Vegetation and Soil Dynamics of a Louisiana Estuary Receiving Pulsed Mississippi River Water Following Hurricane Katrina

**John Day · Robert Lane · Matthew Moerschbaecher ·
Ronald DeLaune · Irving Mendelssohn ·
Joseph Baustian · Robert Twilley**

Abstract We monitored wetland biomass, decomposition, hydrology, and soil porewater chemistry at the Breton Sound estuary, which receives Mississippi River water from the Caernarvon river diversion structure. The estuary was in the direct path of hurricane Katrina in 2005, which caused a dramatic loss of wetlands in the upper basin. From March 2006 to October 2007, we made duplicate measurements at three distance classes from the diversion structure along the estuarine gradient as well as at a reference area, designated Near ($N_{1\&2}$), Mid ($M_{1\&2}$), Far ($F_{1\&2}$), and Ref ($R_{1\&2}$). Above- and belowground live biomass, porewater nutrients (NO_x, NH_4, and PO_4), salinity, sulfide, and soil Eh were measured every 2 months. Water level was monitored with gauges. Above- and belowground decomposition was measured using the litterbag (both) and cotton strip (belowground only) methods. Analysis of porewater parameters showed that stress factors affecting biomass production (porewater salinity, sulfide, flooding, and redox potential) were generally low to moderate, while measurable porewater nutrient concentrations occurred at all sites. Aboveground end of season live (EOSL) standing crop in October ranged from 423 g/m^2 at site M_2 to 1,515 at site F_1, and was significantly greater at site N_1 than at sites N_2, M_1, or M_2. Aboveground EOSL biomass during this study was significantly lower than previously measured in 1999, 2000, and 2001. Peak belowground biomass ranged from 8,315 g/m^2 at site R_2 to 17,890 g/m^2 at site N_1, which is among the highest reported in the literature, and there were significant increases throughout the study, suggesting recovery from hurricane Katrina. The decomposition bag data did not indicate any significant differences; however, the cotton strip decomposition rate was significantly lower at the lowest depth. Wetland surface vertical accretion ranged from 0.49 cm/year at N_2 to 1.24 cm/year at N_1, with site N_1 significantly greater than N_2, M_1, F_2, and R_1, and site N_2 significantly less than all other sites except site R_1. These findings show that marsh productivity and stability is related to a number of factors and no one factor can explain the impacts of the hurricanes.

Introduction

The Mississippi River delta plain formed as a series of delta complexes during the last 6,000–7,000 years (Roberts 1997; Blum and Roberts 2012). Switching of delta lobes, overbank flooding, and crevasses of the Mississippi River were major geological processes in the formation and maintenance of Mississippi delta wetlands (Roberts 1997; Day et al. 2007; Tornqvist et al. 2007). There are diverse wetland types in the delta, primarily the result of salinity gradients and the relative elevation of land to water, which is central to determining both the duration and frequency of flooding. Recent changes of the deltaic plain have led to the loss of about 25 % or 4,500 km^2 of wetlands during the twentieth century (Boesch et al. 1994; Day et al. 2000, 2007; Barras et al. 2008). This loss can be linked to modifications of the delta caused by flood control levees, which have almost completely separated the deltaic plain from the Mississippi River (Kesel 1988, 1989; Mossa 1996; Day et al. 2007). In addition, canal dredging, enhanced subsidence, and spoil bank construction associated mainly

J. Day · R. Lane (✉) · M. Moerschbaecher · R. DeLaune ·
I. Mendelssohn · J. Baustian · R. Twilley
Department of Oceanography & Coastal Sciences,
School of the Coast & Environment, Louisiana State University,
1197 Energy Coast & Env Bldg,
Baton Rouge, LA 70803, USA
e-mail: rlane@lsu.edu

with the oil and gas industries have dramatically altered the natural hydrology of the delta plain, promoting saltwater intrusion and limiting hydrological exchange (Bass and Turner 1977; Swenson and Turner 1987; Day et al. 2000; Shaffer et al. 2009, Morton et al. 2002).

There is concern that future increases in sea level will lead to increased rates of wetland loss (Blum and Roberts 2009). Relative sea-level rise (RSLR) is the combined effect of eustatic sea-level rise (worldwide average of 1–2 mm year^{-1} for the twentieth century; Gornitz et al. 1982) and coastal subsidence (as high as 10 mm year^{-1} in much of the Mississippi delta; Penland and Ramsey 1990). The IPCC (2007) predicted that eustatic sea level would increase in the twenty-first century by about 40 cm. However, satellite measurements over the past decade and half show that sea level is currently rising at about 3.4 mm/year (Leuliette et al. 2004), and recent publications suggest that twenty-first century eustatic sea level rise may be a meter or more (Rahmstorf 2007; Pfeffer et al. 2008; Rohling et al. 2008; Vermeer and Rahmstorf 2009). In order to maintain a stable wetland, rates of vertical accretion must keep up with the rise in eustatic sea level as well as local subsidence, maintaining a stable elevation between land and water. Recent information indicates that by the end of the century, relative sea level rise will be significantly higher than that of the twentieth century, forcing wetland landscapes to adapt to increased water levels at a time when geologic processes that contribute to land elevation have been repressed.

