Wetland Response to Stormwater Discharge at the Pointe au Chien Pumping Station, Pointe au Chien Wildlife Management Area Terrebonne Parish, LA

March 1, 2016

A Report of the:

Barataria-Terrebonne National Estuary Program

By:

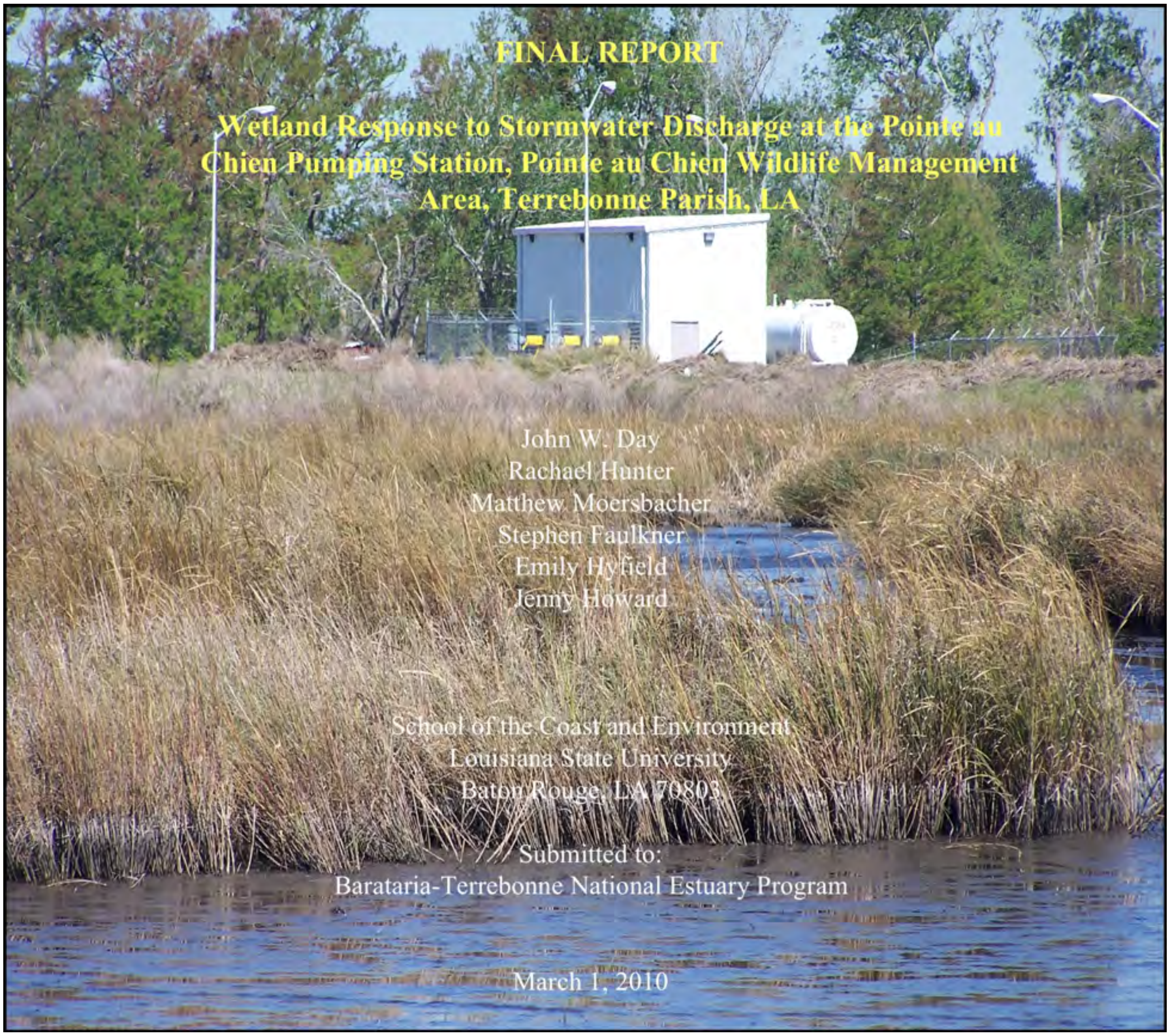
Dr. John Day, Rachael Hunter, Dr. Stephen Faulkner,
Emily Hyfield, Jenny Howard

Prepared for the**:**

Environmental Protection Agency's National Estuary Program

EPA Tracking Numbers 1997-05, 1999-01, 2007-31

This project was also funded and implemented under the following programs:
EPA Gulf of Mexico Program Shellfish Challenge and EPA Gulf of Mexico Program
Assistance ID numbers:
Phase I: MX-97460502   Phase II: MX-97460502-0



FINAL REPORT

Wetland Response to Stormwater Discharge at the Pointe au Chien Pumping Station, Pointe au Chien Wildlife Management Area, Terrebonne Parish, LA

John W. Day
Rachael Hunter
Matthew Moersbacher
Stephen Faulkner
Emily Hyfield
Jenny Howard

School of the Coast and Environment
Louisiana State University
Baton Rouge, LA 70803

Submitted to:
Barataria-Terrebonne National Estuary Program

March 1, 2010

TABLE OF CONTENTS

TABLE OF FIGURES

Executive Summary

This report summarizes baseline data collected at a marsh within the Pointe aux Chenes Wildlife Management Area before and after stormwater discharge to determine impacts of stormwater redirection on wetland ecosystems. This research was divided into several stages designed to specifically answer the following questions:

1. What are the impacts of stormwater on wetland sediment metals concentrations?
2. What are the impacts of stormwater on wetland water quality, specifically nutrient concentrations, pathogens, and dissolved oxygen?
3. What are the effects of stormwater on marsh elevation, specifically local accretion and surface elevation change?
4. What are the effects of stormwater on wetland vegetation, specifically emergent biomass and areal coverage?

The study area was located in the Pointe aux Chenes (PAC) Wildlife Management Area (WMA), south of Houma, Louisiana (Figure 1). The Terrebonne Parish government began installation of three pumps in April 2005 to discharge stormwater into the study area. The pumps became operational in August 2005 and were turned on for the first time immediately following Hurricane Rita. Nine experimental plots were established within the area to be impacted by stormwater discharge, with three distance classes from the point of discharge (Near, Mid and Far). Three replicate plots were identified within each distance class. Three Reference plots were also established in an adjacent marsh not impacted by the stormwater discharge. An additional water quality station was located at the culvert in the Isle de Jean Charles road at the southern end and only outlet of the study area.

Pre- discharge, sediment metals and nutrient concentrations were within the range reported for other marshes within in the Louisiana coastal zone. Following stormwater discharge, mean sediment metals concentrations were below detection limits and very similar to pre-discharge concentrations. These data indicate that the stormwater runoff is not a source of metals contamination within the study area.

Stormwater discharge has had a positive impact on water quality within the study area. Mean salinity was 10.35 pre-discharge and 4.64 post-discharge. Salinity could also have been impacted by the construction of a one-way flap gate in the culvert under the Isle de Jean Charles road. Other water quality parameters were lower after stormwater discharge. Fecal coliform bacteria (FCB, 205 cfu *pre* vs 9.00 cfu *post*), chlorophyll *a* concentrations (64 ug/L *pre* vs 47 ug/L *post*), and total phosphorus concentrations (0.45 mg/L *pre* vs 0.28 mg/L *post*) decreased with stormwater inputs. Since FCB did not increase after introduction of stormwater runoff into the wetland, these results support the hypothesis that birds utilizing the wetland were contributing to the high FCB concentrations. It appears that the stormwater reduces salinity, FCB, and chlorophyll *a* and total phosphorus concentrations in surface waters, improving water quality.

Mean pH (8.28 *post* vs 7.76 *pre*), dissolved oxygen (5.79 mg/L *post* vs 4.42 *pre*), total suspended solids (117 mg/L *post* vs 49 mg/L *pre*), and total nitrogen concentrations (1.83 mg/L *post* vs 1.49 mg/L *pre*), were higher after stormwater discharge than before discharge. The increase in dissolved oxygen is probably due to aeration during pumping into the wetland and oxygen production from benthic algae. Total Suspended Solids (TSS) not only increased in the sites receiving stormwater, but also in the Reference site, indicating that although stormwater might be increasing TSS concentrations in the treatment areas, storm passages and water movement probably re-suspend sediments as well. Mean concentrations of nitrate+nitrite and total nitrogen were higher following stormwater discharge than before discharge. This would indicate that stormwater is adding nitrogen to the system. However, because mean total nitrogen concentrations were also higher in the reference site, the reason for the increasing nitrogen is unclear.

Within the treatment sites, mean accretion measured pre-stormwater discharge (4.62±0.20 cm/yr) was significantly higher than that measured post-discharge (1.50±0.10 cm/yr). Pre-discharge accretion rates were most likely higher than post-discharge rates due to the failure of the local flood protection levee during Tropical Storm Bill. Sediment from the levee was deposited on the sites, especially the Near site. Because of this confounding factor, it is difficult to determine impacts of discharge on surface elevation. Other studies involving discharge of treated municipal effluent on wetlands have shown increasing elevation through an increase in sediment accretion and organic matter deposition as proposed by Breaux and Day (1994). Aerial photographs of the sites before and after discharge do show an increase in mud flats and, thus, it is assumed that stormwater discharge is increasing surface elevation.

During post-stormwater discharge, mean marsh biomass in the treatment sites averaged 2256±140 $g/m^2/yr$, which was significantly higher than mean biomass before discharge (1341±159 $g/m^2/yr$). In addition to increasing vegetative biomass, it is apparent from aerial photographs that areas of mudflat are also increasing. These areas may need to be re-planted or re-seeded if they are not colonized on their own. Plant species diversity should develop as the elevation is raised and freshwater and nutrients are added via pumping.

Introduction

This report summarizes vegetation, sediment, and water quality monitoring data to evaluate the impacts of stormwater discharge to coastal wetlands adjacent to the Pointe aux Chenes Pumping Station in Terrebonne Parish, Louisiana. The work is based on the Monitoring Strategy Workshop for the Pointe aux Chenes Stormwater Project held on February 20, 2000 and sponsored by BTNEP, Terrebonne Parish, and the Gulf of Mexico Program. This report summarizes baseline data collected to determine the initial conditions within the marsh at Pointe aux Chenes (PAC) prior to the installation of the stormwater pump and post-discharge monitoring to determine impacts of stormwater on vegetation productivity, sediment accretion, sediment metals concentrations, and water quality.

The introduction of freshwater restores wetlands and enhances sustainability through increased vegetative productivity and accretion rates (Day et al. 2004). However, the introduction of stormwater into a wetland has the potential to significantly alter water quality, vegetation composition, and marsh structure. The concentrations of nutrients (nitrogen, phosphorus and silicate), fecal coliform bacteria (FCB) and sediments are traditionally high in urban runoff (Schueler 1987). To determine impacts of introduced stormwater runoff to the PAC wetland, baseline monitoring (pre-discharge) and post-discharge monitoring were conducted.

Objectives

The objectives of this study were to:

1. Obtain an understanding of the ecology and water quality of the study area; and
2. Determine the potential for stormwater diversion to restore degraded coastal wetland habitat.

The proposed work was divided into several stages designed to specifically answer the following questions:

1. What are the impacts of stormwater on wetland sediment metals concentrations?
2. What are the impacts of stormwater on wetland water quality, specifically nutrient concentrations, pathogens, and dissolved oxygen?
3. What are the effects of stormwater on marsh elevation, specifically local accretion and elevation change?
4. What are the effects of stormwater on wetland vegetation, specifically emergent biomass and aerial coverage?

Methods

Study Area

The study area is located in the Pointe aux Chenes (PAC) Wildlife Management Area, south of Houma, Louisiana and within the Barataria-Terrebonne estuarine system (BTES; Figure 1). Estimated land loss rates in the Terrebonne basin were 24.6 km^2 per year from 1965 to 1978 and 26.9 km^2 per year from 1978 to 1990 (Reed 1995). Shoreline loss, due to erosion by storms, boat wakes, etc., represents about 31% of the land loss. One of the primary contributing factors to land loss is likely the river levees built to contain historical spring flooding of the Mississippi River, along with regional subsidence (Reed 1995, Day et al. 2007).

Once a freshwater marsh, the PAC wetland is now a brackish/saline marsh community. Saltwater intrusion killed many of the original species and they were replaced with salt-tolerant species. The construction of levees isolated much of the PAC wetland from upland runoff, which was a major source of freshwater. Saltwater intrusion from the south provided a source of salinity to the former freshwater and low salinity wetlands.

The Terrebonne Parish government installed three stormwater pumps in April 2005 that discharge 207 ft^3/sec (133 MGD, million gallons per day) into the study area, when operational. The pumps were installed in August 2005 and were turned on for the first time immediately following Hurricane Rita. Although stormwater discharge was not directly measured, amount of water pumped into the areas was estimated based on the amount of fuel consumed (Table 1). Stormwater pumped in 2008 was more than double what was pumped in previous years, most likely due to storm frequency during that year.

Table 1. Estimated amount of stormwater pumped into the study area, based on fuel consumption of the stormwater pump.

Year	Fuel use (gallons)	Fuel use (gallons/hr)	Approximate pump run time (hrs)	Estimated water pumped (gallons)
2005	4810	10.5	458	824,571,429
2006	5506	10.5	524	943,885,714
2007	4897	10.5	466	839,485,714
2008	14,414	10.5	1373	2,470,971,429

In 2002, three years prior to the pumps becoming operational, experimental plots were established within the wetland at increasing distances from the point of discharge (Near, Mid and Far), with three replicates of each distance (Figure 1). Plots 1, 2, and 3 on Figure 1 are located closest to the discharge point and termed as Near plots. Plots 4, 5, and 6 are the Mid plots and 7, 8, and 9 are the Far plots. Three reference plots (plots 10, 11, and 12 in Figure 1) were also established in an adjacent marsh that is not impacted by stormwater discharge due to separation by a natural ridge. This design allowed for the evaluation of the impact of the stormwater discharge. Boardwalks were constructed at each of these plots and were accessible only by boat (Figure 2). All water samples and water quality measurements were collected or measured from the end of the boardwalks. An additional water quality station was located at the culvert in the Isle de St. Jean Charles road at the southern end of the study area. Within each plot, paired mudflat and vegetated subplots were established for seed bank studies. Pre-discharge monitoring time period was from June 2002 to August 2003. Post-discharge monitoring began in April 2006 and ended in October 2008.

