority of the study period. The highest water levels at the Mid site were also in late May and early June and exceeded 50cm in depth during this time.

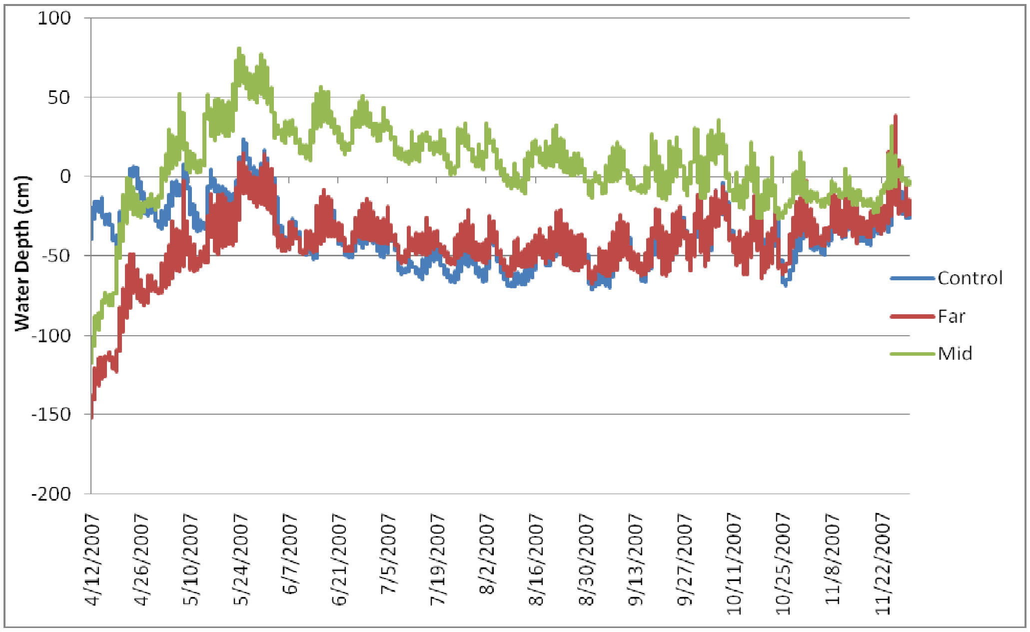

Figure 9. Daily water levels at the control, far, and mid sites.

3.8 Porewater Salinity

Porewater salinity levels were moderate to low. In order to better understand the salinity regime of the estuary, porewater salinity measurements at each of the duplicate study sites were averaged and reported as a means for the Near, Mid, Far, and Control study sites. Results showed that salinity levels at the control site were higher throughout the study period than levels at the Near and Mid sites, but lower than levels at the Far site. Porewater salinity patterns at all sites decreased after September 2006. Mean porewater salinity ranged from 4.8 psu to 0.9 psu at the Near site, from 7.3 psu to 1.8 psu at the Mid site, from 15.0 psu to 4.2 psu at the Far site, and from 9.0 psu to 2.8 psu at the Control site.

The highest value (15.0 psu) was recorded at the Far site during September 2006 and the lowest value (0.9 psu) was recorded at the Near site in June 2007 (Figure 8). Salinity levels peaked at all sites in September 2006, corresponding to the time period with the least amount of freshwater diversion input as well as the month with the highest mean high tide in the Louisiana coastal zone. These results confirm that the diversion flow establishes a year-round salinity gradient in which porewater salinity levels fluctuate among marsh sites but not independently from one another. Results of the Tukey Kramer HSD test showed that salinity at the Near sites was significantly lower than all other study sites while salinity at the Far sites was significantly higher than all other sites. These results have important effects on species composition along the estuarine gradient.

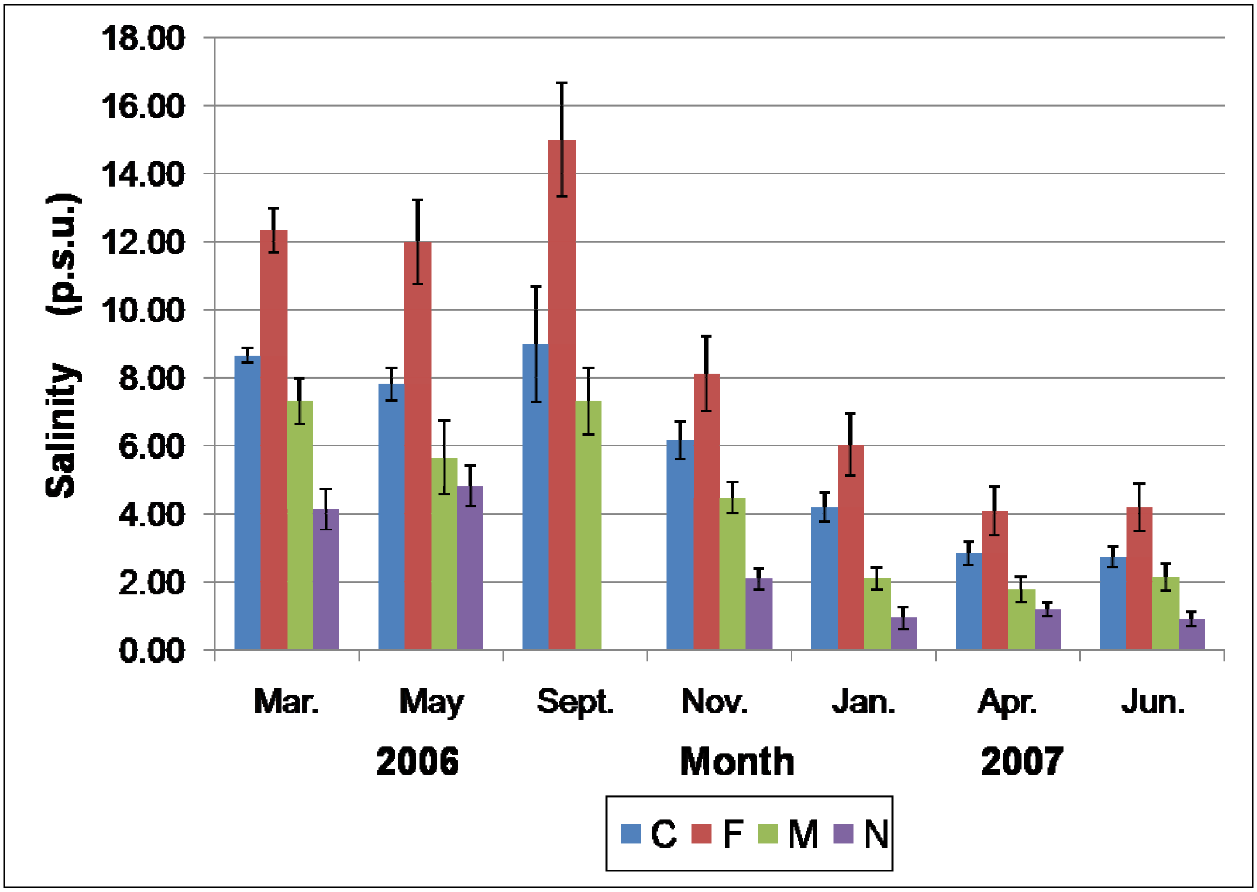

Figure 10. Mean porewater salinity at each site during the study period.