There is strong consensus in the scientific and management community that the reconnection of the Mississippi River to the deltaic plain is essential for the long-term survival of wetlands in coastal Louisiana (Templet and Meyer-Arendt 1988; Kesel 1989; Boesch 1996; Day et al. 1997, 2000, 2007; Gosselink 2001; Boesch et al. 2006). The State of Louisiana and the Federal Government have developed a plan for a series of river diversions structures that will allow Mississippi River water to enter nearby estuaries, mimicking natural flooding events (http://coastal.la.gov/; Chatry and Chew 1985; LDNR 1998; USCOE 2004). The underlying restoration strategy is that river diversions promote land building by restoring geologic processes that stimulate wetland vegetative growth. The combined effects of vegetation trapping of sediments and production of organic matter (particularly belowground biomass) increase or stabilize wetland surface elevation, which is critical due to the projected rates of RSLR described above. The combined effect of geologic and ecologic processes to increase wetland landscape elevation is a key assumption of the restoration program. This study focuses on the upper Breton Sound estuary, which has received Mississippi River water since 1991 via the Caernarvon river diversion structure (Day et al. 2009), and has experienced widespread marsh disturbance during the passages of Hurricanes

Katrina and Rita in 2005 (Barras et al. 2008; Morton and Barras 2011).

Some have attributed the wetland deterioration following Hurricanes Katrina and Rita to high nutrient input to the Breton Sound Estuary from the Caernarvon river diversion (Kearney et al. 2011). The contribution of belowground biomass to soil organic matter depends on several environmental conditions that control biomass allocation, particularly associated with acquisition of growth limiting resources (Tilman 1985; Bazzaz 1997). For example, in nutrient-poor soils, plants tend to allocate more biomass to roots in order to acquire limiting resources, which leads to a more favorable carbon/nutrient balance in the soil horizon (Grime 1977; Chapin 1980; Tilman 1985). Conversely, under high nutrient conditions, plants have the ability to adjust biomass allocation from roots to shoots, potentially decreasing soil strength and resiliency (Darby and Turner 2008b, c; Turner 2010). The Mississippi River has become nutrient enriched over last several decades (Turner and Rabalais 1991), potentially changing fertility gradients of deltaic soils and thus lowering belowground productivity (Darby and Turner 2008c). Kearney et al. (2011) based their conclusion on these factors; however, a number of studies reported that coastal marshes with high nutrient loading did not have decreased belowground production (Anisfeld and Hill 2012; Fox et al. 2012) or increased belowground decomposition (Ibañez et al. 1999; Laursen 2004). In addition, Rivera-Monroy et al. (2011) reported that mangroves affected by hurricane Wilma in south Florida had reduced belowground biomass immediately after the storm but that biomass tripled after the storm.

The purpose of this study was to measure above- and belowground biomass and decomposition, soil porewater chemistry, hydrology, and accretion following hurricanes Katina and Rita at a Mississippi delta marsh receiving river water input from the Caernarvon freshwater diversion. The approach was to use natural environmental gradients downstream from the diversion structure, and thus inputs of both sediment and nutrients, as controls of productivity and decomposition of wetland vegetation. We used Breton Sound estuary as an experimental unit since it is directly influenced by the Caernarvon river diversion. Based on the literature cited above, we hypothesized that above- and belowground productivity and organic matter decomposition would be greatest near the diversion and decrease with distance, and that above- and belowground biomass would increase over time following hurricane Katrina.

Site Description

The Caernarvon river diversion structure is located down-river of New Orleans at River mile 81.5 (131 km) on the east

bank of the Mississippi River near Caernarvon, Louisiana. The water control structure has been in operation since August 1991 and consists of five box culverts with vertical lift gates. The structure has a maximum discharge rate of 226 m^3/s, but an average discharge of only 21 m^3/s (Lane et al. 2006). Peak discharge from the Caernarvon diversion structure during this study was 140 m^3/s, which occurred in March 2007 (Fig. 1). It should be noted that the volume of river water passing through the Caernarvon structure to the estuary is much lower, by an order of magnitude or more, compared to the volume that flowed into the estuary before the levees were built. There were numerous natural crevasses and minor distributaries as well as seasonal overbank flooding along the lower Mississippi River prior to human manipulation (Welder 1959; Kesel 1988, 1989; Roberts 1997; Day et al. 2000), with peak flows ranging from 5,000 to 10,000 m^3/s (Davis 2000).

On August 29, 2005, hurricane Katrina passed over southeastern Louisiana with winds in excess of 200 km/h, and a tidal surge southeast of 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 6-meter storm surge over the estuary, causing massive disturbance to the extensive wetlands in the upper basin (Boesch et al. 2006; Day et al. 2007; Barras et al. 2008). On September 23–24, 2005, hurricane Rita passed south of Breton Sound and led to further disturbance. The hurricanes not only led to widespread disturbance of wetlands but also altered the hydrology by infilling many waterways, only some of which were dredged and reopened. These changes to the hydrology of the area are detailed below.