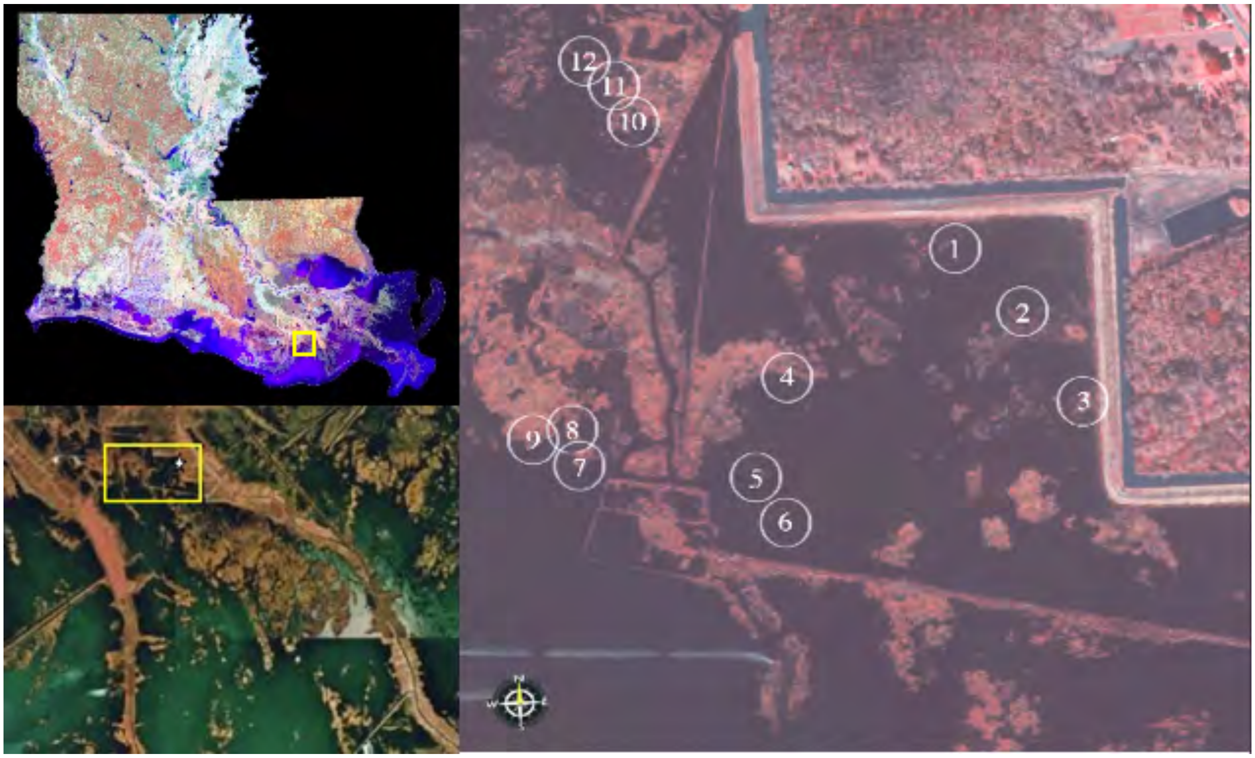


Figure 1. Location of study plots within the Pointe aux Chenes Wildlife Management Area south of Houma, Louisiana.

Figure 2. Sampling boardwalk at one of the study plots.

Seed Bank Characterization

Ten-centimeter deep cores were randomly collected at each subplot with a McCauley auger in June 2002. To compensate for the root material, twice the numbers of cores were taken in the vegetated areas than the mudflat. The cores were transported on ice and refrigerated at $4^{o}C$ for 10 days.

In the greenhouse, basins measuring 32cm x 17.5cm x 10cm were prepared by adding 5.5 cm of sterilized sand. Two, eight-ounce plastic cups, slotted around the bottom sides were added to each basin to water the soil without disturbing the seeds. Each basin was labeled for one of the subsamples and was randomly arranged on the bench in the greenhouse. Each subsample was thoroughly mixed and spread into its individual basin at a depth of approximately one-centimeter. Deionized water was added to each basin until the water was even with the soil surface. To ensure that the sand was not supplying any of the seeds, two control basins were filled with only sterilized sand.

Approximately 250 ml of water were added to each basin every two days. The water level was allowed to rise to the level of the sand to moisten the soil. After 7 days, the first seedling germinated. Seedlings were then allowed to grow until physical differences could be identified. Each seedling was then grouped according to these different physical characteristics. A representative of each species was transplanted to a larger separate container for further identification to species.

All remaining seedlings were identified to genus, counted, and removed after 56 days. After seedling removal, the soil was gently raked allowing for a new germination period. No new germination was observed after 99 days.

Baseline Soil Chemistry

Because stormwater is a collection of runoff from areas surrounding the PAC wetland, metal and nutrient concentrations were monitored. Wetlands have proven to have the ability to absorb metals from contaminated water (Pardue et. al. 1988), therefore baseline concentrations of metals and micro-nutrients were determined. Two soil cores were collected from each study plot in June 2002 and in January 2007. Soil cores were transported on ice and refrigerated for 7 days. After 7 days, the soil cores were thoroughly mixed and sieved to remove root material and further homogenize the soil. The soil was then dried at 100^{o}C for 24 hours. Once dried, KCl extraction and total digestion were performed on the samples. Transition metals (copper (Cu), zinc (Zn), cadmium (Cd), lead (Pb), chromium (Cr), and nickel (Ni)) and micro-nutrients (iron (Fe), manganese (Mn), calcium (Ca), and magnesium (Mg)) were analyzed using inductively coupled argon plasma spectrometry (ICP) and phosphorus, sodium (Na), nitrate (NO_3) and ammonium (NH_4) were analyzed with a Lachat Automated Flow Analyzer (Lachat 1999).

Water Quality Samples

Eight pre-discharge sampling trips were conducted between June 2002 and August 2003 when the sample sites were accessible by boat. Sampling dates included June 6, August 30, October 8, and November 12, 2002, and April 30, June 2, July 2, August 20, 2003. Of the eight trips, two occurred just after tropical storm passages. The October 8, 2002 trip followed the passage of Tropical Storms Isadore (September 26, 2002) and Lilli (October 2, 2002). The July 2, 2003 trip took place two days after Tropical Storm Bill (June 30, 2003).

Post-discharge monitoring began in early 2006. Fifteen water sampling trips were conducted between April 2006 and July 2008. Sampling dates included April 11, June 15, July

25, August 15, and October 20, 2006, January 25, October 19, and December 31, 2007, and January 21, February 28, March 27, April 30, May 20, June 17, and July 25, 2008.

Water quality (temperature, dissolved oxygen (D.O.), pH, specific conductivity, and salinity) was measured using an YSI-85 and Corning Checkmate II pH meter. Water samples were also collected in 1-liter dark, acid-washed bottles for analysis of chemical and biological parameters. The sample bottles were pre-rinsed three times with sample prior to the collection of the sample water. Sample water was transferred to a sterile Whirl-Pak, sealed and stored on ice for fecal coliform bacteria (FCB) analysis within 6 hours of collection. The 1-liter collection bottle was filled a second time and stored on ice until laboratory analysis within 24 hours of sample collection. Each bottle was sub-sampled for total suspended sediments (TSS), organic matter (OM), chlorophyll *a* (Chl *a*), nitrate+nitrite (NO_x), NH_4, ortho-phosphate (PO_4), silicate (SiO_3), chloride (Cl), total nitrogen (TN) and total phosphorus (TP).

Upon receipt of the sample bottles, 125-ml acid washed bottles were rinsed with sample water and filled for total nutrient analysis (TN and TP). Four 20-ml vials were rinsed with filtrate, which passed through a Whatman GF/F 0.7-micron, 2.5-cm, pre-rinsed filter and then filled with filtrate for dissolved nutrient analysis (NO_x, NH_4, PO_4, SiO_3, Cl). The total bottle and dissolved nutrient vials were frozen and analyzed at LSU.

All pre-discharge nutrient analyses were performed on a LACHAT auto-analyzer located in the LSU Wetland Biogeochemistry Institute. Total nitrogen and total phosphorus were analyzed as NO_X and orthophosphate using the persulfate digestion technique (Qualls 1989). All dissolved nutrients were analyzed using U.S. Environmental Protection Agency (USEPA) certified techniques: Nitrate+Nitrite - copperized cadmium column, that reduced all nitrate to nitrite (QuikChem method 10-107-04-1-A; Lachat 1999); Ammonia - Berhelot reaction method

(QuikChem method 10-107-06-1-A; Lachat 1999); Silicate - molybdate-reactive silica method (Lachat 1999, QuikChem method 10-114-27-1-A); Orthophosphate - ascorbic acid QuikChem method 10-115-01-1-B (Lachat 1999); Chloride - ferric iron (QuikChem method 10-117-07-1-A; Lachat 1999). Post-discharge nutrients were measured at the LSU Wetland Biogeochemistry laboratory run by Dr. Ronald DeLaune using EPA-certified methods.

Chlorophyll *a* was analyzed on material capture filter used for the dissolved nutrients. The filter was placed into a petri dish that was wrapped in aluminum foil and frozen until analysis within 7 days. Chlorophyll *a* was determined using a modified version of Burnison (1980) where chlorophyll was extracted using a 40:60 DMSO:90% Acetone solution and measured on a Turner Designs TD-700 flourometer.

Total suspended solids (TSS) and organic matter (OM) were determined gravimetrically using a modified procedure from Greenberg et al. (1985) with GF/F 0.7micron 4.7cm filters. OM was determined by weight difference using combustion of the filters for 35 minutes at 550^{o}C and weighed. Fecal coliform bacteria analysis was performed by the Institute for Ecological Infrastructure Engineering Water Quality Laboratory at LSU utilizing the Fecal Coliform Membrane Filtration Technique (Rose et al. 1975).

Water Levels

In order to monitor water level fluctuations within the study area, instantaneous Infiniti water level recorders were installed in May 2003 at plot 5 within the Mid site and plot 11 within the Reference area (Figure 4). The recorders collected data every 4 hours and were downloaded to a Hewlett-Packard 48 calculator in July 2003 and October 2003. Water level gauges were destroyed during Hurricane Katrina. New gauges were not installed and so no post-discharge data exist.

Accretion and Elevation Change

Two methods were utilized in order to estimate marsh accretion and elevation change. Accretion was measured as accumulation of material over feldspar marker horizons (Baumann et al. 1984). One 0.25-m^2 subplot was established in each of the sampling stations (Figure 3). The marker horizons were installed March 28, 2002 in the Near, Mid, and Far plots and on May 17, 2002 in the Reference plots. About one cm of feldspar clay was applied over the soil surface within a 0.25m^2 quadrant. Five to ten replicate cores were taken from each plot on February 5, 2003, January 25, 2007, March 27 and October 29, 2008. The thickness of the sediment layer above the feldspar was measured by taking core samples using liquid nitrogen to ensure the cores stability.

Change in sediment elevation was measured with a sedimentation-erosion table (SET; Figure 4; Boumans and Day 1993) at each distance class. The SETs were installed February 5, 2003 and baseline elevation was measured in May 9, 2003. Post-discharge elevation was measured on October 29, 2008.

Figure 3. Preparing quadrant for application of feldspar.

Figure 4. Sediment elevation table for measuring sediment elevation (left) and sediment elevation table in use (right).

Vegetation

Percent cover and end-of-season live Biomass (EOSL) were collected at the 12 marsh sites in September or October prior to senescence. EOSL was collected using a 530-cm^2 ring (0.053 m^2) on October 13, 2003, October 20, 2006, October 19, 2007, and October 29, 2008

(Figure 5 right). Each sample was sorted into live and dead matter, dried at 60^{o}C and weighed. Percent cover was estimated in a 1-m^2 area during spring to determine the typical vegetation composition of the marsh during the fall of 2003 (Figure 5 left).

Figure 5. Collection of end-of-season-live biomass using a 530-cm^2 ring (left) and estimation of percent cover using a 1m^2 plot (right).

Statistical Analyses

To determine difference in water quality parameters, accretion, vegetation productivity, and metal concentrations among treatments (Near, Mid, Far, and Ref sites), one-way analysis of variance (ANOVA, $\alpha = 0.05$) was conducted using JMP 7.0 statistical software (SAS Institute Inc. 1999). Plots were replicated for each treatment (n = 3). For significant ANOVA tests, comparison of means were made using Tukey-Kramer Honestly Significant Difference (HSD) test (Sall et al. 2005). To determine differences in water quality, accretion, vegetation productivity, and metal concentrations due to discharge of stormwater, one-way ANOVA was also conducted but data were only analyzed in the Near, Mid, and Far sites.

Results and Discussion

Seed Bank Characterization

There was growth in 92% of the greenhouse basins containing soil from the vegetated sites. Within these basins many seedlings emerged, often representing several different species. The three species that germinated from the mudflat soil were *Echinochloa crusgali* and *Leptochloa fascicularis*. 410 stems were counted in the basins containing the soil from the vegetated subplots. The species represented within the soil from the vegetated area were *Amaranthus australis*, *Eleocharis sp*., *Pluchea odorata*, *Cyperus odoratus*, and *Bacopa caroliniana*. *A. australis* and *B. caroliniana* are found in fresh and brackish wetlands, while *P. odorata* and some *Eleocharis* species are salt tolerant. The salt tolerance of *C. odorous* is unknown.

Through visual estimation of percent cover, both *Spartina patens* (37%) and *Spartina alterniflora* (37%) were the dominant species in the marsh in 2003. Each species represented 37% of the total percent coverage, for a total of 74% of the vegetation present in the marsh. The vegetative community in the marsh differed from what was present in the seed bank. The dominant species present in the seed bank were *Eleocharis sp*. and *Pluchea odorata* (Figure 6). There were 236 stems counted for *Eleocharis sp*. and 146 stems counted for *Pluchea odorata*. Twenty-one stems were counted for *Cyperus odoratus*, seven for *Amaranthus australis*, and two for *Bacopa caroliniana*. None of the stems counted were *Spartina* sp.

Eleocharis sp. is a wind-dispersed seed and *Pluchea odorata* has a long-lived seed. Therefore, it is possible that the seed bank represents vegetation communities of adjacent areas or historical communities. Leck (1989) suggested that the differences observed in the vegetation and the seed bank might be due to both the viability and number of seeds contributed by the dominant species. Leck et al. (1989) also suggested that the conditions present might not allow for germination of the species within the seed bank. The dense canopies of the vegetation currently growing at PAC may explain the absence of the seed bank species in the vegetation community (Baldwin et al. 1996). It has been suggested that increased salinities may contribute to the lack of seed bank species represented in the vegetative community (Baldwin et al. 1996). Previous studies have found that wetlands with a dominant *Spartina* sp. community present did not have *Spartina* sp. present in the seed bank (Baldwin et al. 1996, 1998). Baldwin et al. 1996 found that the seed bank from a *Spartina patens* dominated community had *Eleocharis parvula* as the dominant species present in the seed bank. These findings are consistent with the results presented here.