3.9 Porewater Nutrients

Mean NO_x concentrations for the study period ranged from 0.63mg/l at N2 to an undetectable low level (<0.01) at C1, M2, and F1. NO_x concentration variability ranged by a factor of 62. Mean NO_x concentrations at N1, M1, and C2 was 0.01 mg/l. The mean NO_x concentration at N2 when the January and April 2007 measurements are not averaged is 0.03 mg/l. The mean NO_x concentration at N2 is higher than all other sites. NH_4 concentrations ranged from 5.22 mg/l at N2 to 0.17mg/l at M1. Mean NH_4 concentration was 30 times greater at N2 compared to site M1. Mean PO_4 concentrations ranged from 0.91 mg/l at N1 to 0.20 mg/l at M2. Mean PO_4 concentration was 4.5 times higher at N1 than M2.

Porewater nutrient values varied among the sites (Figures 9-11). There was not a strong seasonal trend in NO_x concentrations during the study period (Figure 9). NO_x concentrations were generally less than 0.1 mg/l and were a small percentage of TIN in porewater. NO_x concentrations were high in January and April 2007 at N2 during periods of high diversion discharge. The results of the Tukey Kramer HSD test showed that mean NO_x levels at N2 were significantly higher ($p < 0.0001$) than all other study sites.

As with NO_x, there was a lack of any strong seasonal trend in NH_4 concentrations at the sampling sites (Figure 10). NH_4 concentrations were highest at the Near sites during months of high diversion flow (May 2006, Nov. 2006, and April 2007). In March, NH_4 concentrations were high at the Reference (3.32 and 5.68 mg/L) and Mid sites (0.47 and 2.30 mg/L) then decreased until September and November when they again increased. NH_4 concentrations at the Far site were less variable throughout the study period.

NH_4 concentrations at the Near site ranged from 1.12 mg/l to 0.18 mg/l at N1 and from 11.04 mg/l to 2.25 mg/l at N2. Both of the lowest recorded NH_4 concentrations at the Near sites occurred in January 2007. NH_4 concentrations at M1 ranged from 0.47 mg/l to 0.07 mg/l, while the range at M2 was from 2.30 mg/l to 0.07 mg/l. The highest recorded concentrations at the Mid sites both occurred in March 2006.

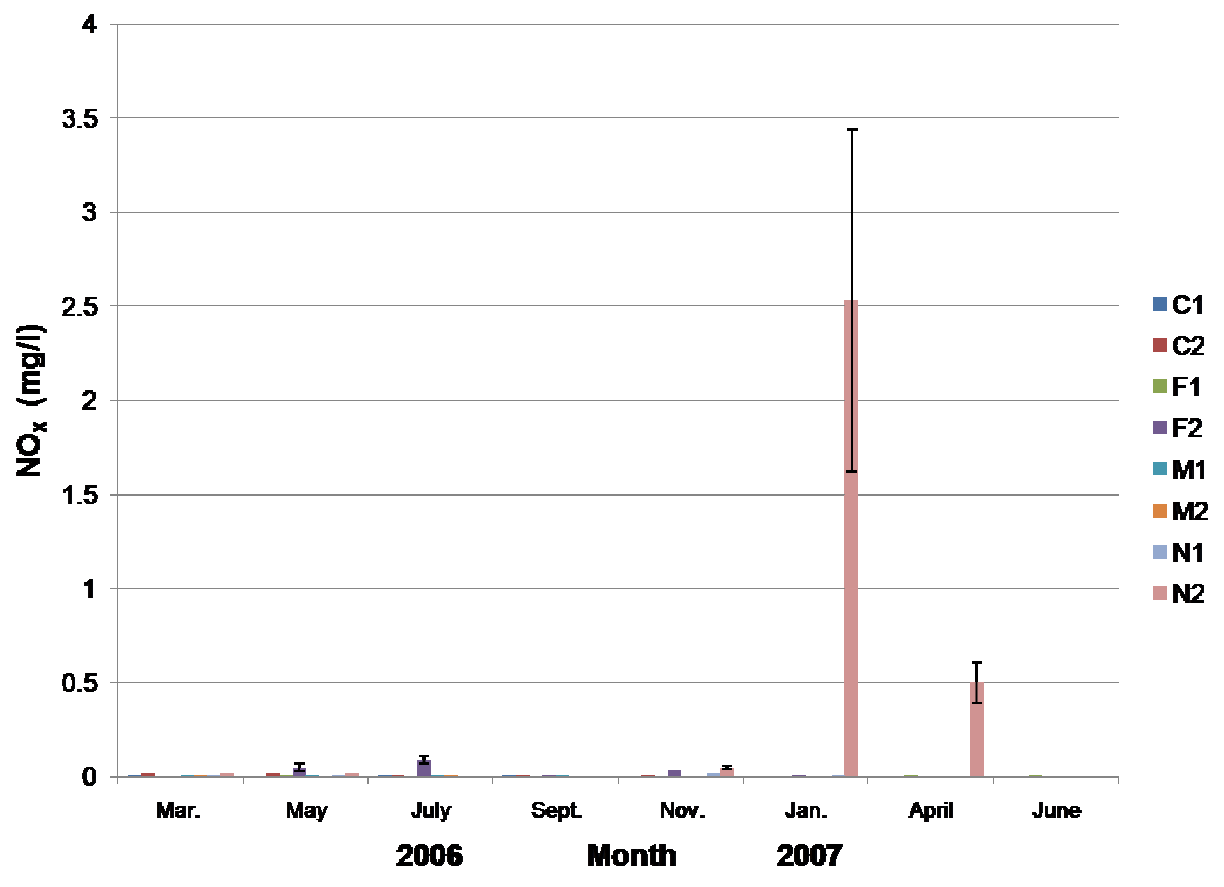

Figure 11. Mean Porewater NO_x concentrations at the study sites (note break in y-axis). What this shows is that NO_x levels were less than 0.1 mg/l with the exception of two samplings at N2 when discharge was high.