There are about 1,100 km^2 of fresh, brackish, and saline wetlands interspersed with waterbodies in the Breton Sound estuary. Mississippi River water diverted through the Caernarvon structure travels 30–40 km before reaching open waters of Breton Sound, and an additional 50 km before reaching the Gulf of Mexico (Fig. 2). Prior to hurricane Katrina, water passed through two major routes: (1) Lake Leary and Bayou Terra aux Boeufs to the east, and (2) Manuel's Canal and River aux Chene to the west (Lane et al.

2007; Snedden et al. 2007). The latter route was blocked by debris after hurricane Katrina, forcing more water to flow via Lake Leary (Fig. 2). Hydrology was also changed along Bayou Mandeville, which connects Big Mar and Lake Leary, by spoil deposited along the western edge of the bayou as a continuous low levee, hydrologically separating the bayou from the wetlands. There was also considerable overland flow, especially south of Big Mar, which was dependent on structure discharge, winds, and water levels in the Gulf of Mexico (Snedden et al. 2007).

The Breton Sound estuary is part of the Plaquemines-St. Bernard delta complex, which was formed between 2,000 to 3,000 years ago (Scruton 1960; Roberts 1997). Since then, approximately half of the original wetlands have disappeared by the processes of shore-face erosion and coastal subsidence (Penland et al. 1988). Mississippi River flow into the estuary decreased drastically with the construction of flood control levees soon after the colonization of New Orleans by the French in 1719 (Welder 1959; Colten 2000). However, there were still major river inputs to the estuary via crevasses, minor distributaries, and overbank flooding until the early part of the twentieth century (Welder 1959; Kesel 1988, 1989; Roberts 1997; Day et al. 2000), as well as when the levee at Caernarvon was destroyed with dynamite in 1927 to relieve flooding upriver at New Orleans (Barry 1998). Flow during the 1927 breech reached almost 10,000 m^3/s. For a review of recent studies at Caernarvon and the Breton Sound estuary, see Day et al. (2009).

Methods

Duplicate study sites were established at three distance classes in the estuary with respect to the diversion structure, designated Near ($N_{1\&2}$), Mid ($M_{1\&2}$), and Far ($F_{1\&2}$), with hydrologic distances of 3.4, 12.3, and 18.1 km, respectively (Fig. 2). Earlier studies have shown that significant changes in environmental parameters occur along this gradient (Lane et al. 1999, 2004, 2007; DeLaune et al. 2003; Delaune and Pezeshki 2003; Snedden et al. 2007; Steyer 2008; Day et al.

Fig. 1 Mississippi River water flow through the Caernarvon diversion structure from January 2006 to October 2007 (monthly average). *Arrows* indicate when sampling occurred for this study
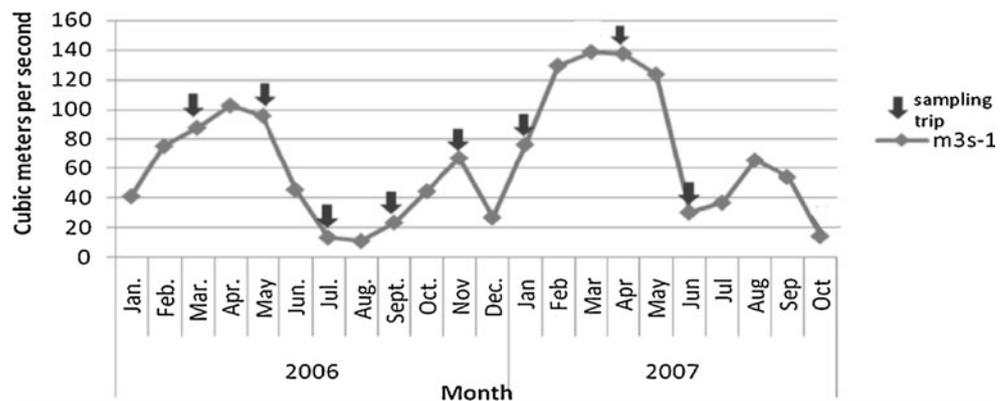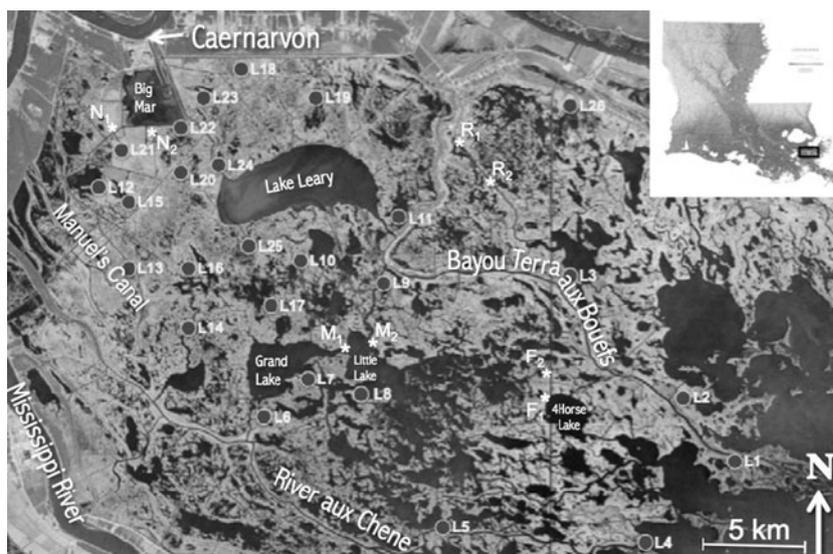
Fig. 2 Major lakes and waterways of the Breton Sound estuary. Intensive sampling site locations are marked with *asterisks* (*N1, N2, M1, M2, F1, F2, R1,* and *R2*), and *L* sites refer to earlier studies of aboveground end of season live biomass that were remeasured for this study