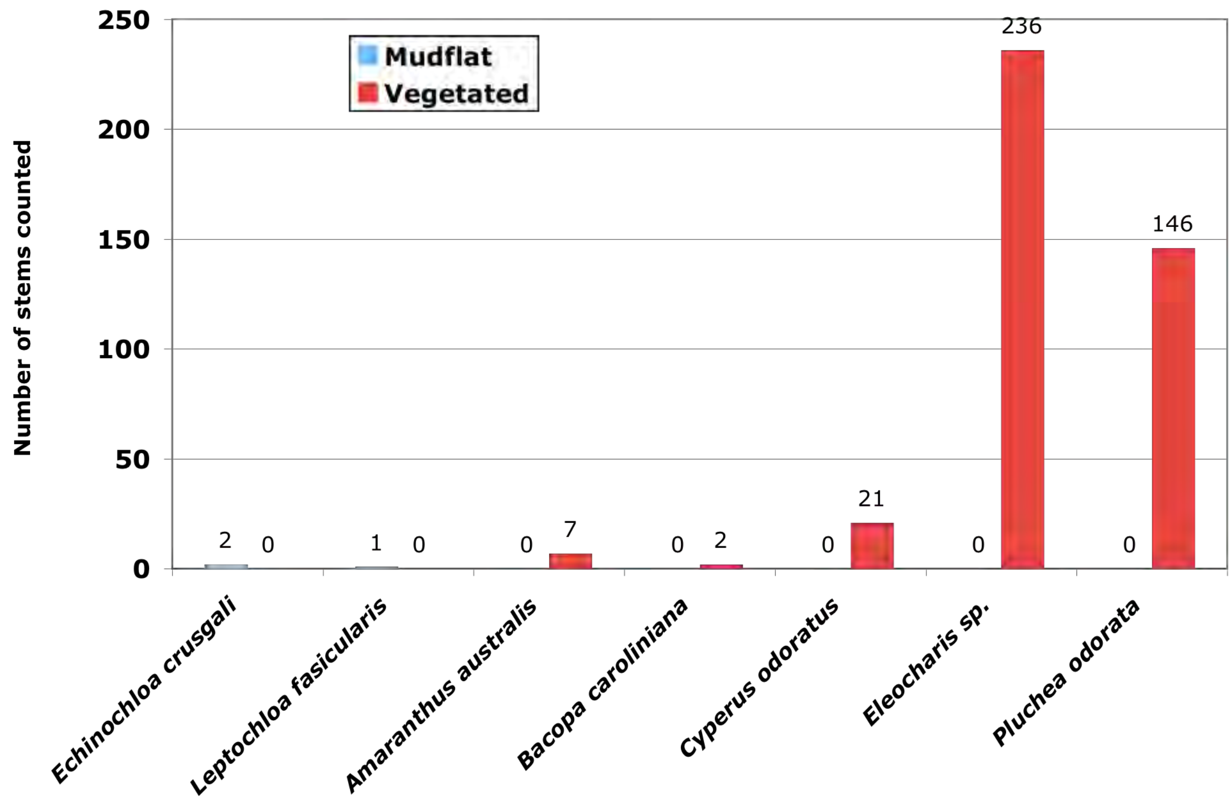


Figure 6. Number of plant stems in seed banks of Pointe aux Chenes marsh prior to stormwater discharge.

The absence of viable seeds within the mudflat area may be due, in part, to the flooding regime. The PAC study area has been isolated from a freshwater source through the construction of levees and saltwater intrusion from Terrebonne Bay increased salinity in the area. High salinity in combination with flooding are known stressors (Baldwin et. al. 1996) and it is probable that the current hydrologic regime of the marsh led to the lack of viable seeds found in the mudflat. Seasonal flooding may have also contributed to the seed accumulation within the vegetated areas. Many wetland species seeds can float and will accumulate in vegetated areas, therefore resulting in a larger seed bank in the vegetated sites compared to mudflat areas (Leck et al. 1989). van der Valk et. al (1978) found that as little as two cm of water can inhibit germination in species and also suggested that while mudflat species may be able to germinate under drawdown conditions, they are unable to donate seeds to the seed bank because they die when flooding occurs.

In addition to inundation and salinity stress, the soil in the mudflat was not well consolidated. The soil was unconsolidated most likely because the area has not supported vegetation for a long period and also because the levee at the site has been under construction and breached in several storms. This breaching has resulted in loose sediment that does not allow the establishment of seeds. The loose nature of the sediment may be subject to mixing through storms and/or animal and human interaction. Mixing of the soil may bury seeds, leading to increased mortality or inadequate sample collection. It has been previously reported that soils of fine texture have lower rates of seedling success and seed viability (Leck et al. 1989).

Given the seed bank present in the vegetated areas, the addition of freshwater from stormwater diversions should enhance the growth and development of freshwater species. The mudflat areas, however, do not currently have the same capacity. These results suggest that the mudflat areas will re-vegetate naturally very little under the current flooding patterns. To restore this area of open water/mudflat, manual replanting may be necessary.

Baseline Soil Chemistry

Pre-Discharge

Of the metals of concern, Zn had the highest mean concentration throughout the study site (18.63 – 23.61 ug/g; Table 2). Sediment metal concentration limits have been established by the USGS for the USEPA. These standard concentrations reflect probable effect concentrations (PECs) for a particular metal, and any concentration above the set PEC is likely to be harmful to the ecosystem (USEPA 2000). The PEC for Zn is 459 ug/g, which is well above the concentrations found in the soils of the PAC wetland. Concentrations of Cd, Cu, Pb, Cr, and Ni were also below set PECs (Table 3). Of the micronutrients measured (Fe, Mn, Ca, Mg, P, and Na), Fe and Na had the highest mean concentrations (63.43 – 114.83 ug/g and 81.53 – 89.96 ug/g, respectively; Table 2). These data indicate that prior to stormwater discharge, sediments of the PAC wetland did not have any metal contamination.

For most of the metals measured, there were differences detected in concentrations among the sites studied (Table 2). Generally, when differences were detected, the Far sites had the highest concentration and the Reference sites had the lowest concentration. This indicates a source further from the land rather than from freshwater runoff into the system.

Table 2. Mean soil nutrient and metal concentrations (ug/g (±SE)) in soils at Pointe aux Chenes Wildlife Management Area prior to stormwater input. Sites with different letter subscripts are significantly different at an α = 0.05.

Analyte	Near	Mid	Far	Reference	P value
Copper	1.96±0.20	2.04±0.30	1.86±0.42	1.10±0.31	0.1539
Zinc	29.90±1.60 ^{b}	29.10±1.60 ^{b}	38.00±4.30 ^{a}	28.70±1.80 ^{b}	0.0455
Cadmium	0.005±0.001^{a}	0.006±0.001^{a}	0.004±0.001^{a}	0.001±0.001 ^{b}	0.0003
Lead	1.66±0.02 ^{a}	1.74±0.11^{a}	1.17±0.34 ^{a}	0.31±0.14 ^{b}	0.0001
Chromium	0.70±0.02 ^{b}	0.73±0.05 ^{ab}	1.33±0.33 ^{a}	0.45±0.05 ^{b}	0.0042
Nickel	0.63±0.03 ^{a}	0.62±0.03 ^{a}	0.78±0.12 ^{a}	0.25±0.07 ^{b}	0.0001
Iron	7563±543 ^{bc}	9259±660 ^{b}	11483±608 ^{a}	6343±216^{c}	0.0001
Manganese	223.8±42.7 ^{b}	670.5±147.6 ^{ab}	1357.7±481.8 ^{a}	433.5±109.8 ^{b}	0.0005
Calcium	3122.2±408.5	3898.1±561.0	3791.3±481.8	3473.1±483.3	0.6659
Magnesium	2691.2±58.0 ^{b}	2805.0±121.8 ^{b}	3344.6±163.5 ^{a}	2461.6±92.3 ^{b}	0.0001
Phosphorus	307.4±21.3 ^{b}	391.3±44.9 ^{ab}	494.7±55.6 ^{a}	270.8±25.0 ^{b}	0.0011
Sodium	8152.6±224.4	8771.2±169.4	8960.3±343.0	8996.3±207.9	0.0679

Table 3. Mean sediment probable effects concentrations (PECs; USEPA 2000) and metals concentrations found in wetlands receiving stormwater runoff for 10 years or less (10-year sites) and 30 years.

Metal	Concentration (ug/g)		
	PECs	**10-year sites**	**30-year sites**
Cadmium	4.98	2.01	1.60
Chromium	111	7.27	9.76
Copper	149	41.20	18.00
Lead	128	16.00	41.30
Nickel	48.6	12.50	15.40
Zinc	459	52.50	68.50

DeLaune et al. (1981), Pardue et al. (1988), and DeLaune and Gambrell (1996) documented metal concentrations along the coastal zone of Louisiana. These studies included areas with known sources of contamination and areas that had no known contamination. Prior to stormwater discharge, the PAC wetland had no known source of contamination and metal concentrations measured in the soil PAC were lower than average values reported for uncontaminated sites (1.7 ug/g Cd, 13 ug/g Cu, 23 ug/g Pb, 50 ug/g Zn; DeLaune et al. 1981).

Recent studies have focused on wetlands' ability to remove metals present in urban runoff. Sholes et al. (1999) investigated the removal of metals by constructed wetlands. They found that with high loading rates during storm events, the removal of metals was high; whereas removal was variable during dry weather. During storms, Scholes et al. (1999) found removal efficiencies for Zn 55%, Pb 62%, Cu 85%, Ni 77% and Cr 63%. Walker and Hurl (2002) found that the metals concentrations decreased as stormwater traveled through a wetland. Walker and Hurl (2002) were interested in the importance of sedimentation and chemical and biological processes in the removal of heavy metals. They found that the total amounts of metals decreased through the wetland, but the relative concentration varied with distance. Zinc, lead, and copper concentrations all decreased with distance from the inlet, chromium remained constant, while arsenic concentrations increased with distance. The researchers concluded that processes other than sedimentation were responsible for metal removal. Based on these studies, after stormwater discharge began, we expected removal of heavy metals by the sediment at the sites closest to the discharge site for the majority of the metals measured.

Post-Discharge

After stormwater discharge, mean concentrations were below detection limits (<1.0 ug/g) in sediments of study sites for many of the metals analyzed (Table 4). Only one core was collected at each study site so no replicates were available and, thus, statistics could not be conducted. However, post-discharge metals concentrations were still well below the PECs reported in Table 3 and pre- and post-discharge concentrations were very similar. These data indicate that the stormwater runoff was not a source of metals contamination within the study area during the period of measurement.

Table 4. Mean soil nutrient and metal concentrations (ug/g) in soils at Pointe aux Chenes Wildlife Management Area after stormwater input.

Analyte	Near	Mid	Far	Reference
Copper	1.49	1.67	0.59	1.70
Zinc	8.68	8.09	6.48	10.90
Cadmium	BDL*	BDL	BDL	BDL
Lead	BDL	BDL	BDL	BDL
Chromium	BDL	BDL	BDL	BDL
Nickel	BDL	BDL	BDL	BDL
Iron	NM**	NM	NM	NM
Manganese	NM	NM	NM	NM
Calcium	5472.38	2000.00	2459.84	2713.79
Magnesium	2800.00	2371.13	2475.59	2856.90
Phosphorus	60.72	25.90	63.53	35.76
Sodium	6935.24	5690.72	6075.59	6308.62

*Below detection limit of 1.0 ug/g.
**Not measured.

Water Quality

Salinity

Pre-Discharge

During the pre-discharge sampling period mean salinity was 10.3±1.2, 10.4±1.1, 10.5±1.0, 9.9±1.2, and 10.5±1.7 at the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 7). No significant differences were detected among sites for this sampling period (P = 0.9962).

There were no distinct spatial patterns in salinity but there were strong temporal changes associated with tides and climatic factors (frontal passages and tropical storms). Increased precipitation, wind and upland runoff related with the passage of Tropical Storms Isadore and Lilli, decreased the salinity from 14ppt to 6ppt over the entire study area through October and November 2002. The impact of these storm events is representative of the freshening effect that stormwater discharge should have on the study area.

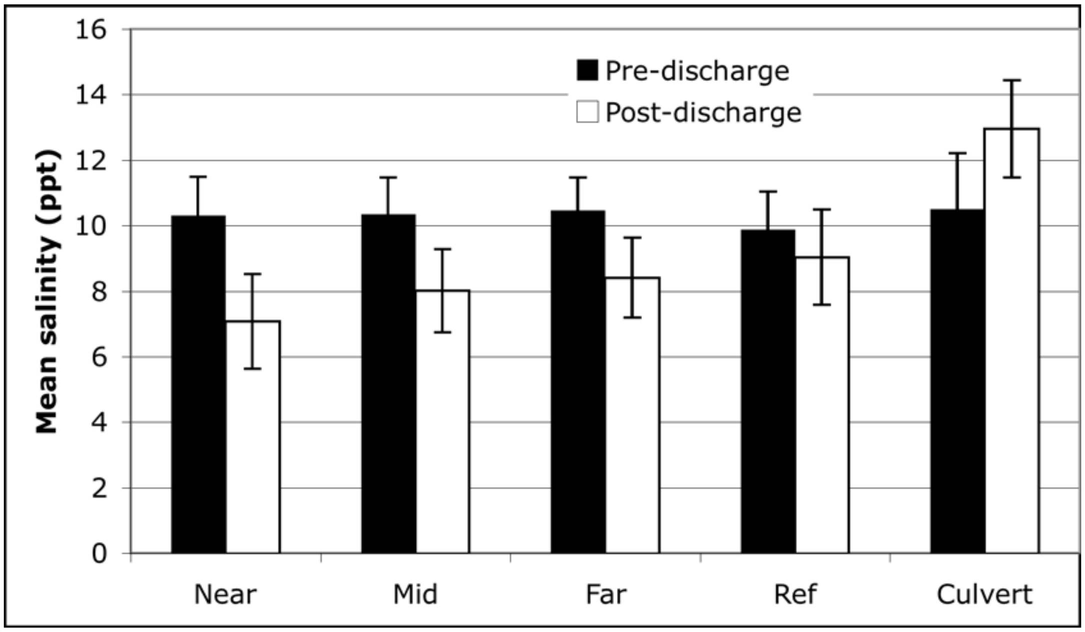


Figure 7. Mean salinities in each sampling site before and after stormwater discharge.

Post-Discharge

During the post-discharge monitoring period, mean salinity was 7.1±1.4, 8.0±1.3, 8.4±1.2, 9.0±1.5, and 13.0±1.5 ppt at the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 7). No significant differences in salinity were measured among the sites (P = 0.2739).

The first two sampling trips of 2006 (April and June 2006) showed unusually high salinities (>20 ppt) when compared to salinities measured during the rest of the post-discharge period (<10 ppt). Salinities were high during this time period compared with the rest of the post-discharge data. A discussion with the PAC WMA personnel showed that the pumps were not working during these months. When these two sampling periods were removed, mean salinities for the post-discharge period were 3.5±0.5, 4.9±0.5, 5.5±0.5, 5.4±0.4, and 11.3±1.1 for the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 8). When all sites were examined after discharge began, mean salinities in the Reference and Culvert sites were significantly higher than the other sites (P = 0.0038).