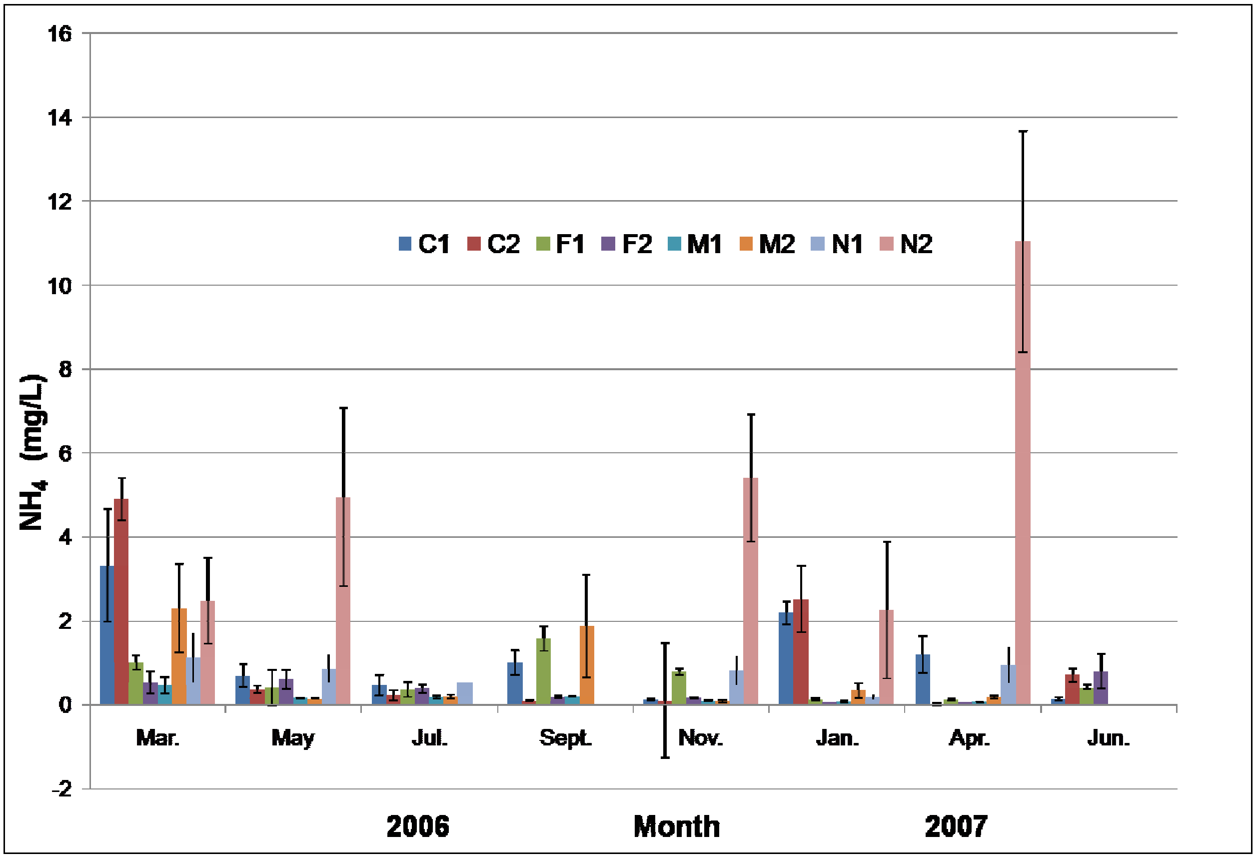

Figure 12. Mean Porewater NH_4 concentrations at each site during the study period.

NH_4 concentrations at the Far sites ranged from 1.46 mg/l to 0.08 mg/l at F1 and from 0.80 mg/l to 0.05 mg/l at F2. Both of the lowest values recorded at the far sites occurred in April 2007. NH_4 concentrations at C1 ranged from 3.32 mg/l to 0.12 mg/l and from 5.68 mg/l to 0.01 mg/l at C2. Both of the highest concentrations recorded at the Reference site occurred in March 2006. The results of the Tukey Kramer HSD test showed that NH_4 concentrations were significantly higher at site N2 ($p < 0.0001$) than at all other sites.

PO_4 concentration peaks occurred in March 2006 at the Mid, Far, and Reference sites (Figure 11). After this initial peak, concentrations decreased and leveled for the remainder of the study period. PO_4 concentrations at the Near sites peaked in May 2006 and declined throughout the winter months before increasing again in April 2007. PO_4 concentrations ranged in variability by a factor of 4 while NO_x ranged by a factor of 62 and NH_4 concentrations ranged by a factor of 30. In general, the Near sites had higher PO_4 concentrations than the other sites. The range of PO_4 concentrations at N1 was from 1.48 mg/l to 0.38 mg/l and from 1.53 mg/l to 0.02 mg/l at N2. The highest PO_4 concentrations occurred at the Near sites in May 2006, while the lowest concentrations were recorded in January 2007 (Figure 11). PO_4 concentrations at the Mid sites ranged from 0.72 mg/l to 0.07 mg/l at M1 and from 0.69 mg/l to 0.03 mg/l at M2. Far site PO_4 concentrations ranged from 0.34 mg/l to 0.06 mg/l at F1 and from 1.02 mg/l to 0.01 mg/l at F2. PO_4 concentrations at the Reference sites ranged from 0.37 mg/l to 0.04 mg/l at C1 and from 0.99 mg/l to 0.15 mg/l at C2.

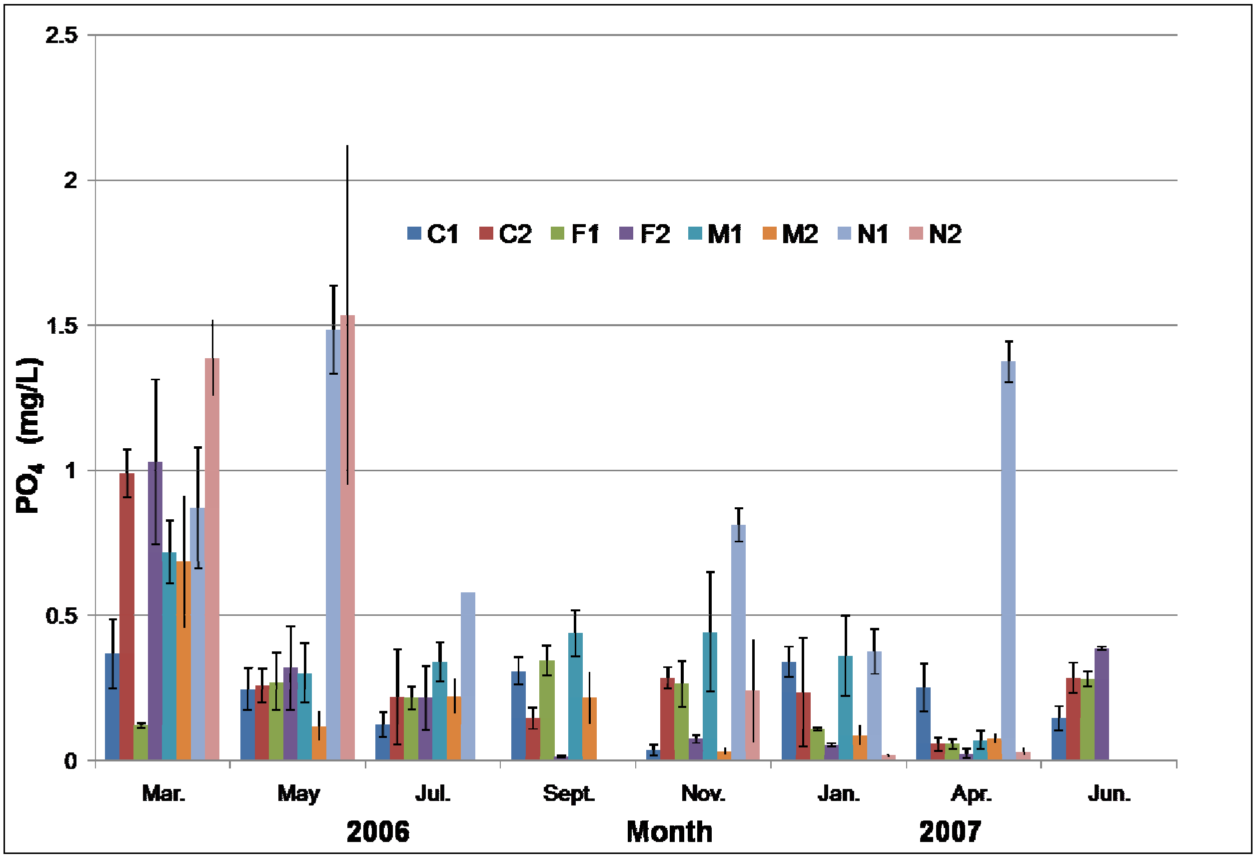

Figure 13. Mean PO_4 concentrations at each site during the study period.

The results of statistical analysis showed that PO_4 concentrations were significantly higher at site N1($p < 0.0001$) than all other sites with the exception of N2. When duplicate sites were combined, the results of the Tukey Kramer HSD test showed that the Near site had significantly higher concentrations of PO_4 than the Mid, Far, and Control study sites.

3.10 Sulfide

Sulfide concentrations fluctuated among sampling trips at the different sampling sites, with little seasonal change (Figure 14). The highest monthly sulfide concentrations occured at either the Control or Mid sites for every sampling period except May 2006. Concentrations were generally higher in the spring of 2006 before declining at each site during the remainder of the study period. F2 had the lowest mean sulfide concentration. Mean sulfide concentrations were significantly higher ($p = 0.02$) at sites C2, M1, and F1 when compared to site F2.