2009). Two reference sites ($R_{1\&2}$) located outside the immediate influence of diverted river water were also sampled. Study sites were located 10–30 m from the water's edge in relatively homogenous vegetation. Thus, these stations represented streamside wetlands, although not immediately adjacent to water bodies. Site visits were made eight times, approximately each 2 months, to sample wetland biomass and soil pore water characteristics: March 23–24, May 11–12, July 12–13, September 7–8, and November 9–10, 2006, and January 30–31, April 11–12, and June 5–6, 2007.

In addition to the 8 intensive study sites, 26 additional historical sites were sampled in late September 2006. These study sites were selected from locations where previous end of season live (EOSL) measurements were taken that showed significant changes in biomass with distance from the diversion structure (DeLaune et al. 2003; Day et al. 2009; and unpublished data; Fig. 2). The 26 historical sites were sampled for above- and belowground biomass, and porewater chemistry as described below. Floral species composition based on relative dominance was assessed at both intensive and historical study sites during late September 2006 using the Braun-Blanquet method (Poore 1955).

Water Level

In order to observe general patterns of flooding frequency and hydrology, water level was measured at each location (the first of each replicate) using YSI sondes installed inside piezometers. Water level was recorded every 15 min. Data were referenced to the surrounding wetland surface elevation position.

Porewater Sampling

Interstitial porewater was sampled in triplicate at each study site using a narrow diameter plastic tube with holes along the last several centimeters connected to a 50-ml syringe (as in McKee et al. 1988). The rigid plastic tube (3 mm diameter) was inserted into the soil to 15 cm depth and suction applied by syringe. A three-way valve between the syringe and collection tube allowed the expulsion of air and debris. This apparatus collected relatively clear interstitial water from the wetland substrate within a few seconds without exposure to the atmosphere. The first 5–20 ml of each sample was discarded to flush contaminants.

Five-milliliter samples of porewater were collected for sulfide analysis. These samples were immediately transferred to a vial with an equal volume of antioxidant buffer, and returned to the laboratory to be analyzed for total sulfide (as parts per million S^{2-}) with a Lazar sulfide electrode (Model IS-146, Lazar Research Laboratories, Los Angeles, CA). A standard curve was constructed with a series of dilutions of Na_2S prepared with the antioxidant buffer. Samples for June 2007 were contaminated and were not used in the statistical analysis.

Approximately 60–80-ml samples of porewater were collected and immediately stored on ice for nutrient analysis. Usually within 4–6 h of collection, but never more than 24 h, the samples were filtered through pre-rinsed 0.7-um Whatman GF/F glass fiber filters into new clean scintillation vials and frozen until analysis. The samples were transported to the School of the Coast & Environment Analytical Services Laboratory at Louisiana State University for analysis. Nitrate+nitrite (NO_x) was determined using the automated cadmium reduction method, ammonium (NH_x) by the automated phenate method, and phosphate (PO_4) by the automated ascorbic acid reduction method (APHA 1985), all with an Alpkem© autoanalyzer (APHA 1985). Salinity was determined using an YSI Model 85 salinity meter (www.ysi.com).

Aboveground Biomass

Aboveground biomass was sampled each 2 months using triplicate 0.25-m^2 clip plots (0.5 by 0.5 m) (Shew et al. 1981). Vegetation within each clip plot was cut as close to the wetland surface as possible, placed in labeled paper bags, and brought back to the laboratory where live vegetation was separated from dead, dried at 60 °C, and weighed. The dry weight of the live material was extrapolated to live dry weight per square meter (grams dry weight per square meter).

Aboveground net annual primary productivity (NAPP) was calculated using the following four methods: (1) the Milner–Hughes method of summing all positive changes in biomass measurements throughout the year (Milner and Hughes 1968); (2) the maximum–minimum (Max–Min) method based on the maximum difference in live biomass over the growing seasoning (Stroud and Cooper 1968; Stroud 1976); (3) the EOSL method of measuring all live material present just prior to senescence in late September or early October (White et al. 1978; Burdick et al. 1989); and (4) the peak standing crop (PSC) method that uses peak biomass found during the study.