Within the treatment sites (Near, Mid, and Far), pre-discharge mean salinity (10.4±1.0) was significantly higher than mean post-discharge salinity (4.6±0.4; P = 0.0001). These data demonstrate that surface water salinity in the study area was lower following stormwater pumping. However, in addition to the discharge of freshwater into the area, the installation of a flap gate for salinity control also contributed to a reduction in surface water salinity. This flap gate was installed in the summer of 2005 in the culvert under the Isle de Jean Charles road and its purpose was to decrease the amount of saltwater moving into the wetland. There is no way to separate the impacts of the stormwater discharge and the flap gate on salinity control, and most likely both factors contributed to lower salinities.

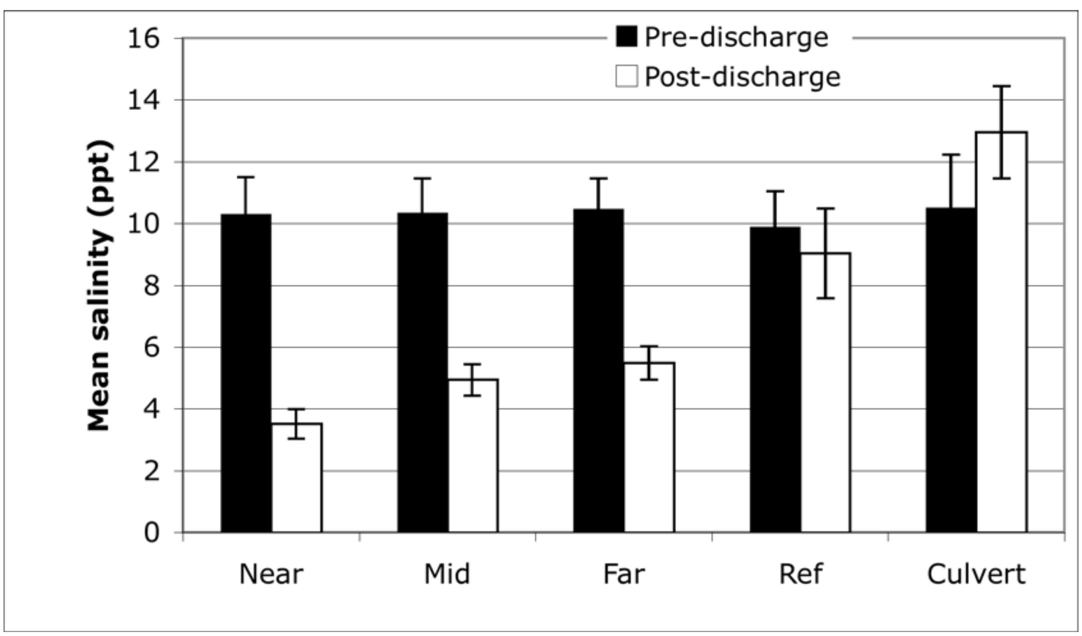


Figure 8. Mean salinities in surface water measured before and after stormwater discharge (Post-discharge data does not include April and June 2006 – see text).

Temperature

Pre-Discharge

Water temperature was measured throughout the study period. Water temperature followed a seasonal trend with higher temperatures in the summer months and cooler values during winter months. Mean temperatures were 28.1±0.6, 28.1±0.5, 27.7±0.5, 27.7±0.7, and 27.4±1.3°C for the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 9). There were no significant differences in temperature among sampling sites prior to stormwater discharge (P = 0.9399).

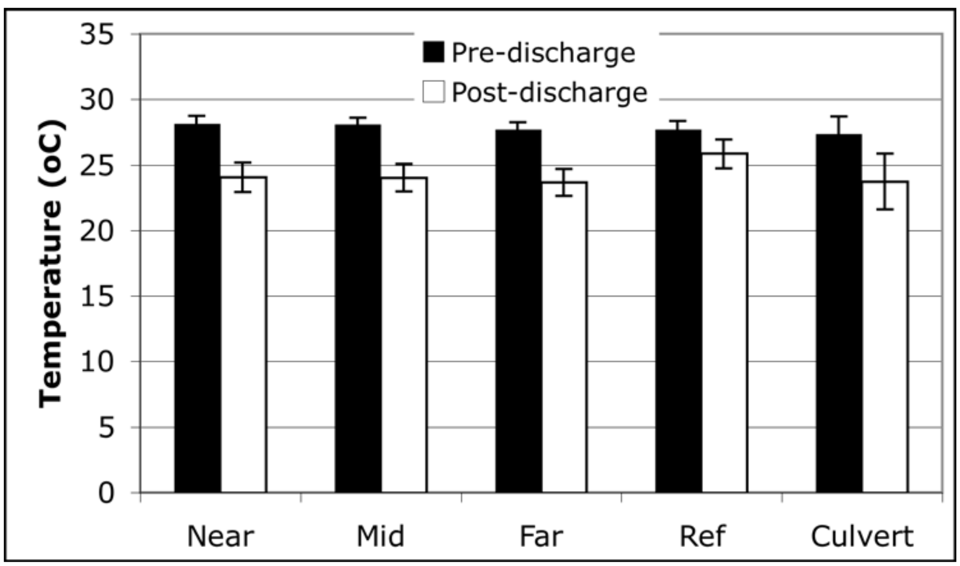


Figure 9. Mean temperature in surface water measured before and after stormwater discharge.

Post-Discharge

After stormwater discharge started, water temperature followed a seasonal trend with higher temperatures in the summer months and cooler values during winter months. Mean temperature was 24.0±1.1, 24.0±1.0, 23.7±1.0, 25.8±1.1, 23.7±2.1°C for the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 9). There were no significant differences in temperature among sampling sites after stormwater discharge (P = 0.6397). Among treatment sites, mean temperature was significantly lower after stormwater discharge (23.9±0.5) than before discharge (28.0±0.8; P = 0.0001). This effect is most likely from the input of cooler precipitation into warmer water in the wetland.

pH

Pre-Discharge

Prior to discharge, pH was consistently alkaline (>7.0) and averaged 7.7, representative of the marine influence in the area. The lowest average pH (7.1) occurred on October 8, 2002 after the two large storm events. Mean pH was 7.7±0.1, 7.8±0.1, 7.7±0.1, 7.7±0.1, 8.1±0.2 for the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 10). There were no significant differences in pH among sampling sites prior to stormwater discharge (P = 0.0873).

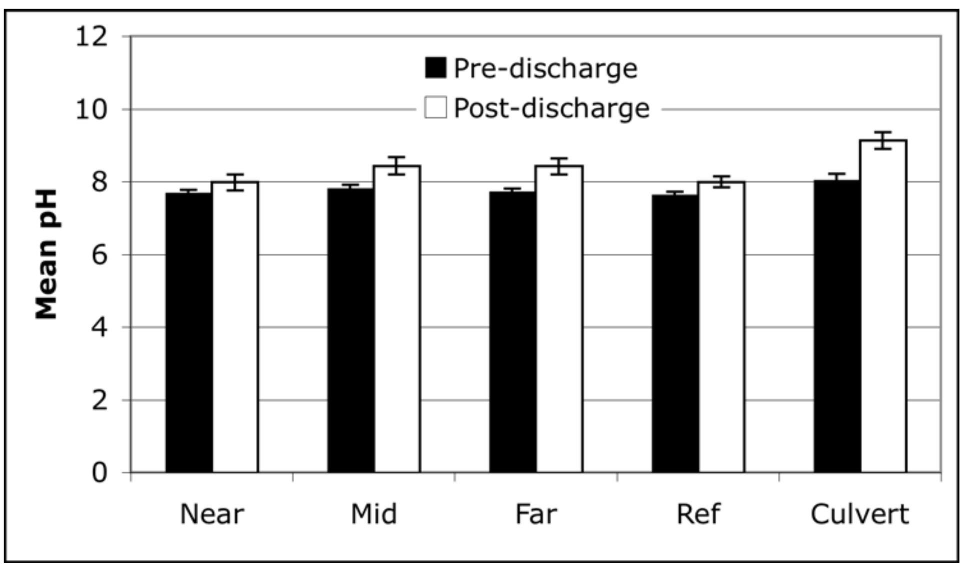


Figure 10. Mean pH in surface water measured before and after stormwater discharge.

Post-Discharge

After stormwater discharge began, mean pH was 8.0±0.2, 8.4±0.2, 8.4±0.2, 8.0±0.1, 9.1±0.2 for the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 10). There were no significant differences in pH among sampling sites after stormwater discharge (P = 0.0564). Among treatment sites, mean pH was slightly higher after discharge of stormwater (8.3±0.1) than before discharge (7.8±0.1, P = 0.0001). Ocean water is very well buffered due to high concentrations of bicarbonate and, thus, has a relatively narrow pH range (usually 7.5-8.4; Day et al. 1989). The pH of freshwater and low-salinity flooded waters can increase during daytime due to respiration of periphyton and plankton, breakdown of detritus, dissolution of carbonated materials, and flux of carbon dioxide (Reddy and DeLaune 2008).

Dissolved Oxygen

Pre-Discharge

Prior to stormwater discharge, mean D.O. concentrations were 4.6±0.4, 4.6±0.5, 4.1±0.5, 4.0±0.4, and 6.2±0.9 mg/L for the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 11). There were no significant differences in D.O. among sampling sites prior to stormwater discharge (P = 0.1016). D.O. was lowest after the October 2002 storm event when sediment was re-suspended and the high freshwater runoff led to an increase in oxygen demand. The culvert site had D.O. concentrations consistently greater than 3.5 mg/l.

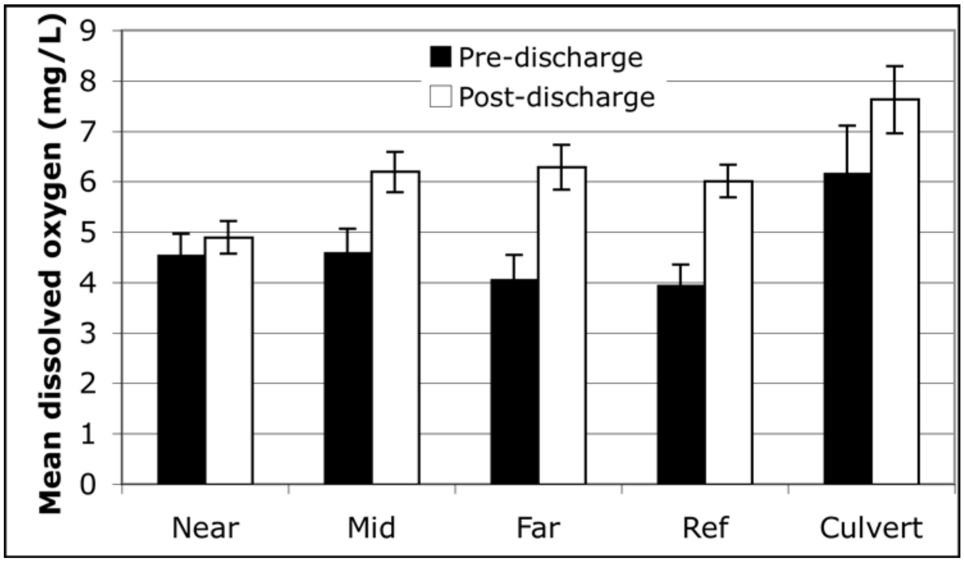


Figure 11. Mean dissolved oxygen concentrations in surface water at Pointe aux Chenes wetland before and after stormwater discharge.

Post-Discharge

Mean D.O. concentrations were 4.9±0.3, 6.2±0.4, 6.3±0.4, 6.0±0.3, 7.6±0.7 mg/L for the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 11). After stormwater discharge began, mean D.O. in surface water at the culvert site was significantly higher than that of the Near site (P = 0.0055). Among the treatment sites, mean D.O. was higher after stormwater pumping began (5.8±0.2) than before pumping (4.4±0.3, P = 0.0004). One hypothesis for the increased D.O. is aeration during pumping and the addition of precipitation (which usually has a higher dissolved oxygen than estuarine waters) into the wetland.

Fecal Coliform

Pre-Discharge

Mean fecal coliform bacteria (FCB) concentrations were 277.2±134.5, 191.6±91.2, 146.7±50.3, 90.4±26.9, 44.1±14.4 colonies/100 mL for the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 12). There were no significant differences in FCB among sampling sites prior to stormwater discharge (P = 0.4480).

FCB are the most common pollutant in rivers and streams and in Louisiana it has been reported that 37% of surveyed river miles, 31% of lakes, and 23% of estuarine water had some level of contamination (Hill et al. 2006). A prominent source of FCB is from upland run-off, therefore, analysis was completed to determine local FCB concentrations. Fecal coliform bacteria levels reached unacceptable levels after the passage of the tropical storms suggesting that the introduction of large quantities of upland runoff increases FCB values in the estuary. Levels reached >500 colonies/100ml after Tropical Storm Isadore (9/26/02) and Lili (10/2/02) and >300 colonies/100ml after Tropical Storm Bill (7/2/03). However, the presence of large numbers of birds in the area during the July sampling trip may have elevated FCB level during this period.

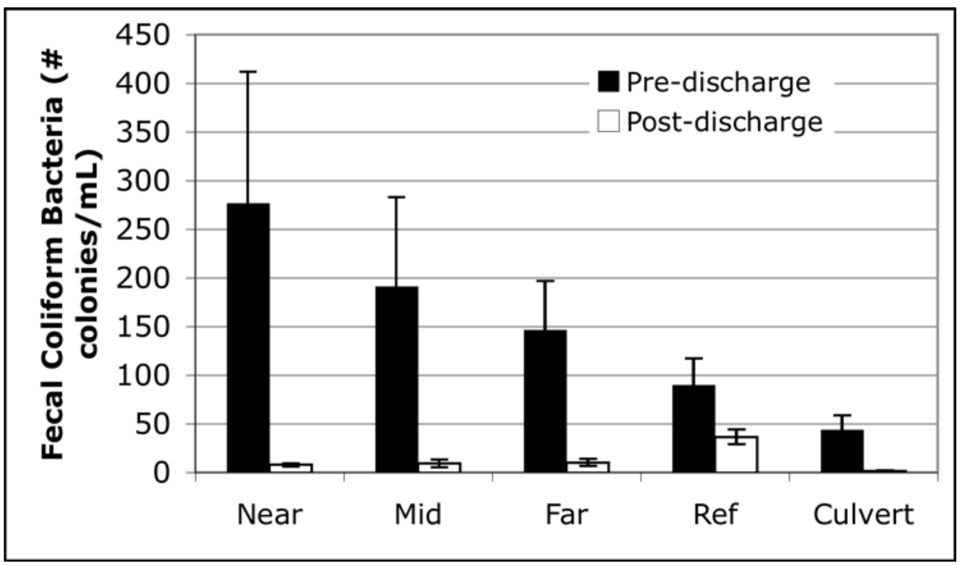


Figure 12. Mean fecal coliform bacteria in surface waters of the Pointe aux Chenes wetland before and after stormwater discharge.

Post-Discharge

Mean FCB concentrations were 7.7±1.5, 9.2±4.0, 10.2±3.6, 36.5±7.7, and 1.0±1.0 colonies/100 mL for the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 12).

FCB concentrations were significantly higher in surface waters of the Reference sites than in any of the other sites (P = 0.0007).