Sulfide concentrations ranged from a high of 59.3 ppm at C2 in March 2006 to undetectably low levels at several sites (Table 7). Near site sulfide concentrations ranged from 27.2 ppm to 0.1 ppm at site N1, and from 10.7 ppm to an undetectably low level in January 2007 at N2. Sulfide concentrations at the Mid site ranged from 32.7 ppm to 1.3 ppm at M1 and from 35.9 ppm to 0.7 ppm at M2. Far site sulfide concentrations ranged from 29.7 to 0.5 ppm at F1, and from 4.7 ppm to below detection limit at F2 in July 2006 and January 2007. Sulfide concentrations at the Control sites ranged from 11.2 ppm to 1.8 ppm at C1, and from 59.3 ppm to an undetectable low in April 2007 at C2.

3.11 Accretion

Short-term marsh accretion for 21 months at the study sites varied slightly among sites (Figure 15). Accretion was greater than all other sites at N1 (2.18±0.14cm) but less than all other sites at N2 (0.86±0.13cm). The high degree of variability between the two near sites was likely due to site specific species composition and hydrology. Accretion at the Mid sites ranged from 1.83±0.12cm at M2 to 1.47±0.12 cm at M1. Accretion at the Far site ranged from 2.01±0.12 cm at F1 to 1.47±0.12 cm at F2. Accretion at the Control site ranged from 2.41±0.21 cm at C2 to 1.41±0.12 cm at C1.

N1 (2.18cm), C2 (2.13cm), F1 (2.01cm), and M2 (1.83cm) had significantly higher accretion rates when compared to M1 (1.47cm), and F2 (1.47cm). C1 (1.41cm) and N2 (0.86cm) had the lowest mean accretion values and were significantly lower than all other study sites. There was not a significant correlation between accretion and the amount of aboveground or belowground productivity at the study sites.

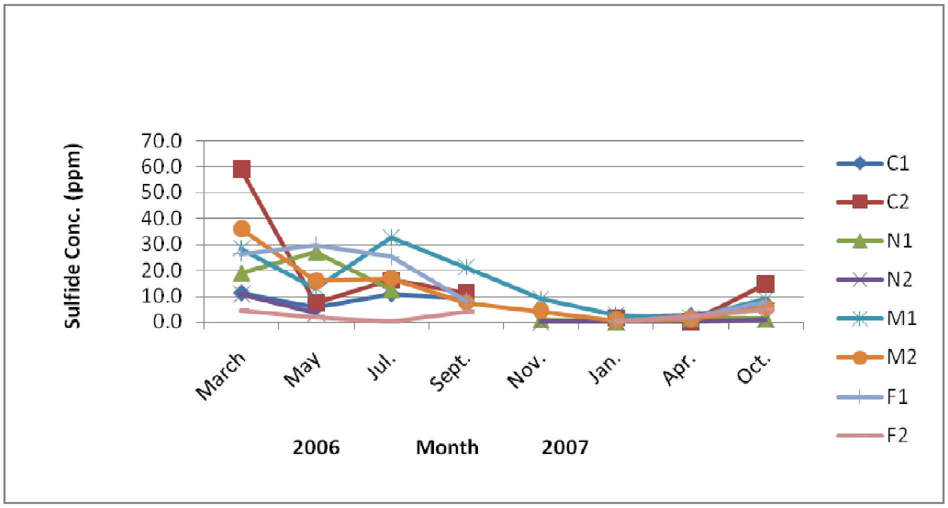

Figure 14. Sulfide Concentrations at each of the eight intensive study sites from March 2006 – Oct. 2007.

Table 7. Sulfide concentrations (ppm) at the eight intensive sites during the study period. N/A means sulfide levels were undetectable.

Month	C1	C2	N1	N2	M1	M2	F1	F2
March 2006	11.2	59.3	18.9	10.7	28.5	35.9	26.5	4.1
May 2006	5.7	7.3	27.0	3.2	13.1	15.8	29.7	1.6
July 2006	10.9	16.2	12.6	n/a	32.7	16.8	25.3	n/a
Sept. 2006	9.1	11.2	n/a	n/a	21.3	7.4	8.4	3.8
Nov. 2006	n/a	n/a	0.9	n/a	8.91	4.16	n/a	n/a
Jan. 2007	1.8	1.5	0.1	n/a	2.7	0.7	0.5	n/a
Apr. 2007	2.5	n/a	1.65	n/a	1.3	1.3	1.6	2.0
Oct. 2007	7.2	14.8	1.3	1.1	8.9	5.6	7.8	4.7

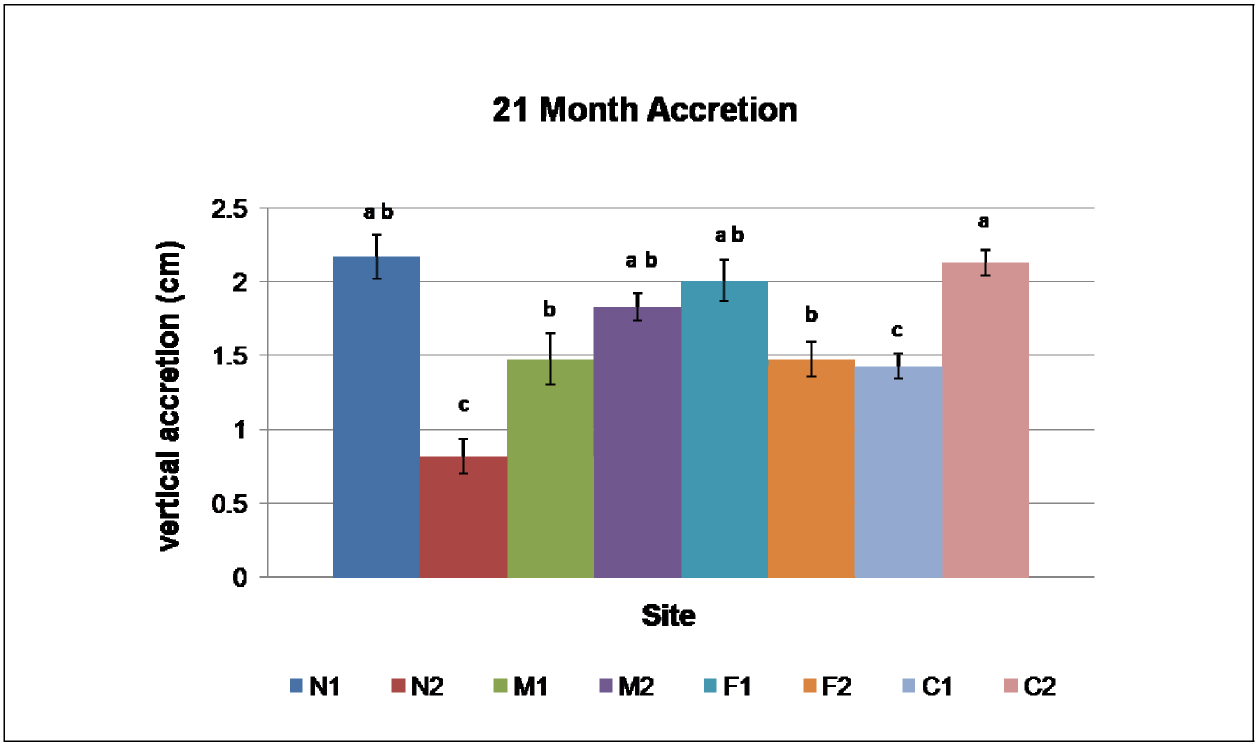

Figure 15. Vertical marsh accretion measurements at each of the eight intensive study sites during the study period from March 2006 – Nov. 2007.