EOSL aboveground biomass was measured during late September 2006 at the 8 intensive sites and at the 26 historical sampling sites (Fig. 3) using triplicate 0.25-m^2 clip plots as described above. The data collected in our study were compared to end of season live biomass measured in 1999, 2000, and 2001 using similar methods (Day et al. 2009).

Belowground Biomass

Belowground wetland biomass was collected each 2 months using triplicate 7.5×30-cm thin-walled stainless steel tubes with sharpened bottoms (Gross et al. 1991; Delaune and Pezeshki 2003). Belowground wetland biomass was collected at the same locations and times as aboveground biomass. The cores were sealed and transported top-end-up to Louisiana State University where they were cut into 5-cm sections to facilitate processing of the root material, and to measure the vertical distribution of belowground biomass. In the laboratory, each core segment was washed in a 2-mm sieve over a 0.5-mm sieve to prevent the loss of dead and fine root material (Darby and Turner 2008). Roots and rhizomes were separated from small particulate material with a 2-mm mesh set under running water, and live and dead fractions separated using the criteria of live material being white and turgid and dead material being dark and flaccid (Valiela et al. 1976; Symbula and Day 1988; Fitter 2002). The live fractions were then dried at 60 °C to a constant weight. Net annual belowground primary productivity was calculated by the Max–Min and Milner–Hughes methods described above.

Soil Redox Potential

Soil cores were taken in the field and carefully sealed to prevent air entry. Soil redox potential of each soil core was measured at the LSU Biogeochemistry Laboratory using bright platinum electrodes (Patrick et al. 1996). The electrodes were placed in the intact cores to 15 cm depth, and allowed to equilibrate in situ for at least 1 h prior to measurement (Faulkner and Richardson 1989). Each electrode was checked for accuracy with quinhydrone in pH 4 and 7 buffers, which give 218 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, IL). The potential of a calomel reference electrode (+244 mV) was added to each value to calculate Eh (Patrick et al. 1996). The cores from June 2007 were lost and not used in the statistical analysis.

Accretion

Wetland surface accretion was measured using feldspar marker horizons established at the eight intensive study sites (Cahoon and Turner 1989; DeLaune et al. 1983, 2003). Powdered feldspar clay was placed on the wetland surface approximately 1 cm thick at three randomly placed 0.25-m^2 plots at each of the study sites in March 2006. Accretion was measured destructively during December 2008 by extracting plugs using a shovel, cleanly slicing the plug into several sections, and measuring the thickness of material above the

Fig. 3 Mean daily water levels at the study locations. Zero is set at the wetland surface elevation for each site
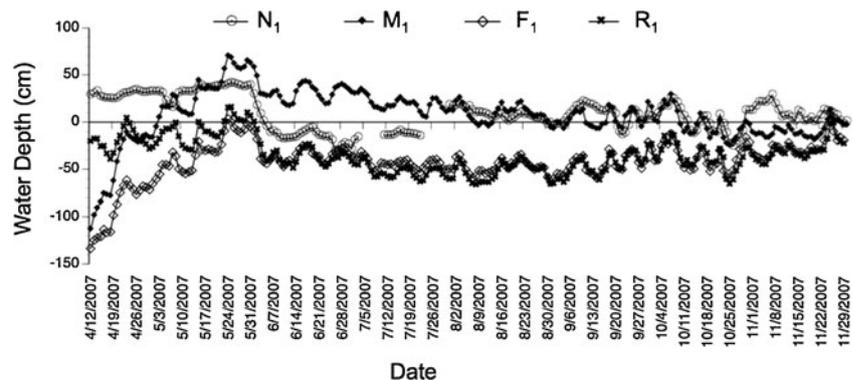
horizon at several locations (Lane et al. 2006). The rate of vertical accretion was calculated by dividing the thickness of material above the horizon by the amount of time the horizon had been in place (21 months).

Decomposition

Decomposition rates were measured using the litterbag (Harmon et al. 1999) and the cotton strip assay (Slocum et al. 2009) techniques. Litterbag decomposition was measured of above- and belowground plant tissue at 6 and 12 months. Twenty-four bags (10×30 cm) with 1-mm mesh were inserted at each site, 12 above- and 12 belowground in June 2007. Aboveground bags contained 5 g of oven-dried leaves and stems, and were placed directly on the wetland surface. Belowground bags were filled with oven-dried roots and rhizomes, and inserted between 10 and 20 cm below the wetland surface. At each site, half of the above- and belowground bags contained local plant materials from that individual site, and the other half contained a mixture of plant materials from all study sites. The comparison of local and mixed materials determined if there were inherent differences in decomposition between the sites, or if observed differences were a result of the vegetation at a given site being more easily decomposable.