Among treatment sites, FCB concentrations were significantly higher prior to stormwater discharge (205.1±46.6) than after discharge (9.0±3.0; P = 0.0336), indicating dilution due to stormwater addition. Since FCB did not increase after introduction of stormwater runoff water into the wetland, these results support the hypothesis that birds utilizing the wetland were likely contributing to the high FCB concentrations. It appears that the stormwater is actually diluting FCB concentrations in surface waters, improving water quality.

Total Suspended Sediments

Pre-Discharge

Total suspended sediments (TSS) in the study area prior to discharge were generally low with values less than 50 mg/l (Figure 13). A change in suspended sediments was not seen after the passage of tropical storms or during the placement of dredged material on the levee surrounding the study area. There were likely short term increases in TSS during the storm passage but these were not captured by our sampling. Prior to stormwater discharge, no difference in TSS concentrations was measured among sites (P = 0.2080).

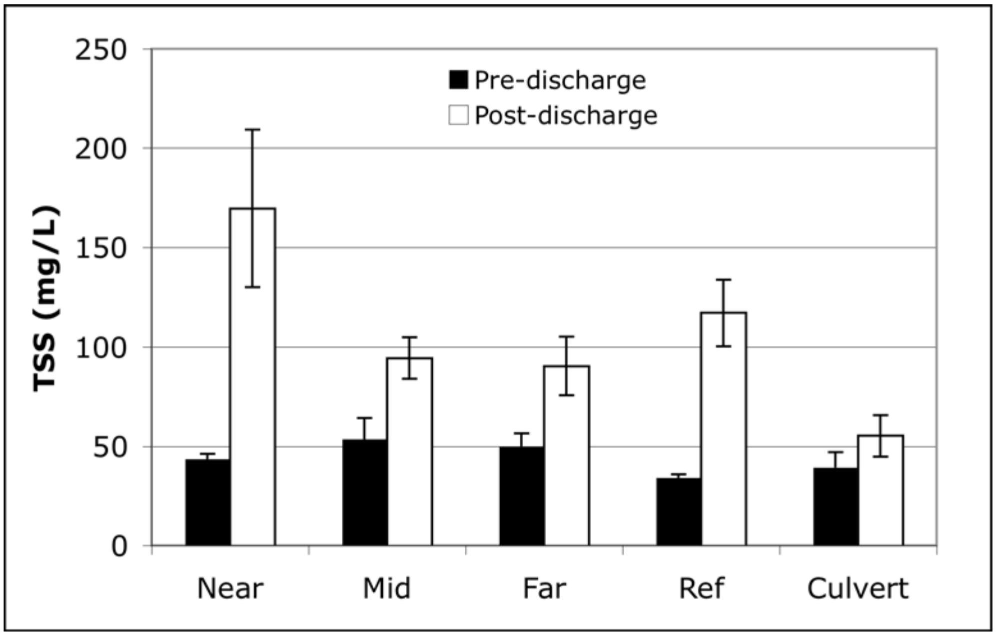


Figure 13. Total suspended solids in study sites before and after stormwater discharge.

Post-Discharge

After stormwater discharge began, total suspended sediments increased, ranging from a mean of 55±11 mg/L at the Culvert to 170±40 mg/L at the Near site (Figure 13). There were no significant differences among study sites due to the high variability ($P = 0.4500$). Among treatment sites, mean TSS concentrations were significantly higher after discharge (117.42±11.95 mg/L) than before discharge (49.28±16.62 mg/L; $P = 0.0011$). TSS not only increased in the sites receiving stormwater, but also in the Reference site. These data indicate that, although stormwater might be increasing TSS concentrations in the treatment areas, storm passages and water movement probably re-suspend sediments as well.

Chlorophyll a

Pre-Discharge

Mean chlorophyll *a* concentrations were 64.9±7.7, 56.6±9.2, 69.4±11.2, 51.8±4.5, and 29.7±9.6 ug/L in surface waters of the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 14). Mean chlorophyll *a* concentrations were not significantly different among sites (P = 0.1178).

Chlorophyll *a* concentration is a measure of algal cells in the water column. Many chlorophyll values in Pointe aux Chenes were over 20 ug/L and averaged between 30 and 70 ug/L, dependent upon site (Figure 14). The highest values occurred during low water periods, suggesting the re-suspension of benthic algae into the water column. The high values (>120 ug/L) during August 2003 may be related to an isolated algal bloom and high summer water temperatures. Because the area received low amounts of upland runoff prior to pumping, it is unlikely that upland sources of nutrients caused the elevated chlorophyll levels. The high population of birds and fecal matter from the birds is a likely source of nutrients.

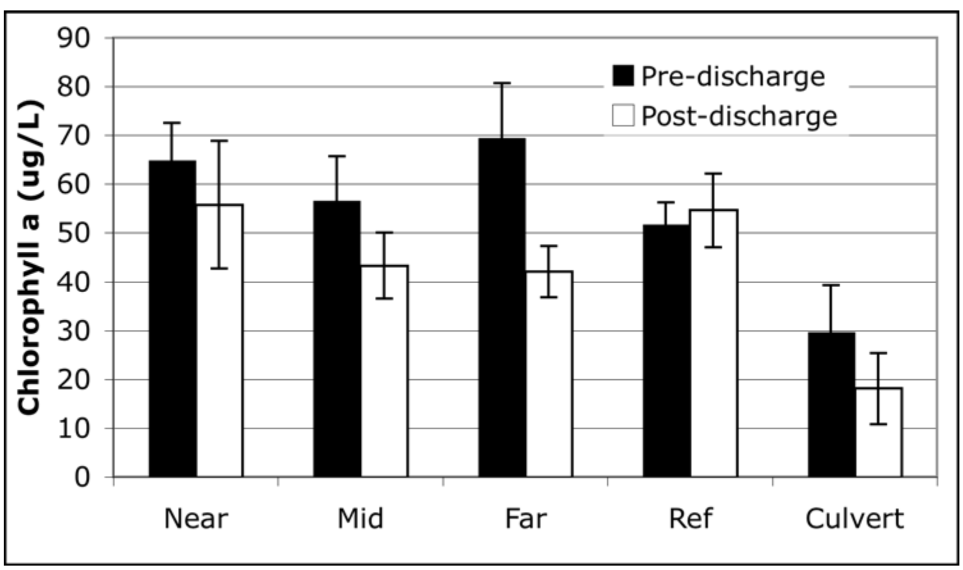


Figure 14. Mean chlorophyll a concentrations in surface water at the Pointe aux Chenes wetland.

Post-Discharge

Mean chlorophyll *a* concentrations were 55.8±13.1, 43.3±6.7, 42.1±5.2, 54.6±7.6, and 18.1±7.3 ug/L in surface waters of the Near, Mid, Far, Reference, and Culvert sites, respectively (Figure 14). There was no difference in Chlorophyll *a* concentrations among the study sites (P = 0.1594).

Among treatment sites, mean chlorophyll *a* concentrations were significantly higher prior to stormwater discharge (64.0±6.4) than after discharge (46.9±4.8; P = 0.0597). These data indicate that chlorophyll levels were reduced through dilution and flushing by stormwater.

Nitrogen

Pre-Discharge

Surface water nitrogen (N) concentrations in the Pointe aux Chenes wetland were characterized by a high proportion of organic N and dissolved inorganic nitrogen (DIN) dominated by ammonium (Table 5). Nitrate+nitrite (NO_x) concentrations were very low throughout the pre-discharge study period, with values consistently below 0.04 mg/l and averaging 0.02 mg/l. These data are lower than average values in the Mississippi River (1.60 mg/L), Fourleague Bay (0.16 mg/L), and Breton Sound, La. (0.44 mg/L; Justic et al 1995; Perez et al 2003; Lane et al. 1999, 2004). Low NO_x concentrations are likely due to low freshwater input. High NO_x levels in a coastal system are generally due to freshwater input. Low concentrations in a system like Pointe aux Chenes are likely due to nitrification of part of the large ammonium pool. There were no significant differences among sites in NO_x concentrations (P = 0.9805).

Table 5. Nitrogen concentrations in sampling sites both before and after stormwater discharge in the Pointe aux Chenes Wetland.

Site	Concentration (mg/L)		
	Nitrate+Nitrite	**Ammonium**	**Total Nitrogen**
Pre-Discharge			
Near	0.02±0.01	0.37±0.10	1.54±0.08
Mid	0.02±0.01	0.38±0.08	1.43±0.08
Far	0.02±0.01	0.34±0.11	1.50±0.16
Reference	0.02±0.01	0.49±0.15	1.81±0.17
Culvert	0.02±0.01	0.26±0.11	1.18±0.23
Post-Discharge			
Near	0.10±0.03	0.13±0.02	1.85±0.14
Mid	0.11±0.03	0.13±0.02	1.82±0.09
Far	0.07±0.02	0.10±0.01	1.82±0.08
Reference	0.02±0.01	0.10±0.01	2.21±0.13
Culvert	0.05±0.03	0.06±0.01	0.87±0.16

Mean ammonium (NH_4) concentrations were much greater than mean NO_X concentrations and were the primary form of DIN (Table 4). Perez et al (2003) reported an average NH_4 value of 0.097 mg/l in Fourleague Bay, Louisiana. Average NH_4 values in the Mississippi River and Breton Sound are 0.06 and 0.010 mg/L, respectively (Lane et al. 1999, 2001, 2004). Prior to stormwater discharge, NH_4 averaged 0.38 mg/l in Pointe aux Chenes, substantially greater than Fourleague Bay, Breton Sound, and the Mississippi River. Biological assimilation and uptake of nitrogen is likely responsible for the low values during the summer months in 2003. Overall, DIN concentrations were above average for Louisiana coastal and wetland systems. There were no significant differences in NH_4 concentrations among sites prior to stormwater discharge ($P = 0.8118$).

Organic nitrogen comprised the majority of total nitrogen (TN) in the Pointe aux Chenes wetland. TN concentrations averaged between 1.18 to 1.81 mg/L (Table 5) and were not significantly different among sites ($P = 0.1330$). There were no clear spatial or temporal patterns in TN concentrations. Fourleauge Bay and Breton Sound TN values averaged 0.51 and 1.19

mg/L, respectively (Perez et al 2003; unpublished data), which were lower than those in the Pointe aux Chenes wetland.

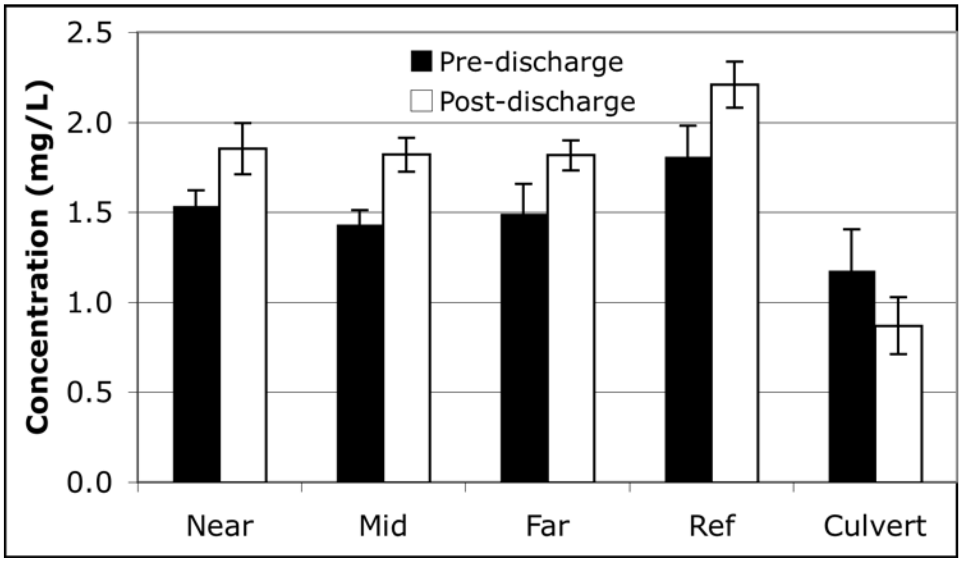


Figure 15. Total nitrogen concentrations in surface waters of Pointe aux Chenes wetland before and after stormwater discharge.

Post-Discharge

After stormwater discharge, mean NO_x concentrations in surface water ranged from 0.02 mg/L at the Reference site to 0.11 mg/L at the Mid site (Table 5), and there were no differences in concentration among sites (P = 0.0560). Among the treatment sites, mean NO_x concentration was significantly higher after stormwater discharge (0.065±0.009) than before discharge (0.020±0.012; P = 0.0036).

Mean NH_4 concentrations ranged from 0.06 mg/l at the Culvert to 0.13 mg/L at the Near and Mid sites (Table 5). There were no significant differences in concentrations among sites (P = 0.2252). When data were examined only from the treatment sites, NH_4 concentrations were higher before stormwater discharge (0.36±0.03) than after discharge (0.11±0.02; P = 0.0001).

Mean TN concentrations ranged from 0.87 mg/L at the Culvert to 2.21 mg/L at the Reference site (Table 5). TN concentrations in surface waters of the Near, Mid, Far, and

Reference sites were significantly higher than that in the Culvert site (P = 0.0001). When only data from the treatment sites were examined, TN concentrations were significantly higher after discharge (1.83±0.06) than before (1.49±0.08; P = 0.0007).

An increase in nitrogen concentrations after stormwater discharge shows that stormwater is adding nitrogen to the system. However, concentrations of the different nitrogen species are not very high and should not lead to eutrophication. Schueler et al. (2007) reviewed data from 40 emergent stormwater wetlands and showed that up to 55% of total nitrogen was removed.

Phosphorus

Pre-Discharge

Mean ortho-phosphate (PO_4) concentrations ranged from 0.07 mg/L in the Culvert site to 0.29 mg/L at the Reference site (Table 6). Overall values were higher in the Reference plots and in late summer, likely due to benthic re-mineralization during the warm summer months. Average PO_4 concentrations in Fourleague Bay, the Mississippi River and Breton Sound were 0.018, 0.24, and 0.041 mg/L, respectively (Perez et al. 2003; Justic et al 1995; Lane et al. 1999, 2004) and were similar to concentrations measured at the PAC wetland.

Table 6. Phosphorus concentrations in sampling sites before and after stormwater discharge into the Pointe aux Chenes Wetland.

Site	Concentration (mg/L)	
	Ortho-Phosphate	**Total Phosphorus**
Pre-Discharge		
Near	0.12±0.03	0.59±0.10
Mid	0.08±0.02	0.38±0.07
Far	0.10±0.03	0.39±0.08
Reference	0.29±0.06	0.56±0.09
Culvert	0.07±0.03	0.21±0.07
Post-Discharge		
Near	0.07±0.01	0.30±0.05
Mid	0.08±0.01	0.27±0.04
Far	0.08±0.01	0.26±0.04
Reference	0.04±0.01	0.29±0.05
Culvert	0.03±0.01	0.14±0.06

Mean total phosphorus (TP) concentrations ranged from 0.21 mg/L at the culvert site to 0.59 mg/L at the Reference site (Table 6). TP increased following each of the large storm events and during the late summer months. The storm events caused sediment re-suspension which most likely mobilized sediment phosphorus. Inorganic phosphorus was measured as PO_4, and by subtracting PO_4 from TP, it is clear that the majority of TP was composed of organic phosphorus. Average TP concentrations in Fourleague Bay and Breton Sound (0.083 and 0.093 mg/L, respectively) were less than concentrations measured at sites in the PAC wetland (Perez et al 2003; unpublished data).