4. DISCUSSION

4.1 Summary

The results of this study showed that there were high levels of biomass, especially belowground, in the marshes of the upper Breton Sound basin that are affected by the Caernarvon diversion. Live belowground biomass was significantly higher at site N1 than at all other study sites except F2. Belowground biomass increased at each site throughout the study period, suggesting recovery after hurricane Katrina. Stress factors affecting biomass production (porewater salinity, sulfide, flooding, and redox potential) were generally low to moderate and nutrient levels did not appear limiting. Below, we compare our results with other studies and analyze the factors affecting biomass production.

Species composition at the different sites reflected salinity levels on a gradient away from the diversion. The changes in species composition along the salinity gradient were typical of estuarine marsh ecosystems (Penfound and Hathaway 1937). Freshwater marsh species present at the Near sites were not present at the Mid, Far, and Control sites. The Far sites contained *Spartina alterniflora* which was absent at the Near sites and present in lesser abundance at the Mid and Control sites. *Spartina patens* occurred over the whole salinity range. Biomass samplings at paired intensive sites did not always contain the same species. Species specific rates of production cannot be determined from this study; however, the ranges of production amongst sites are important for understanding system-wide production levels along the estuarine gradient in Breton Sound.

Innate genetic differences among freshwater marsh species may account for some of the variability in production levels at the Near sites (Mitsch and Gosselink 2000). For example, an adaptation to deeper water levels lead to higher rates of productivity in *Typha angustifolia* when compared to *Typha latifolia* (Kvet and Husak 1978). The difference in the amount of belowground biomass of perennials and annuals affected productivity at the Near sites.

Perennial plants generally store a greater amount of biomass belowground than annual plants (Kvet and Husak 1978). Previously published estimates of above- and belowground net primary production of freshwater marsh species range from a low of 500 g/m^2/yr^{-1} for the broad-leaved monocot *Acorus calamus* up to 6000 g/m^2/yr^{-1} for *Phragmites communis* (Kvet and Husak 1978) Hopkinson et al. (1978) estimated annual production rates for *Sagittaria falcata* to be 1501 g/m^2 in a coastal Louisiana marsh. Net primary production aboveground for *Panicum hemitomon* in a floating coastal marsh in Louisiana was 1700 g/m^2/yr^{-1} (Sasser et al. 1982). Annual NPP at the Near sites is at the high end of these previously published production estimates.

The data indicates that salinity in Breton Sound is the dominant factor influencing plant species composition (Adams 1963; Lane 1999 and 2007). Another major factor effecting species composition at each site is the frequency and duration of flooding. Individual species adaptations to salt stress and flooding play important roles in determining the presence and abundance of species along the estuarine gradient in Breton Sound. Plants which have adapted to deal with longer rates of inundation may be more susceptible to slight changes in porewater salinity. An example is *Alternanthera philoxeroides* which was present at the low salinity N1 but absent at all other study sites. Porewater salinity levels of 1.5 p.s.u. and higher have been shown to decrease plant vigor of *Panicum hemitomon* and lead to the localized decline of the species (Willis and Hester 2004).

Plant species composition may be altered due to storm events such as Hurricane Katrina. A pulse of high salinity water into a freshwater/brackish system can cause negative effects to salt intolerant species. Previous studies have shown that accurately determining the species composition of an intermediate marsh after a major disturbance is difficult due to the biotic and abiotic factors influencing species during the recovery period (Bhattacharjee et al. 2007). Competition and herbivory are two additional factors which effect species composition within

Breton Sound but were not monitored in this study. Marsh recovery from disturbance occurs over a period of time. Species composition, especially at the fresher sites, will most likely change as the process of recovery continues. The species present at the near sites will change over time based on biotic factors such as competition and facilitation. Over time, a greater diversity of plants may become established due to an increased chance of seed dispersal as well as changes in the competitive vigor of the plants already present. The ability of a plant to adapt to varying levels of salinity will also drive species composition at the study sites. An increased frequency of large storm events will lead to the elimination of freshwater plants with low tolerances to salinity. Site N2 will most likely become colonized by rooted plants once nearby plant communities extend their range and begin colonization of the site. However, if the site once again experiences a disturbance that alters the local hydrology and prevents the establishment of rooted plants, vine-like species such as *Vigna luteola* and *Ipomoea sp*. will continue to dominate.

Previous studies have shown that belowground productivity in coastal salt marshes range from a low in North Carolina of 460 g/m^2/yr (Stroud and Cooper 1968) to a high in Louisiana of 11,676 g/m^2/yr (Darby 2008, Table 7). Higher productivity generally occurs at lower latitudes due to a longer growing season (Mendelssohn and Morris 2000). Belowground productivity at all sites besides N1 was within the range of previously published data (Table 8). N1 (14485 g/m^2/yr) had a higher rate of belowground productivity than any previously published results (Table 8). The high productivity at site N1 is likely the result of riverine inputs, which increase nutrient availability, decrease porewater salinity, raise redox potentials, and lower sulfide concentrations (DeLaune et al. 1990; Mendelssohn and Kuhn 2003; Carpenter et al. 2007). This site is located in the direct path of diverted water, as compared to N2 which is a folded marsh located behind a spoil bank. The high productivity at F2 was probably due to decreased sulfide concentrations from increased tidal flushing, relatively low salinities, and nutrient input from the river from the south. Although porewater salinity levels were higher and redox potentials lower at F2 than at the Near sites, repeated tidal flushing decreases sulfide accumulation (DeLaune et al. 1983). This site was also well drained.

Belowground biomass peaked during April 2007 at all sites except the Mid sites which peaked in June 2007. The seasonal variation in belowground biomass throughout the year indicates that increased production begins during the winter months, peaks in the spring, and declines until the next winter. Longer growing seasons combined with milder winter growing conditions in Louisiana contribute to less seasonality in above and belowground plant biomass peaks (Good et al. 1982; McIntire and Dustan 1976). Darby (2006) found similar trends in the growth patterns of belowground biomass, but with greater seasonality than was found in this study. Belowground biomass increased throughout the study suggesting recovery from Hurricane Katrina.

Practically all biomass at the study sites occurred in the upper 30 cm of the soil column and the greatest amount of belowground biomass production occurred at the 0-20cm depth range. These results are consistent with those reported by White et al. (1978) who found root mass in a Louisiana salt marsh to extend no more than 20 cm beneath the soil surface. The Control and Mid site had fairly even distribution down to 20 cm and rarely if ever had significant biomass below a depth of 25cm. Near site biomass was highest in the upper 25 cm. The greatest amount of biomass at depths >20cm was found at the Far site. The presence of root mass at depths

Table 8. Reported aboveground and belowground productivity g/m^2/yr^{-1} values for *Spartina alterniflora* marshes (unless other species specified). Estimation method appears in perentheses next to reported value. Max-min. estimate reported for this study. PSC= Peak Standing Crop method. NR= height form not reported.