Soil decomposition was measured using cotton strips placed in the wetland soil during August 2007 and June 2008. The strips (10×30 cm) were inserted vertically into the soil, three strips per treatment, and removed after 12 days in 2007 and 11 days in 2008. Reference strips were inserted at each site and immediately removed. All strips were washed, dried, and cut into 2-cm sub-strips. Depth intervals were based on the midpoint of each sub-strip. A Dillon Quantrol Snaphot tensometer was used to measure the force required in Newtons to break the sub-strips, and the cotton tensile strength loss (CTSL) was calculated with the following equation:

$$\%\text{CTSL}\left(d^{-1}\right) = \left[(1 - (N/C))/D\right] \times 100$$

where N is the tensile strength of the experimental sub-strip in Newtons, C is the mean tensile strength of the reference sub-strips, and D is the number of days the strip was in the ground. Details are given by Slocum et al. (2009).

Statistical Analysis

An analysis of variance was carried out with interaction between time and sublocation (replicate plots) nested within location, with the factors of interest being above- and belowground biomass, soil redox potential, and porewater salinity, nutrients (NO_x, NH_x, PO_4), and sulfate. Least square means were calculated for significant interactions and main effects. Data were modeled using simple and multiple regression analyses, and the parameters of the resulting models were adjusted to eliminate the effects of multicollinearity between parameters, as well as correlations that were most likely caused by high data variation combined with low sample size. Above- and belowground litterbag decomposition data were analyzed separately using an ANOVA with sublocation nested within location and source material, location, and sampling time as the main effects. Post hoc Tukey adjustments were made to determine significant differences between means. The cotton strip data were analyzed to determine the effects of treatment, sampling period, and depth on cotton strip decomposition using an ANOVA with a factorial design and depth as a repeated measure, followed by Fisher's Protected LSD test when significant differences were found. All statistical test were performed at an $\alpha = 0.05$ level using SAS version 9.1 (SAS Institute Inc 2003) or JMP statistical software produced by SAS Institute, Inc. (Sall et al. 2005).

Results

The analysis of variance found significant time×location interactions for porewater salinity and nutrients, and significant time×sublocation interactions for above- and belowground biomass, soil redox potential, and sulfate (Table 1). These significant interactions guided graphical data presentation. There were also significant main effects over time for all parameters, and by location (and sublocation) for NH_4, salinity, redox, as well as above- and belowground biomass (Table 1).

Water Level

Mean daily water levels at the study sites were calculated from hourly data recorded from April 12 to November 27, 2007. Water level data at the Near site were extrapolated from the water level gage height at the Caernarvon structure due to gauge malfunction. The highest water levels occurred at all sites in late May and early June, after which levels gradually decreased (Fig. 3). Individual flooding events lasted from 18–54 and 5–97 days at the Near and Mid sites, respectively, while flooding events at the Far and Ref sites lasted 1–2 and 2–5 days, respectively.

Soil Redox Potential and Sulfide Concentrations

Soil redox potential had great temporal and spatial variation (Fig. 4). Temporally, redox levels increased from the first measurements during March 2006, were higher during May and July, then decreased in September (range, −244.7±82 to 140.3±12 mV), and remained low through January and April 2007. Spatially, the lowest redox values were generally at

Table 1 Results of statistical analysis of variance, given as p values

	NO$_x$	NH$_4$	PO$_4$	SO$_4$	Salinity	Redox	AGB	BGB
Time	<.0001*	0.0017*	<.0001*	<.0001*	<.0001*	<.0001*	0.0444*	<.0001*
Location	0.9131	0.0471*	0.5761	.	<.0001*	<.0001*	<.0001*	0.0036*
Sublocation	0.9869	0.0419*	0.3961	0.0976	<.0001*	<.0001*	<.0001*	<.0001*
Time×sublocation	<.0001*	<.0001*	<.0001*	<.0001*	<.0001*	0.0042*	<.0001*	0.0017*
Time×location	0.0004*	0.0003*	<.0001*	0.1117	0.0368*	0.3486	0.0627	0.0814

Asterisks indicate significant statistical differences (p<0.05)

the Mid and Ref sites. The highest redox values were at site F$_2$ during May and July 2006, followed by site N$_2$ in May 2006, January and April 2007. Site N$_1$ also had relatively high redox values (Fig. 4). Site N$_2$ had the highest range of any of the sites.

Sulfide concentrations (reported as parts per million S^{2-}) ranged from below detection limit (<0.1 ppm) to 59.3 ppm, and decreased over the course of the study (Fig. 4) as the area freshened. Sulfide concentrations ranged from below detection limit to 27.2 and 10.7 ppm at sites N$_1$ and N$_2$, respectively. Sulfide concentrations at the Mid locations ranged from 0.7 to 35.9 ppm. Far locations sulfide concentrations ranged from below detection to 29.7 ppm. Concentrations at the Ref location ranged from below detection to 59.3 ppm.