Post-Discharge

Following stormwater discharge, mean PO_4 concentrations ranged from 0.03 mg/L at the Culvert site to 0.08 mg/L at the Mid site (Table 6). Mean PO_4 concentration in surface water of the Far site (0.08 mg/L) was significantly higher than that in the Reference or Culvert sites (0.04 and 0.03, respectively; $P = 0.0129$). Total phosphorus (TP) concentrations ranged from 0.21 mg/L at the Culvert site to 0.59 mg/L at the Near site (Table 6). No differences in TP concentrations among sites were measured after discharge ($P = 0.3549$).

When mean PO_4 concentrations in the treatment sites were examined, concentrations were not significantly different due to stormwater discharge ($P = 0.1087$). Among treatment sites, however, mean TP concentrations were higher in surface waters prior to stormwater discharge (0.45±0.04 mg/L) than after discharge (0.28 ±0.03 mg/L; $P = 0.0001$; Figure 16). These data indicate that stormwater is not a significant source of phosphorus to the wetland. Although PO_4 typically follows the same trends as TSS because phosphorus tends to sorb onto sediment surfaces, most likely the TSS was primarily organic matter and, thus, would not promote the sorption and removal of PO_4 with settling of solids.

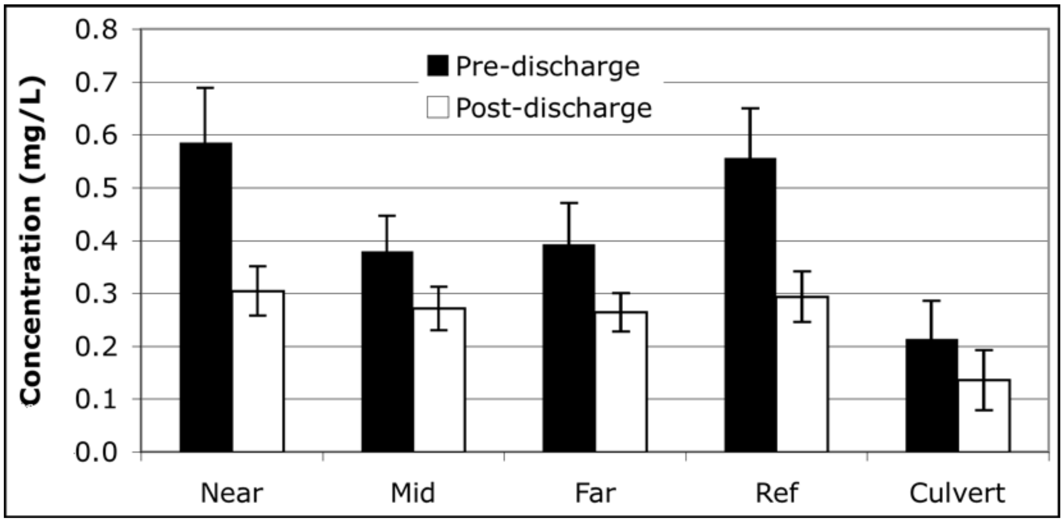


Figure 16. Mean total phosphorus concentrations in surface waters of wetland sites before and after stormwater discharge.

Silicate

Pre-Discharge

Mean silicate (SiO_3) concentrations ranged from 3.05 mg/L at the Culvert to 5.86 mg/L at the Reference site (Figure 17). By comparison, concentrations in Breton Sound averaged 2.05 mg/L (unpublished data). Similar to PO_4, SiO_3 concentrations were the highest in the warm summer months likely due to regeneration. SiO_3 also increased after the passage of tropical storms Isadore and Lilli. The high concentrations in surface waters may be partially due to the dissolution of diatoms. Prior to stormwater discharge, mean silicate concentrations were not significantly different among wetland sites ($P = 0.3570$).

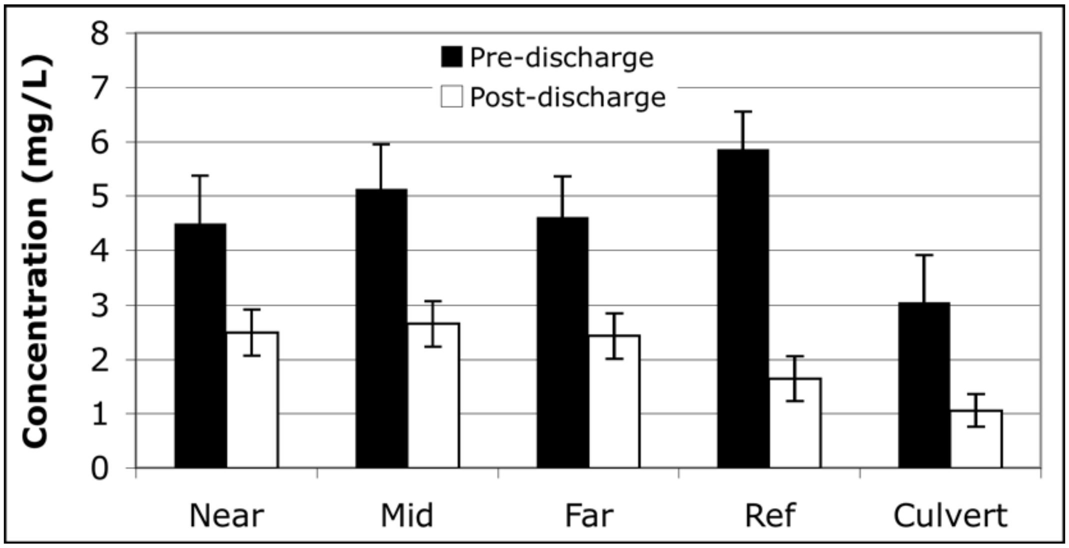


Figure 17. Mean silicate concentrations in surface water of Pointe aux Chenes wetland study sites before and after stormwater discharge.

Post-Discharge

Mean silicate concentrations ranged from 2.65 mg/L at the Mid site to 1.06 mg/L at the Culvert (Figure 17). No significant differences were measured among study sites after stormwater discharge ($P = 0.1742$). When only treatment sites were examined, mean silicate concentration was significantly higher pre- discharge (4.75±0.39 mg/L) than post- discharge (2.52±0.27 mg/L; $P = 0.0001$). These data indicate that the freshwater is diluting silicate concentrations in surface waters.

Stoichiometric Ratios

Pre-Discharge

Using the atomic Si:N:P ratio of 16:16:1 (Redfield 1958) as a criterion for balanced nutrient composition with respect to phytoplankton uptake, deviations from this ratio can be used to indicate *potential* nutrient limitation. An N:P ratio > 16:1 indicates P limitation while a ratio < 16:1 indicates N limitation. Likewise, a DSi:N ratio < 1:1 indicates Si limitation while a ratio

> 1:1 indicates N limitation. Dissolved (D)Si:N ratios greater than 1:1 favor diatoms (Conley et al. 1993).

Mean N:P ranged from 5.00±1.19 at the Reference site to 14.33± 6.23 at the Culvert site, indicating nitrogen limitation throughout the wetland (Figure 18). No difference was measured in N:P ratios among sites (P = 0.0887). Mean DSi:P ratio ranged from 14.81±3.39 at the Near site to 25.46±7.27 at the Culvert site, indicating P limitation with respect to Si (Figure 19). No difference was measured in DSi:P ratios among sites (P = 0.2162). Mean DSi:N ranged from 2.85±0.47 at the Near site to 3.57±0.54 at the Reference site, indicating N limitation with respect to Si (Figure 20). No difference was measured in DSi:N ratios among sites (P = 0.8232). These results show that DSi is in excess and this should favor diatom growth. Overall, the results indicate that nitrogen is the limiting nutrient for phytoplankton growth.

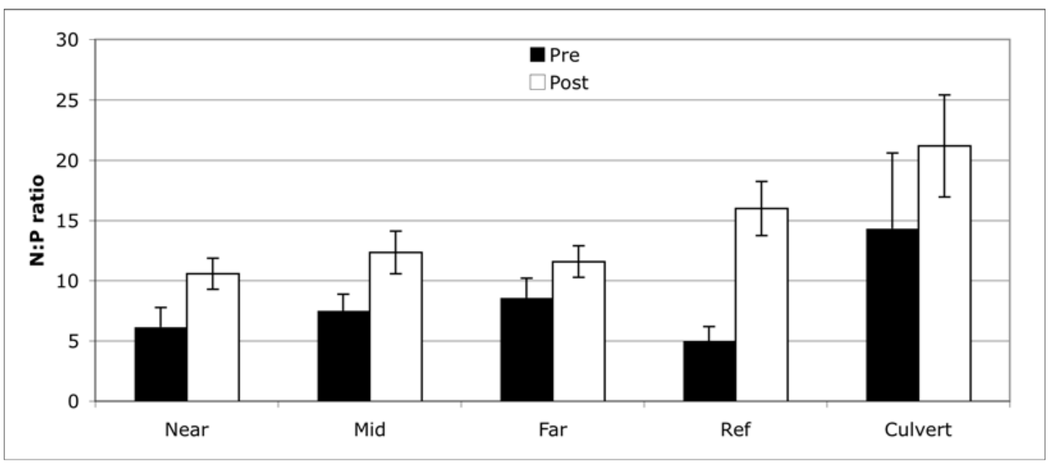


Figure 18. Nitrogen to phosphorus ratios in study sites before and after stormwater discharge.

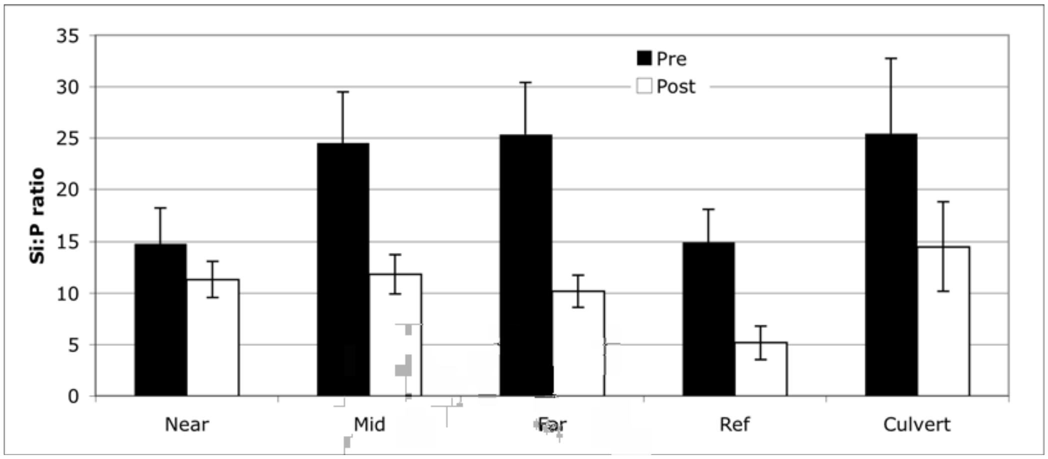


Figure 19. Dissolved silicate to phosphorus ratios in study sites before and after stormwater discharge.

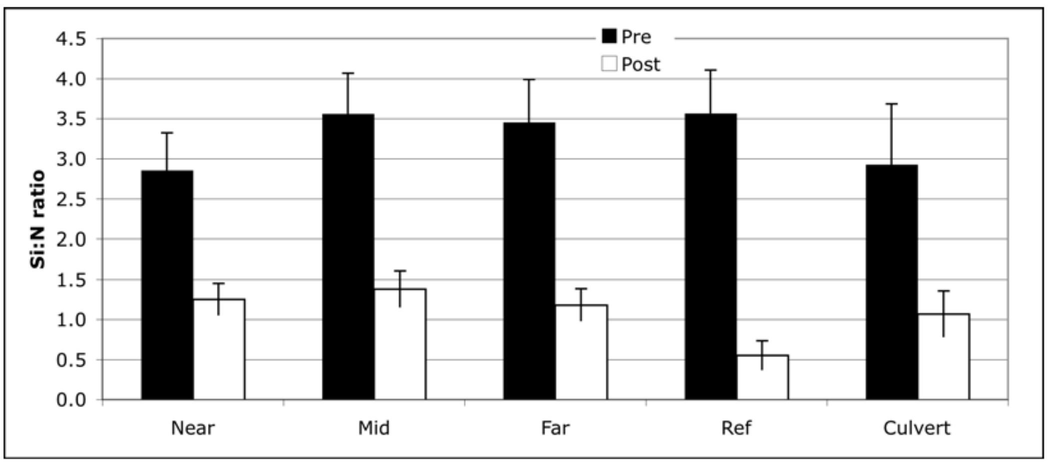


Figure 20. Dissolved silicate to nitrogen ratios in study sites before and after stormwater discharge.

Post-Discharge

Following stormwater discharge, mean N:P ratios ranged from 10.56±1.29 at the Near site to 21.16±4.23 at the Culvert site (Figure 18). Mean N:P ratios in surface water measured at the Culvert were significantly higher than those in the Near site ($P = 0.0149$). Among treatment sites, mean N:P ratios in surface waters following stormwater discharge were significantly higher than ratios reported prior to discharge ($P = 0.0027$). With an increase in total nitrogen to the wetland, mean N:P ratios are still below 16:1, but with additional nitrogen coming into the system, these ratios may eventually increase.

Following stormwater discharge, mean DS:P ratios ranged from 5.16±1.62 at the Reference site to 14.47±4.32 at the Culvert site (Figure 19). No significant differences in mean DS:P ratios were found ($P = 0.3049$). Among treatment sites, mean DS:P ratios in surface waters prior to stormwater discharge were significantly higher than ratios reported after discharge ($P = 0.0001$). Silicate concentrations were lower after stormwater discharge and this is most likely why mean DS:P ratios decreased. Relative to silicate, stormwater has decreased phosphorus limitations.

Following stormwater discharge, mean DS:N ratios ranged from 0.55±0.18 at the Reference site to 1.38±0.23 at the Culvert site (Figure 18). No significant differences in mean DS:N ratios were found ($P = 0.0511$). Among treatment sites, mean DS:N ratios in surface waters prior to stormwater discharge were significantly higher than ratios reported after discharge ($P = 0.0001$). Silicate concentrations were lower after stormwater discharge and this is most likely why mean DS:N ratios decreased. Relative to silicate, stormwater has decreased nitrogen limitations.