Sampling Location	Height Form/Species	Above NPP	Below NPP	Source
North Carolina	Short	650	460	Stroud & Cooper 1968
North Carolina	Tall	1300	500	Stroud 1976
Massachusetts	NR	420	3500	Valiela et al. 1976
Georgia	Short	1350	2020	Gallagher & Plumley 1979
	Tall	3700	2110	
New Jersey	Short	500	2300	Smith et al. 1979
Nova Scotia	NR	803	1051	Livingstone & Patriquin 1981
Georgia	Medium	2840	4780	Schubauer & Hopkinson 1984
South Carolina	Short	1272	5445	Dame & Kenny 1986
	Medium	775	------	
	Tall	2460	2363	
Brazil	NR	-----	569	Lana et al. 1991
Louisiana	Short	1821	11676	Darby et al. 2007
Louisiana	NR	831(PSC) 831(Milner-Hughes) 1231(Smalley) 1873(Wiegert-Evans) 1437(Lomnicki)	-----	Kaswadji et al. 1991
Michigan	*Typha latifolia*	1604 (1978) 1284 (1979)	-----	Dickerman et al. 1986
South Carolina	NR	402 (1984) 1042 (1985) 700 (1986) 621 (1987) 408 (1988)	-----	Morris & Haskin 1990
Louisiana	*Distichlis s.*	3237	-----	Hopkinson et al. 1978
	Juncus r.	3416		
	Phragmites c.	2318		
	Sagittaria f.	1501		
	Spartina a.	2658		
	Spartina c.	1355		
	Spartina p.	6043		
Louisiana	NR	1410	-----	Kirby & Gosselink (1976)
Louisiana	NR	1527 (Smalley)	-----	White et al. 1978
	NR	2895 (Wiegert)		
	Spartina p.	1342 (Smalley)		
	Spartina p.	1428 (Wiegert)		
Mississippi	*Distichlis s.*	1484	-----	de la Cruz (1974)
	Spartina p.	1922		
	Juncus r.	1697		
	Spartina a.	1964		
Georgia	NR	2883	-----	Odum & Fanning (1973)
Georgia	NR	973	-----	Smalley (1958)
Louisiana	Several species included	652 (C1) (max-min)	6506	This study
		1158 (C2)	5776	
		872 (N1)	14,485	
		964 (N2)	6028	
		1236 (M1)	7453	
		326 (M2)	7346	
		1123 (F1)	4776	
		392 (F2)	10,131	

beyond 20cm is the result of a highly oxidized root zone with higher redox potentials and decreased soluble sulfide concentrations. The slightly deeper rooting zone at the Far site may be attributed to consistently higher redox potentials at this site when compared to the Control and Mid sites. Low biomass below 20 cm at the Control and Mid sites correlate to lower redox potentials. A higher redox potential at depth implies a greater oxidized soil root zone, which facilitates above and belowground production (Mendelssohn and Morris 2000). These conditions lead to greater belowground production at depth with the two sites experiencing higher redox potentials.

Based on the results of this study, the edaphic factors effecting belowground growth are low to moderate. This combined with relatively high nutrient concentrations leads to high biomass production. There was a net increase in belowground biomass at all study sites from March 2006 to October 2007. There is no indication that variations in nutrient levels inhibited belowground biomass production at any of the study sites. The overall net increase in belowground growth at each site may be evidence of marsh recovery and increased marsh stability.

Aboveground NAPP for this study ranged from a low of 326 $g/m^2/yr$ at M1 to a high of 1236 $g/m^2/yr$ at M2. Lower NAPP values including those at M1, F2 (392 $g/m^2/yr$), and C1 (652 $g/m^2/yr$) are more similar to estimates made in northeastern coastal salt marshes, such as Massachusetts (420 $g/m^2/yr$) and New Jersey (500 $g/m^2/yr$; Valiela et al. 1976; Smith et al. 1979). NAPP at M2, N1, N2, F1, and C2 were more similar to the previous regional NAPP values contained in Table 7. EOSL aboveground biomass was as high as 5434 g/m^2 during the three year EOSL sampling carried out from 1999-2001 (Lane unpublished data). The greatest amount of EOSL aboveground biomass in this study was 3620 g/m^2.

Aboveground biomass was generally higher during the growing season as has been reported for many studies of coastal marsh productivity (Valiela et al. 1976; Smith et al. 1979; Schubauer and Hopkinson 1984, Hopkinson et al. 1978). Aboveground biomass patterns were more seasonally variable than belowground biomass. Decreased aboveground biomass in winter is due to low growth rates as a result of cooler temperatures (Kaswadji et al. 1990). This trend is consistent with other studies that reported similar declines in aboveground production during winter months in southern Louisiana salt marshes (Kirby and Gosselink 1976; Hopkinson et al. 1978).

Above- and belowground biomass variability reflects photosynthate incorporation. Previous research by Hull et al. (1976) and Lytle and Hull (1980) have shown that little photosynthate incorporation occurs in belowground tissues during the summer months. This decreased belowground production continues until aboveground biomass senesces in the fall when a transfer of energy occurs, and belowground biomass increases until another energy transfer occurs in the spring. Photosynthate incorporation increases in aboveground tissue in the spring. The lack of a strong seasonal variation in biomass in Breton Sound most likely is the result of a subtropical environment, which allows for longer growing seasons than marshes in more northern latitudes where freeze events limit marsh productivity.

The patterns of above and belowground biomass in the Breton Sound estuary are a result of a combination of edaphic factors that affect marsh growth. These factors include porewater salinity, redox potential, sulfides, and nutrients as well as flooding levels. Soil redox potential measurements exhibited a strong seasonal trend at all sites, with the highest levels occurring in spring and then declining until the fall. Consistently lower redox potentials at the Mid and Control sites suggest that decreased plant productivity, as the result of root oxygen deficiency, occurs at these sites when compared to the Near and Far sites (Mendelssohn and Morris 2000).

Mean water levels at the Mid site were higher than those at the Control and Far sites for the majority of the study period, indicating that the increased water levels led to decreased redox potential at these sites.

Mean redox potentials at each site during the study period were calculated and plotted with results from Howes et al. (1981, Fig. 16) which shows the difference in the degree of soil reduction measured in a salt marsh containing tall and short forms of *Spartina alterniflora*. Tall form *S. alterniflora* occurs in more oxygenated soils compared with short form *S. alterniflora* that is present under more reduced soil conditions (Howes et al. 1981; Mendelssohn and Morris 2000). The mean redox potentials for the Near sites and F2 are within the range in which tall *S. alterniflora* grows while the Mid sites, Control sites, and F1 have mean redox potentials typical of short form *S. alterniflora.* This figure illustrates that more reduced soil conditions at these sites may contribute to decreased biomass production.

Porewater salinity increased with distance from the diversion as a result of freshwater input through the diversion structure. Lane et al. (1999, 2007) reported that salinity in surface waters in the upper Breton Sound Basin increased with distance from the diversion, indicating that surface water salinities are strongly related to porewater salinity. A similar salinity gradient was reported for the Atchafalaya Bay region as a result of Atchafalaya River discharge (Day et al. 1992). Nitrogen uptake by marsh plants decreases as salinity increases (Morris 1984, Burdick and Mendelssohn 1987; Webb and Mendelssohn 1996). Thus, the salinity gradient should favor more efficient nutrient uptake and higher productivity nearer the diversion, as occurred at N1. However, biomass did not decrease away from the diversion due to differences in the frequency and duration of flooding along with the variability in species composition at the study sites.