Porewater Chemistry

Salinity at the Near sites was significantly lower than at the Far sites during all sampling events, and were lower than the
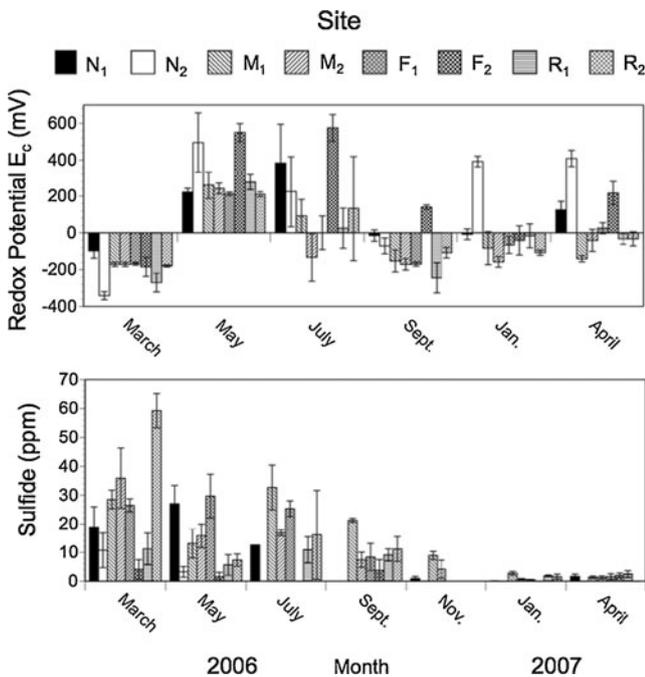
Ref and Mid sites most of the time (Fig. 5). Salinity levels peaked in September 2006, corresponding with the period of least amount of river diversion input and highest mean coastal water levels. Mean porewater salinity ranged from 0.9 to 4.8 psu at the Near sites, 1.8 to 7.3 psu at the Mid sites, 4.2 to 15.0 psu at the Far sites, and 2.8 to 9.0 psu at the Ref sites. These results confirm that the river diversion input establishes a porewater salinity gradient in the Breton Sound estuary. This conclusion is supported by the findings of Hyfield et al. (2008) who reported that groundwater input through the levees accounted for less than 0.05 % of freshwater inputs to the upper basin.

Dissolved inorganic N and P porewater concentrations were sometimes elevated at the Near sites, but other sites and times had generally low values, especially NO$_x$. Mean
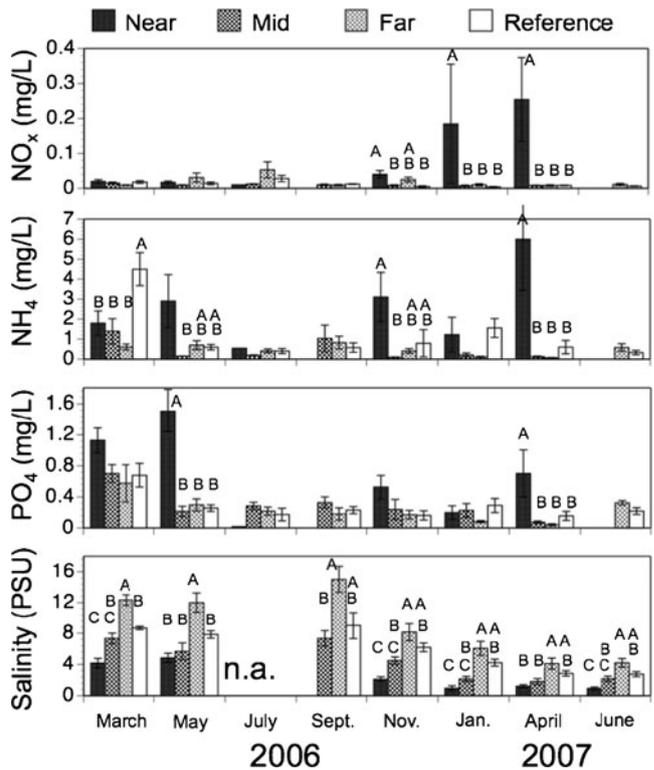

Fig. 4 Soil redox potential and sulfide concentrations at each sublocation over time. Data are means±1 standard error

Fig. 5 Soil porewater NO$_x$, NH$_4$, PO$_4$, and salinity concentrations over time. Data are means±1 standard error, and *letters* indicate significant statistical differences (*p*<0.05)

porewater NO_x concentrations ranged from below detection limit (<0.01 mg/L) to 0.05 mg/L at the Mid, Far, and Ref sites. Only at the Near sites was NO_x high reflecting the impact of the diversion (Fig. 5). The highest NO_x concentrations (3.52 mg/L) were found at site N_2 in January and April 2007 during periods of high diversion discharge, which were significantly higher than at the other study sites ($p=0.0415$ and 0.0189, respectively). Mean porewater NH_4 concentrations ranged from 0.07 to 5.99 mg/L, with the highest concentrations found at the Near and Ref sites, primarily in the spring (Fig. 5). Ammonium concentrations often were greater than 1.0 mg/L at the Near sites but were generally near 1.0 or less at the other sites. Phosphate (PO_4) concentrations were elevated at the Near sites in March, May, November, and April. At other times and locations, concentrations were generally 0.3 mg/L or less.