Water Level

Pre-Discharge

Water level variations in the study area reflected both astronomical tides as well as climatically driven fluctuations. The minimum and maximum water level ranges at stations 5 and 11 were 0.27 to 0.49 m and 0.28 to 0.50 m, respectively (Figure 21). The lower values are typical of the astronomical tide while the higher values are typical of wind-induced fluctuations. Water level fluctuations were similar in the Reference (PAC 11) and Mid sites (PAC 5).

Tropical Storm Bill led to an increase in water levels on June 30, 2003. Water levels increased by 1.2 m in a 24-hour period.

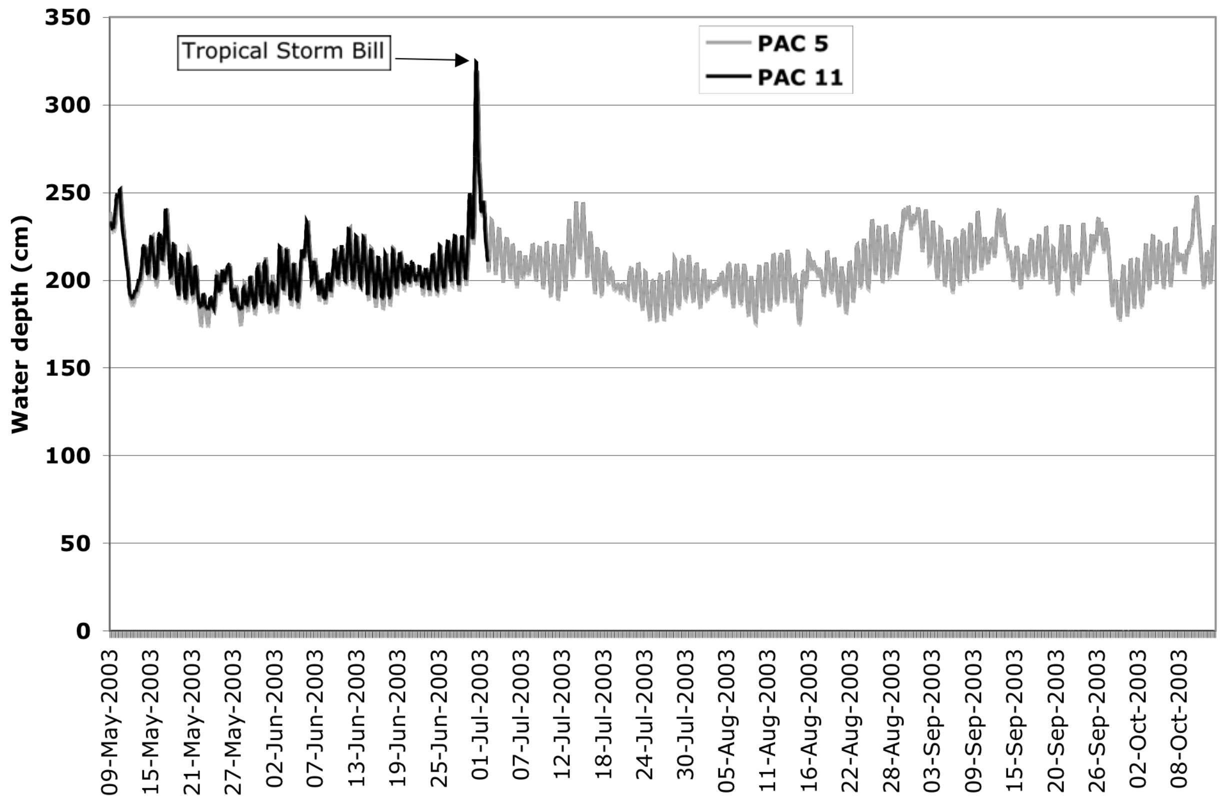


Figure 21. Surface water depths in the Pointe aux Chenes wetland in the Mid (Site 5) and Reference (Site 11) sites prior to stormwater discharge.

Post-Discharge

Water level gauges were destroyed during Hurricane Katrina. New gauges were not installed and so no post-discharge data exist. However, observations during field trips indicate that the stormwater discharge did not have significant impacts on the surface water depths within the PAC wetland. Water depths in the area are controlled by coastal water levels and there are no barriers to water flow from the pumps to the culvert where water flows out of the site.

Feldspar Marker Horizons

Pre-Discharge

Lack of sediment input to a marsh is a major cause of marsh deterioration in coastal Louisiana. However, at the PAC wetland prior to stormwater discharge, there were high accretion rates at all stations. Accretion rates ranged from 3.19 cm/yr at the Near site to 5.98 cm/yr at the Far site (Figure 22). Accretion at the Far site was significantly higher than all of the other sites ($P = 0.0022$). This is to be expected because, prior to pumped discharge, the main source of sediments was from the larger water bodies south of PAC.

Accretion rates were well above the value of about 1 cm/yr estimated as necessary to offset relative sea level rise (Baumann et al. 1984, Day et al. 2000). Rates were also much higher than accretion rates measured between Bayou Barre and Bayou Terrebonne north of Lake Barre using cesium dating (0.88 – 0.96 cm/yr, Nyman et al. 2006). The high values are likely a result of the deposition of sediments re-suspended during the passage of the two tropical storms that occurred during the year of pre-discharge data collection, as has been reported by others (Baumann et al. 1984, Michener et al 1997, Reed 1989).

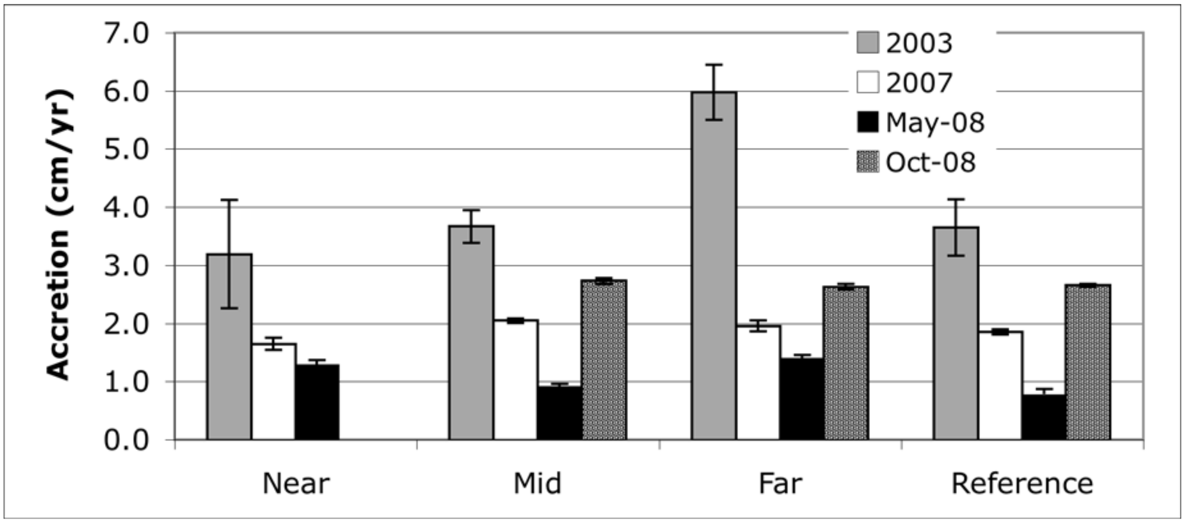


Figure 22. Accretion rates measured in sampling sites both before (2003) and after (2007 and 2008) stormwater discharge in the Pointe aux Chenes Wetland.

Post-Discharge

Mean accretion rates in 2007 (after stormwater discharge) ranged from 1.65 cm/yr in the Near site to 2.05 cm/yr in the Mid site (Figure 22). Accretion in the Mid and Far sites were significantly higher than that in the Near site (P = 0.0048). In May 2008, mean rates ranged from 0.79 cm/yr in the Reference site to 1.41 cm/yr in the Far site. Accretion in the Far and Near sites was significantly higher than accretion in the Mid and Reference sites (P = 0.0001). In October 2008, accretion rates were higher than those recorded in May of the same year. Mean rates ranged from 2.63 cm/yr in the Far site to 2.73 cm/yr in the Mid site. No differences were measured in accretion rates among the sites (P = 0.2657).

Within the treatment sites, mean accretion measured prior to stormwater discharge (4.62±0.20 cm/yr) was significantly higher than that measured in October 2008 (2.68±0.05 cm/yr; P = 0.0001). Pre-discharge accretion rates were most likely higher than post-discharge rates due to the passage of two tropical storms during the pre-discharge monitoring period. Hurricane Lili caused a breach in the levee system surrounding the study area, causing suspension of sediment and deposition within the area (Figure 23). Tropical Storm Bill also inundated the area with floodwaters and caused scouring of the levee at numerous locations. These sediments most likely contributed to accretion. Following the initiation of stormwater discharge, Hurricane Rita caused flooding of the area, with levee scouring and overtopping as well. As can be seen in Figure 22, accretion rates increased between May 2008 and October 2008, probably due to the impacts of Hurricanes Gustav and Ike. These data indicate that, within the PAC wetland for the period studied, storm related deposition is more important for accretionary dynamics than input of stormwater.

Figure 23. Location (left) and levee breach (right) during Hurricane Lili in October 2002.

Sediment Erosion Tables (SET)

Initial elevation measurements were taken at all SET sites on May 9, 2003 and then re-measured five years later on October 29, 2008. Mean sediment elevation change was +3.2, +2.4, and +2.6 cm/yr in the Near, Far, and Reference sites, respectively. Elevation increase was significantly higher in the Near site than in the Far and Reference sites ($P = 0.0001$). No elevation measurements were taken in the Mid site in 2008 because the base support pipe could not be located, due to burial during Hurricane Gustav, which made landfall near the Pointe aux

Chenes WMA on September 1, 2008. Mean elevation change was similar to accretion rates in the study area (Figure 23).

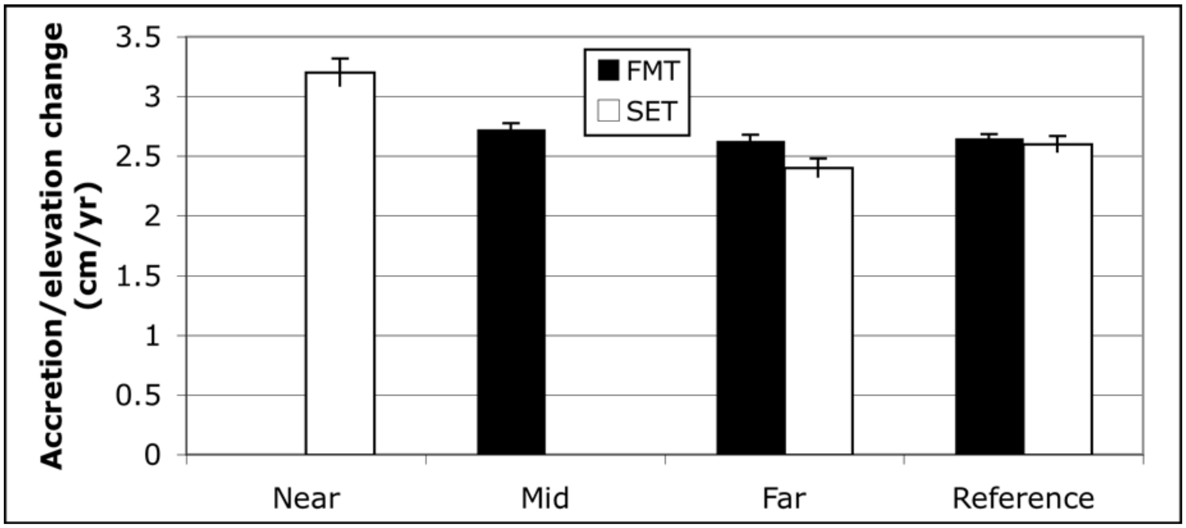


Figure 24. Mean accretion measured using the feldspar marker technique (FMT) compared to sediment elevation change measured using the sediment elevation table (SET).

Breaux and Day (1994) proposed a restoration strategy by hypothesizing that adding nutrient-rich, secondarily-treated municipal effluent to hydrologically isolated and subsiding wetlands could promote vertical accretion through increased organic matter production and deposition. This hypothesis could also apply to stormwater addition. Data from this study suggest that stormwater discharge is increasing wetland elevation, most likely through an increase in sediment accretion and an increase in organic matter deposition as proposed by Breaux and Day (1994). However, because of the frequency of tropical storms and hurricane passage in this area, storm-related accretion was more important.

End-of-Season Live Biomass

Pre-Discharge

Prior to stormwater discharge, above ground end-of-season biomass averaged 1428 g/m^2 throughout the marsh. These data correspond to the high-end of productivity in an unimpacted coastal Louisiana brackish marsh in Terrebonne Parish (500 – 1600 $g/m^2/yr$, LaPeyre et al. 2009), and thus biomass measured in the PAC wetland was relatively high. Mean annual end-of-season live biomass ranged from 1178 g/m^2 at the Far site to 1678 g/m^2 at the Reference site (Figure 24). Marsh productivity in Lake Ponchartrain and Breton Sound were 2541-4411 $g/m^2/yr$ and 1342 $g/m^2/yr$, respectively (Cramer et al 1981; White et al 1978). There were no differences in annual end-of-season live biomass among sites prior to stormwater discharge ($P = 0.6494$).

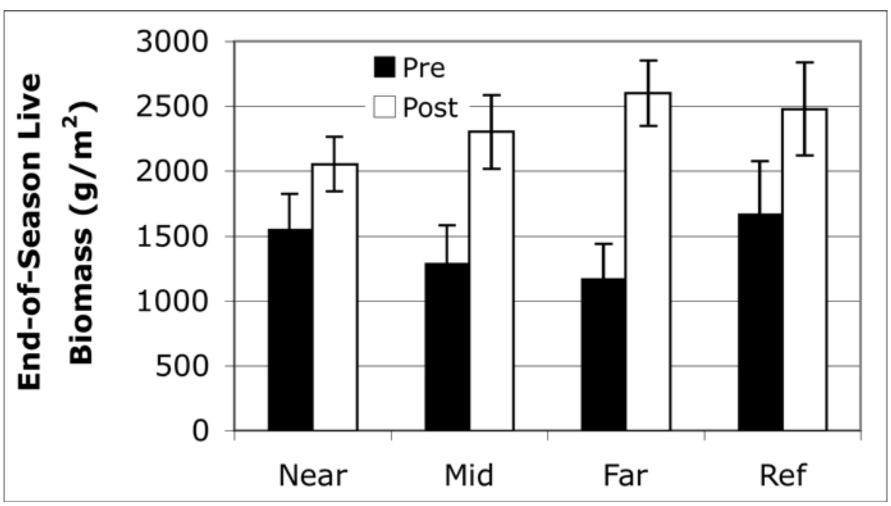


Figure 25. End-of-season live biomass measured in study sites before and after stormwater discharge.