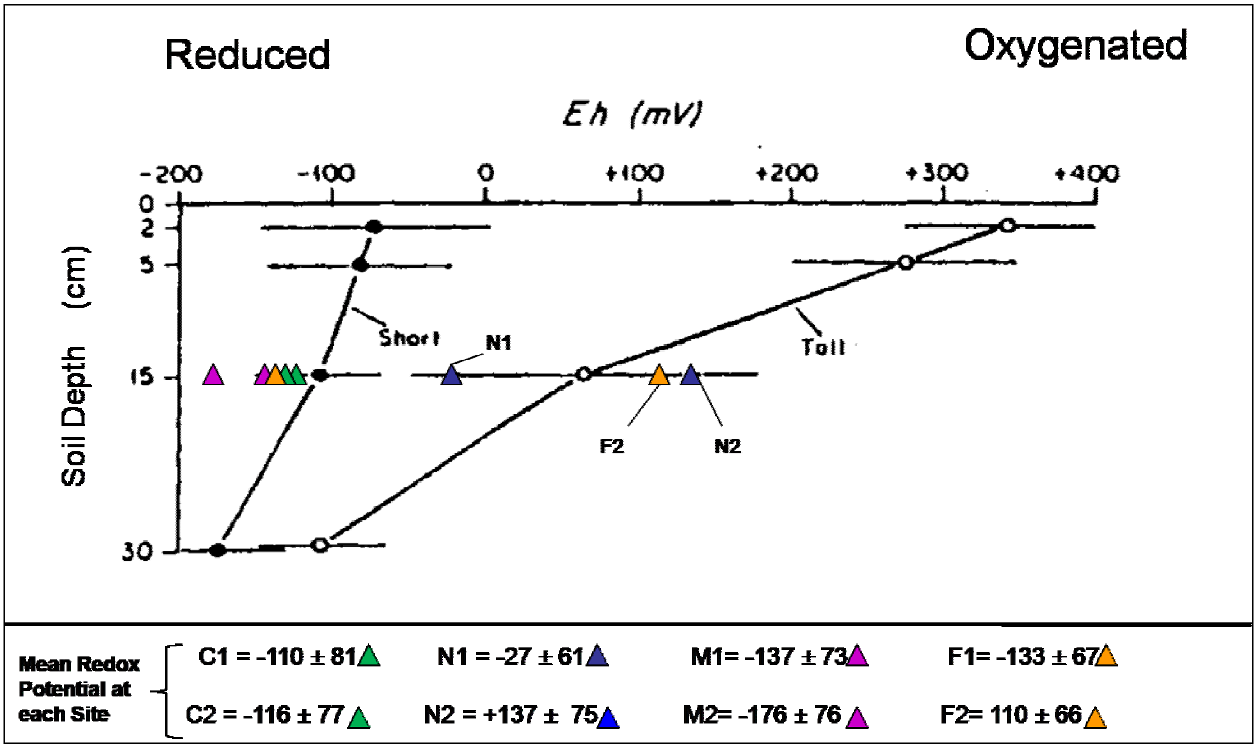

Figure 16. Mean redox potentials from this study compared to those from Howes et al. 1981. Modified from Howes et al. (1981).

Numerous studies have shown that marsh productivity varies depending on the species and its ability to deal with salt stress and flooding (Spalding and Hester 2007). For example, *Spartina patens* production is more negatively affected by increased flooding depth than changes in salinity below 6 p.s.u. (Burdick et al. 1989; Pezeshki and DeLaune 1991; Hester et al. 1996, 2001, Spalding and Hester 2007) while *Panicum hemitomon* productivity is stimulated by a certain degree of increased flooding (Fisher 2003; Willis and Hester 2004, Spalding and Hester 2007). The interactive effects of increased salinity and flooding also effects individual plant species production in a highly variable manner based on the species adaptations to salt stress and flooding. Increased salinity and increased flooding has been shown to cause dieback of the oligohaline (<5 p.s.u.) marsh species *Sagittaria lancifolia* (Webb and Mendelssohn 1996). When flooding was reduced in the presence of increased salinity, plant dieback did not occur. *S. lancifolia* was present at both of the Near sites.

Porewater salinity was greater at the Control sites compared to the Mid sites because the Control site is less directly impacted by the diversion (Lane et al. 2007). Porewater salinity was highest in September when the diversion discharge was lowest and there is greatest penetration of Gulf waters into Louisiana estuaries due to high Gulf levels (Morris et al. 1990). Porewater salinity levels were highest when redox levels were lowest, likely partially reflecting sulfate reduction.

The conditions at the Control site suggest that tidal saltwater inputs and longer periods of inundation during the month of September contributed to high porewater salinity levels and low redox potentials throughout the basin. Decreased flushing, due to low discharge rates coupled with increased evapotranspiration rates during the late summer months also may be contributing variables.

Sulfide concentrations during the growing season seem to be related to lower biomass at the Control and Mid sites. *Spartina alterniflora* biomass growth is negatively impacted by soluble sulfide levels greater than 1mM total soluble sulfide (Bradley and Dunn 1989; Koch and Mendelssohn 1989; Koch et al. 1990). Sulfide concentrations exceeded this concentration at C2 and M2 during March 2006. Sulfide concentrations were below the 1 mM threshold level at all other sites during the study. Sulfide levels appeared to be the factor most influencing ($p < 0.05$) belowground growth when analyzed using multiple regression statistics. However, the amount of variation explained is low ($R^2 = 0.13$).

Mean sulfide concentration at each site during the study period was compared to data published by Mendelssohn and Morris (2000, fig. 17). Figure 17 illustrates the relationship between sulfide concentration and biomass production of *Spartina alterniflora*. Sulfide concentrations of 1 mM or greater decreased plant biomass production. The mean sulfide concentration at each of the study sites was less than 1 mM. Mean sulfide concentrations were highest at C2 (0.58 mM), the Mid sites (0.46mM; 0.34mM), and F1 (0.45mM). These are the same four sites that had the lowest redox potentials. The results suggest that the influence of saltwater from the Gulf combined with more reduced soil conditions is leading to the production of sulfides at these sites. However, the sulfide concentrations present at each site are not so high as to drastically inhibit biomass production. These results also suggest that freshwater from the diversion is helping to decrease sulfide concentrations at the Mid and Far sites.

There were greater flushing and lower water residence times at the Near and Far sites. Decreased redox potentials at the Mid site were most likely due to longer periods of flooding, higher water levels, and reduced flushing of toxins from plant roots (Ponnamperuna 1972; DeLaune 1983). Flushing events occur at the Near site due to riverine inputs from the

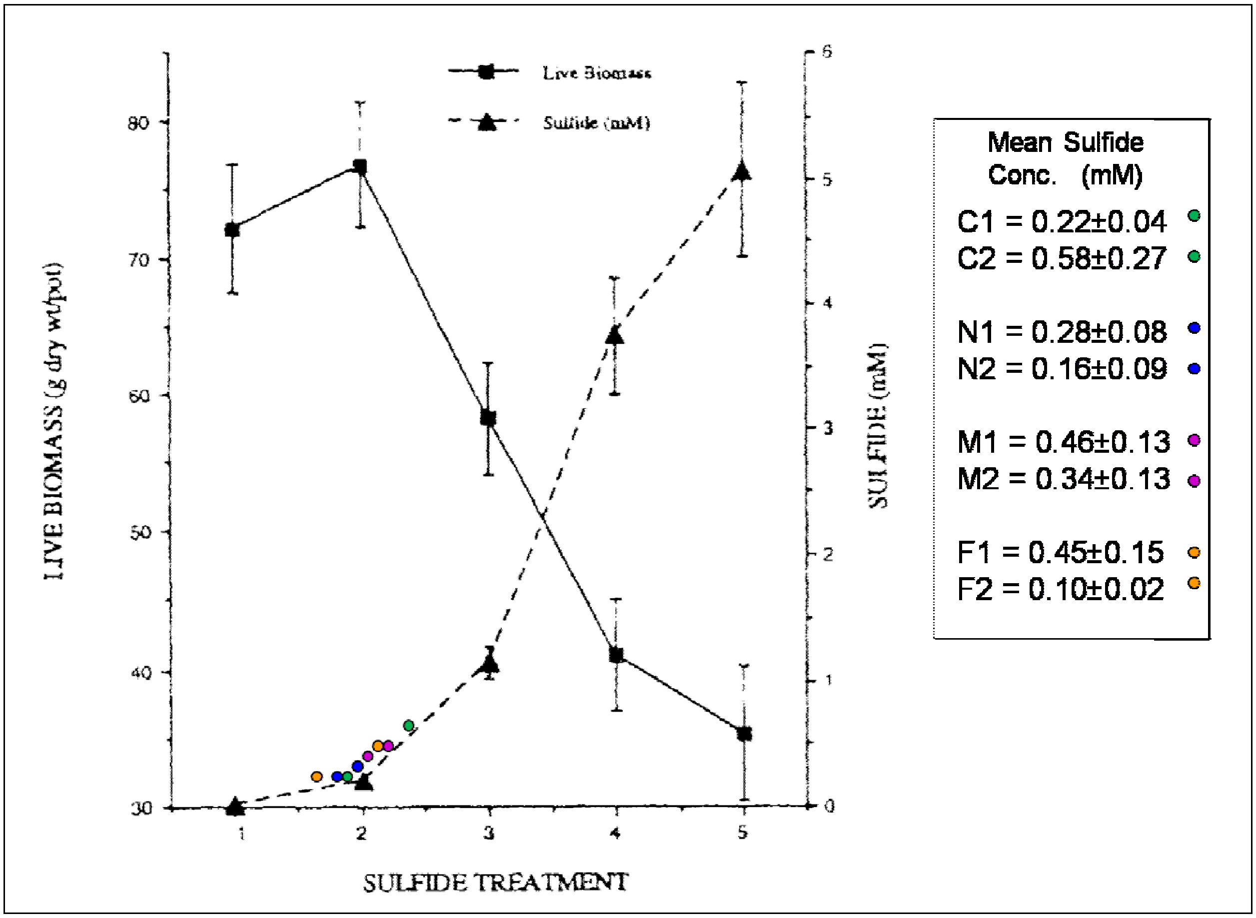

Figure 17. The growth response of *Spartina alterniflora* to varying sulfide concentration. Results from the current study were added to the curve to demonstrate the effects sulfides may be having on biomass production. Modified from Mendelssohn and Morris (2000).

Caernarvon diversion, and at the Far site due to high frequency tidal inundation. Increasing the frequency and volume of riverine discharge could potentially decrease sulfide levels and increase belowground biomass productivity at the Mid site.

Soil nutrients were present in adequate quantities at each site throughout the study to suggest that soil nutrients were not limiting to plant growth. Studies have shown decreases in nutrient and sediment concentrations, and increases in salinity as diverted river water flows through the Breton Sound estuary (Lane et al. 2004, 2006a). Lane et al. (1999) also showed increased nutrient levels at the far marshes due to inputs from the Mississippi river. Flooding of the marsh surface occurs often near the Caernarvon structure when it is in operation, but decreases rapidly with distance (Snedden 2006). The results of porewater chemistry from this study are consistent with previous studies, which found that nutrients are quickly assimilated near the diversion and are generally lower at sites further from the diversion (Lane et al. 2004), although, as noted above, nutrient levels are sometimes higher at the most distant marshes due to input from the south from the Mississippi River.

Nutrient concentrations fluctuated at the Near sites, while concentrations at the other study sites remained fairly constant throughout the study period. Delaune et al. 2005 found similar results at the Davis Pond river diversion where nitrate levels decreased with distance from the freshwater diversion structure. NO_x and NH_4 concentrations at N2 were significantly higher ($p < 0.0001$) than at any of the other sampling sites. PO_4 concentrations at N1 were significantly higher ($p < 0.0001$) than all sampling sites except N2. These results confirm previous reports (Mitsch et al. 2001, Lane 1999 and 2007) that nutrients are being provided to the upper estuary by the Caernarvon diversion (Wissel 2005). However, these nutrient amendments are not completely assimilated by marsh plants in the upper estuary due to variability in site specific hydrology as well as differences in species composition. Hydrologic impediments to diversion discharge flow are likely one reason for the lower production levels measured at N2 in comparison to N1. N2 had significantly higher levels of NO_x and NH_4 in soil porewater when compared with the other study sites ($p < 0.0001$). Elevated levels of NO_x and NH_4 in the porewater of N2 may be due to the lack of rooted plant species present at the site. The decreased levels of nutrients at the Mid and Far sites suggest that assimilation of diversion transported nutrients is occurring before the areas around Grand and Little lakes where the Mid sites were located (Lane et al. 2004). In addition, any nutrients transported into the estuary from the south (the Bird's foot delta) are present in soil porewater at significantly lower levels than those at the near sites.

Lane et al. (2006) reported wetland vertical accretion rates in the Breton Sound estuary to range between 0.75 and 1.57 cm y^{-1}. The same study estimated RSLR rates in Breton Sound to range from 0.15 to 0.45 cm y^{-1}. This is consistent with the results of vertical marsh accretion in this study, which ranged between 0.49 cm y^{-1} at N2 and 1.25 cm y^{-1} at N1, suggesting that all the study sites are stable or accreting at a rate which offsets RSLR.

There has been speculation that increased nutrient loading due to river diversions will lead to lower belowground biomass production and make marshes less sustainable. Darby (2006) added nutrients to salt marsh plots and reported that at high nutrient loading, belowground productivity was reduced. We did not find this in our study. One factor explaining these results is that Darby added very high levels of nutrients ranging from 80 to 900 g $N/m^2/yr$. By comparison, total nitrogen (TN) loading to upper Breton Sound is from 1.9 to 3.2 g $N/m^2/yr$ (Hyfield et al. 2008). Most other marsh fertilization studies and natural loadings are less than 200 g $N/m^2/yr$. In addition to nutrients, river diversions deliver freshwater that lowers salinity, mineral sediments that contain nutrients as well as increase bulk density, and iron that precipitates sulfide (DeLaune et al. 1990; Mendelssohn and Kuhn 2003; Carpenter et al 2007). In conclusion, our study did not find lowered below ground biomass or reduced below ground allocation of nutrients with distance from the diversion.

4.2 Management Implications

The results of this study suggest that marsh net primary belowground production is highly variable and ranges from 4776 $g/m^2/yr$ up to 17890 $g/m^2/yr$. A net increase in belowground primary production occurred at all study sites during the study period. This finding suggests that marsh recovery through increased root structure establishment is occurring at all study sites. Net primary production aboveground ranged from 326 $g/m^2/yr$ to 1581 $g/m^2/yr$. The variability in marsh primary production within the estuary is the result of both biotic and abiotic factors. Several abiotic factors were monitored in this study. Porewater salinity in the Breton Sound estuary is significantly lower at the study sites nearest the diversion. Porewater nutrient levels are significantly higher at the Near sites. Increased inundation at the Mid sites is resulting in significantly lower redox potential at these sites when compared to the Near sites. The cause of this increased inundation should be further analyzed in order to determine best management practices.

The effects of Hurricane Katrina on marsh hydrology appear to be a major factor influencing marsh production along the estuarine gradient. For example, a change in marsh elevation as a result of the storm impeded overland marsh sheet flow at N2. The effects of this change are evident not only when comparing productivity levels at the two Near sites but also when assessing species composition between sites. The variability in marsh elevation and local hydrology are significant factors effecting productivity which should be accounted for in future studies.

These findings of the effects of Hurricane Katrina on marsh productivity may be enhanced further through more in depth comparisons with historic data, seasonal system-wide assessments of the relative cover and species composition in relation to total marsh area, and an increased number of sampling sites. The results of water level data, porewater salinity, and redox potential data, show that the estuary's hydrology is an important factor influencing the abiotic conditions under which marsh plants exist. Further research directed at the complex interactions occurring between meteorological forcings, diversion discharge, and tidal flux could lead to more effective management of the Caernarvon freshwater diversion. Operating the diversion consistently in response to saltwater intrusion events may help to maintain marsh species composition along the estuarine gradient while increasing species survival and rates of production.

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