Post-Discharge

After stormwater discharge, mean vegetation end-of-season live biomass ranged from 2053±211 $g/m^2/yr$ in the Near site to 2599±253 $g/m^2/yr$ in the Far site (Figure 24). There were no differences in productivity among sites ($P = 0.4539$). After stormwater discharge, mean marsh end-of-season live biomass in the treatment sites averaged 2256±140 g/m^2/yr, which was significantly higher than before discharge (1341±159 g/m^2/yr; P = 0.0001). In addition to increasing vegetation biomass, it is apparent from aerial photographs that areas of mudflat are also increasing, as there seems to be a significant increase in the land areas from aerial images from 2004 and 2007 (Figure 25). These areas may need to be re-vegetated if they are not colonized on their own.

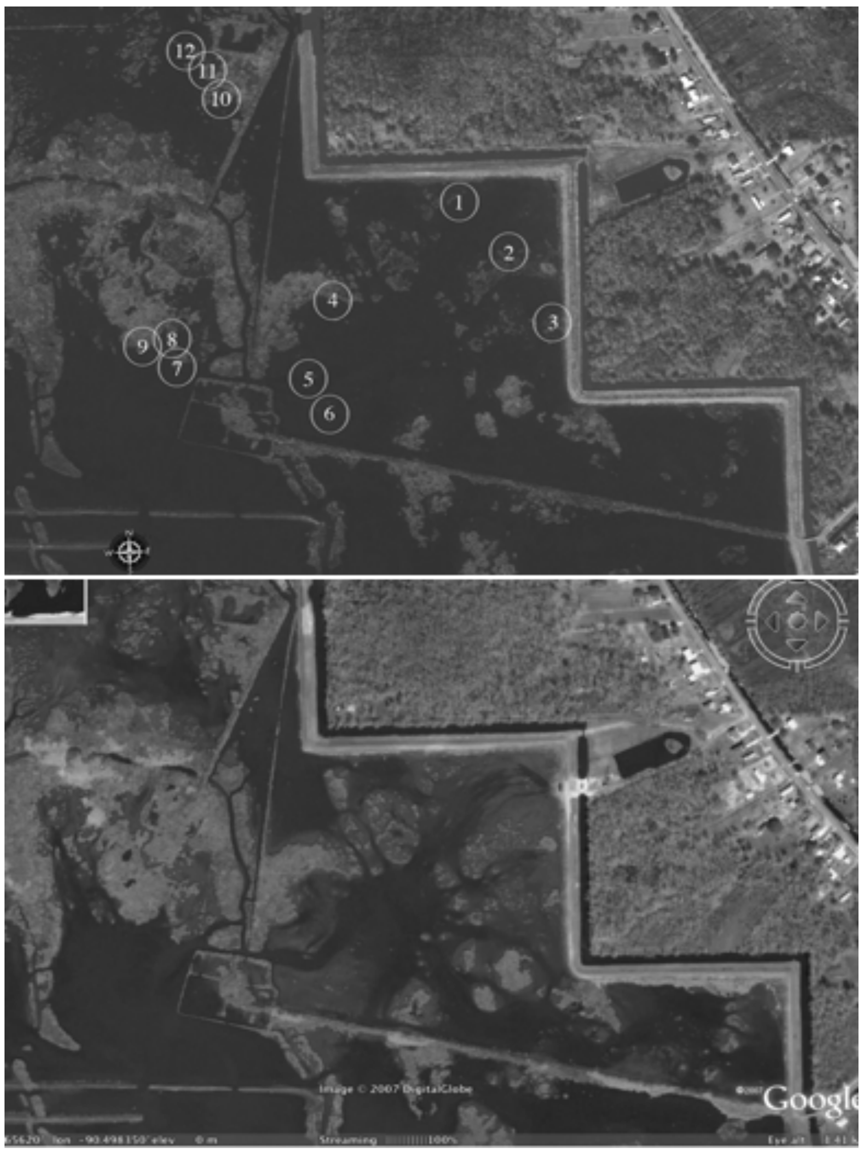


Figure 26. Digital orthophoto quarter quadrangle photo (top) taken in 2004 and an aerial photograph from Google Earth (bottom) taken in 2007 of Pointe aux Chenes Wildlife Management Area.

Conclusions

Stormwater discharge has had a positive impact on water quality of the Pointe aux Chenes wetlands, reducing salinity, fecal coliform bacteria, chlorophyll a concentrations, and total phosphorus concentrations. There was also increased dissolved oxygen, total suspended solids, and total nitrogen concentrations after discharge began. In addition, the data indicate that stormwater runoff is not a source of metals contamination.

Accretion measured with Feldspar markers and elevation change using the sediment elevation tables had contradictory results. Mean accretion measured prior to stormwater discharge was significantly higher than that measured after discharge, most likely due to the passage of two tropical storms during the pre-discharge monitoring period. These data indicate that, within the PAC wetland, passage of storms may have more of an influence on accretionary dynamics than input of stormwater. However, mean sediment elevation change was significantly higher in the Near site than in the Far and Reference sites, suggesting that stormwater discharge is causing an increase in the wetland elevation.

Vegetative biomass was increased from stormwater discharge as were mudflat areas, which is evident from aerial photographs taken in 2004 and 2007. These areas might need to be replanted if natural seeding does not occur. Plant species diversity should develop as the elevation is raised and freshwater and nutrients are added via pumping.

No evidence of negative impacts of stormwater discharge to the PAC marsh were found during this research but several positive impacts were noted. Based on these results, it is recommended that stormwater discharge be used as a restoration technique in other coastal parishes within the Barataria-Terrebonne National Estuary Program to nourish marshes and improve water quality. Wetlands adjacent to pumping stations should be considered a high priority for receiving stormwater discharge. Where possible, other discharges should be re-directed from direct flow into waterways and into wetlands instead. These additional projects should be monitored for nutrients and pumping data so that nitrogen and phosphorus loading rates may be calculated. Correlation of loading rates and positive/negative impacts will lead to improved management plans for these systems. In addition, long-term monitoring for accretion and biomass should be incorporated into any restoration plan using stormwater discharge. Dependent upon salinity, plantings of wetlands species should also be considered.

1. Do you anticipate that the use of stormwater redirection would have a significant impact on coastal marsh integrity and coastal water quality, especially for oyster harvesting?

References

Baldwin, A.; K. McKee and I. Mendelssohn. 1996. The Influence of Vegetation, Salinity, and Inundation on Seed Banks of Oligohaline Coastal Marshes. American Journal of Botany 83(4):470-479.

Baldwin, A. and I. Mendelssohn. 1998. Effects of salinity and water level on coastal marshes: an experimental test of disturbance as a catalyst for vegetation change. Aquatic Botany 61: 255-268.

Baumann R. H., Day Jr J. D., and Miller C. A. (1984) Mississippi deltaic wetland survival: sedimentation versus coastal submergence. Science 224:1093-1095.

Breaux, A.M. and J.W. Day, Jr. 1994. Policy Considerations for Wetland Wastewater Treatment in the Coastal Zone: A Case Study for Louisiana. Coastal Management. 22:285-307.

Boesch, D.F., M.N. Josselyn, A.J. Mehta, J.T. Morris, W.K. Nuttle, C.A. Simenstad, and D.J.P. Swift. 1994. Scientific Assessment of Coastal Wetland Loss, Restoration and Management in Louisiana. Journal of Coastal Research Special Issue No.20. 103p.

Boumans, R., and J.W. Day, Jr. 1993. High precision measurements of sediment elevation in shallow coastal areas using a sedimentation-erosion table. Estuaries 16:375-380.

Buresh, R.J. and W.H. Patrick. 1981. Nitrate Reduction to Ammonium and Organic Nitrogen in an Estuarine Sediment. Soil Biology and Biochemistry 13:279-283.

Burnison, B.K. 1980. Modified dimethyl sulfoxide (DMSO) extraction for chlorophyll analysis of phytoplankton. Canadian Journal of Fisheries and Aquatic Sciences 37:729-733.

Conley, D.J., C.L. Schlexke, and E.F. Stoermer. 1993. Modification of the biogeochemical cycle of silica with eutrophication. Marine Ecology Progress Series 101:179-192.

Cramer, G.W., J.W. Day, and W.H.Conner. 1981. Productivity of four marsh sites surrounding Lake Ponchartrain, Louisiana. The American Midland Naturalist 106:65-72.

Day, J.W., Jr., D.F. Boesch, E.J. Clairain, G.P. Kemp, S.B. Laska, W.J. Mitsch, K. Orth, H. Mashriqui, D.J. Reed, L. Shabman, C.A. Simenstad, B.J. Streever, R.R. Twilley, C.C. Watson, J.T. Wells, and D.F. Whigham. 2007. Restoration of the Mississippi Delta: Lessons from Hurricanes Katrina and Rita. Science 315:1679-1684.

Day,J.W., L.D. Britsch, S. Hawes, G. Shaffer, D.J. Reed, and D. Cahoon. 2000. Pattern and process of land loss in the Mississippi Delta: a spatial and temporal analysis of wetland habitat change. Extuaries 23:425-438.

Day, J.W., Jr., C.A.S. Hall, W.M. Kemp, and A. Yanez-Arancibia. 1989. Estuarine Ecology. John Wiley and Sons, Inc., New York, New York.

DeLaune, R.D. and R.P. Gambrell. 1996. Role of Sedimentation in Isolating Metal Contaminants in Wetland Environments. Journal of Environmental Science Health A31(9):2349-2362.

DeLaune, R.D., C.N. Reddy, and W.H. Patrick. 1981. Accumulation of Plant Nutrients and Heavy Metals through Sedimentation Processes and Accretion in a Louisiana Salt Marsh. Estuaries 4(4):328-334.

Greenberg, A.E., R.R. Trussell, L.S. Clesceri, M.A.H. Franson, eds. 1985. Standard Methods for the examination of water and wastewater. American Public Health Association. Washington, D.C.

Hill, D.D., W.E. Owens, and P.B. Tchounwou. 2006. The impact of rainfall on fecal coliform bacteria in Bayou Dorcheat (North Louisiana). J. Environ. Res. Public Health 3:114-117.

Justic, D., N. Rabalais, R.E. Turner and Q. Dortch. 1995. Changes in nutrient structure of River-dominated coastal waters; stoichiometric nutrient balance and its consequences. Estuarine, Coastal and Shelf Science 40:339-356.

Lachat Instruments, Inc. 1999. QuikChem Methods.

Lane, R.; J. Day; and B. Thibodeaux. 1999. Water Quality Analysis of a Freshwater Diversion at Caernarvon, Louisianan. Estuaries 22(2A): 327-336.

Lane, R., J.W. Day, Jr., G.P. Kemp, and D.K. Demcheck. 2001. The 1994 experimental opening of the Bonnet Carre Spillway to divert Mississippi River water into Lake Pontchartrain, Louisiana. Ecological Engineering 17: 411-422.

Lane, R.R., J.W. Day, D. Justic, E. Reyes, J. Day, and E. Hyfield. In press. Changes in stoichiometric Si, N, and P ratios of Mississippi River water diverted through coastal wetlands to the Gulf of Mexico. Estuarine, Coastal and Shelf Science.

LaPeyre, M.K., B. Gossman, and B.P. Piazza. 2009. Short- and long-term response of deteriorating brackish marshes and open-water ponds to sediment enhancement by thin-layer dredge disposal. Estuaries and Coasts 32:390-402.

Leck, M.; C.T. Parker; R. Simpson. 1989. Ecology of Soil Seed Banks. San Diego, California, USA, Academic Press.

Michener, W.K., E.R. Blood, K.L. Bildstein, M.M.Brinson and L.R. Gardner. 1997. Climate change, hurricanes and tropical storms, and rising sea level in coastal wetlands. Ecological Applications 7:770-801.

Nyman, J.A., R.J. Walters, R.D. DeLaune, and W.H. Patrick, Jr. 2006. Marsh vertical accretion via vegetative growth. Estuarine, Coastal and Shelf Science 69:370-380.

Pardue, J.H., R.D. DeLaune, C.J. Smith, and W.H. Patrick, Jr. 1988. Heavy Metal Concentrations along the Louisiana Coastal Zone. Environmental International 14: 403-406.

Perez, B.C., J.W. Day, D. Justic and R. Twilley. 2003. Nitrogen and phosphorus transport between Fourleague Bay, La and the Gulf of Mexico: the role of winter cold fronts and Atchafalaya River Dischage. Estuarine, Coastal and Shelf Science 57:1065-1078.

Reddy, K.R. and R.D. DeLaune. 2008. Biogeochemistry of wetlands: Science and applications. CRC Press, Boca Raton, Florida.

Reddy, K.R., W.H. Patrick, Jr., and R.E. Phillips. 1976. Ammonium Diffusion as a Factor in Nitrogen Loss from Flooded Soils. Soil Science Society of America Journal 40(4): 528-533.

Redfield, A.C. 1958. The biological control of chemical factors in the environment. American Scientist 46:206-222.

Reed, D. 1989. Patterns of sediment deposition in subsiding coastal salt marshes, Terrebonne Bay, Louisiana: The role of winter storms. Estuaries 14:222-227.

Reed, D.J., Ed. 1995. Status and trends in hydrologic modification, reduction in sediment availability, and habitat loss/modification in the Barataria-Terrebonne estuarine system. Barataria-Terrebonne National Estuary Program Publication #20, Barataria-Terrebone National Estuary Program, Thibodaux, Louisiana. 338 pages plus Appendices.

Rose, R.E., E.E. Geldreich & W. Litsky. 1975. Improved membrane filter method for fecal coliform analysis. Appl. Microbiol. 29:532.

Schueler, T.R. 1987. Controlling urban runoff: a practical manual for planning and designing urban BMPs. Department of Environmental Programs, Metropolitan Washington Council of Governments, Washington, D.C.

Schueler, T., D. Hirschman, M. Novotney, and J. Zielinski. 2007. Urban stormwater retrofit practices Version 1.0: A user's manual. Manual 3 in the urban subwatershed restoration manual series. Center for Watershed Protection. Ellicott City, Maryland.

Smalley, A.E. 1958. The role of two inverterbrate populations, *Littorina irrorata* and *Orchelium fidicenium* in the energy flow of a salt marsh ecosystem. Ph.D. Dissertation, University of Georgia, Athens. 126p.

Smith, C.J. and R.D. DeLaune. 1986. Fate of Ammonium in a Gulf Coast Estuarine Sediment. Journal of Environmental Quality 15:293-297.

Qualls, R.G. 1989. Determination of total nitrogen and phosphorus in water using persulfate oxidateion : a modification for small sample volumes using the method of Koroleff (1983). Appendix A. pp. 131-138. *In* The biogeochemical properties of dissolved organic matter in a hardwood forest ecosystem: their influence on the retention of nitrogen, phosphorus, and carbon. Ph.D. dissertation, University of Georgia Institute of Ecology, Athens, Georgia, USA. University Microfilms, Inc., no. DEX9003448.

van der Valk, A.G. and C.B. Davis. 1978. The Role of Seed Banks in the Vegetation Dynamics of Prairie Glacial Marshes 59(2): 322-335.

White, D.A., T.E. Weiss, J.M. Trapani, and L.B. Thien. 1978. Productivity and decomposition of the dominant salt marsh plants in Louisiana. Ecology 59:751-759.