Aboveground Biomass

Aboveground biomass ranged from zero (site N_2, January and April 2007) to 1,581.3±236 g/m^2 (site N_1, June 2007), with an overall mean of 720.5±37 g/m^2, and generally tended to be highest during the growing season from May to November and lowest during the winter (Fig. 6). Peak aboveground biomass, or PSC, had an overall mean of 1,262.2±136 g/m^2 and varied temporally between locations, mostly occurring between May and November. End of season live (EOSL) biomass ranged from 289.3±264 to 1,289.3±573 g/m^2, with an overall mean of 827.0±129 g/m^2. Regression analysis failed to detect a correlation between EOSL and PSC biomass data ($p=0.0909$).
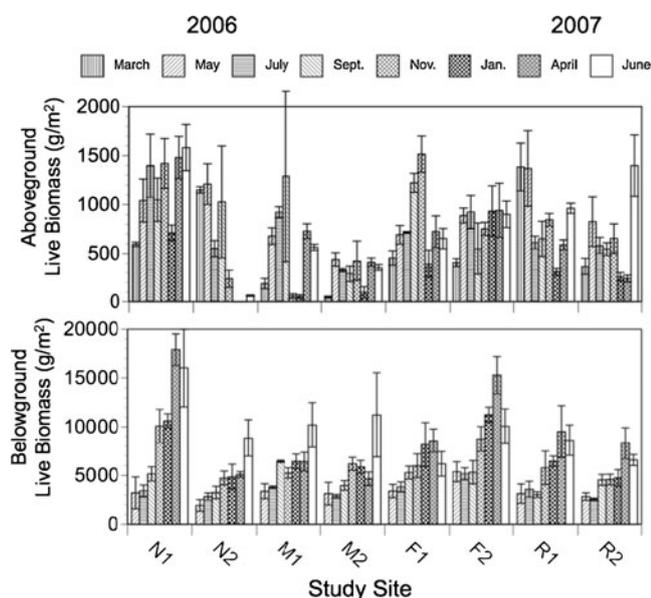
Aboveground NAPP ranged from 392 to 1,265 g/m^2/year, with an overall mean of 848.0±125 g/m^2/year, using the Milner–Hughes method and 392 to 1,236.0 g/m^2/year, with an overall mean of 838.1±123 g/m^2/year, using the Max–Min method (Fig. 7). End of season live biomass was correlated with NAPP using the Max–Min method ($p=0.0282$), but this correlation was absent when using the Milner–Hughes method ($p=0.3154$). Peak standing crop, however, was correlated to NAPP using both the Max–Min and Milner–Hughes methods ($p=0.0084$ and 0.0194, respectively).

End of season live (EOSL) aboveground biomass measured in 2006 at the historical sampling sites ranged from 117.3±39 to 3,620.0±238 g/m^2, with an overall mean of 915.2±444 g/m^2 (Fig. 8). The highest biomass was at site L12 (3,620.0±238 g/m^2), located approximately 10 km southwest of the diversion structure (Fig. 3). There were significant linear trends of decreasing aboveground EOSL biomass with distance from the diversion structure for years 2000 ($p=0.0024$), 2001 ($p=0.0046$), and 2006 ($p=0.0448$), as well as for the combined data set ($p=0.0010$). Mean aboveground EOSL biomass was lower in 2006 (919.8±101) compared to 1999 (1,779.4±169 g/m^2), 2000 (2,274.0±355 g/m^2), or 2001 (1,522.4±179 g/m^2), and EOSL biomass was greater in year 2000 compared to all other years ($p<0.0001$; Fig. 8). Multiple regression analysis failed to detect a significant correlation between aboveground EOSL biomass and porewater chemistry.

Fig. 6 Above- and belowground live biomass at each sublocation over time. Data are means±1 standard error
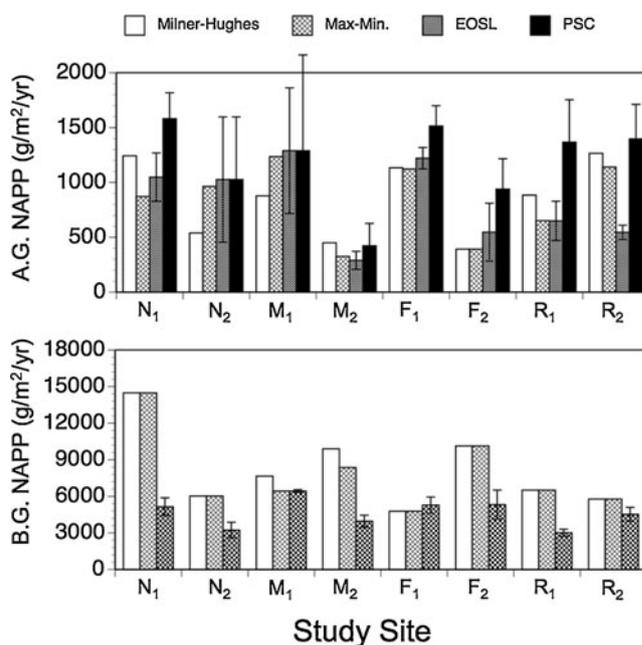

Fig. 7 Above- (A.G.) and belowground (B.G.) NAPP (grams per square meter per year) calculated using the Milner–Hughes, Max–Min, end of season live (EOSL), and peak standing crop (PSC) methods
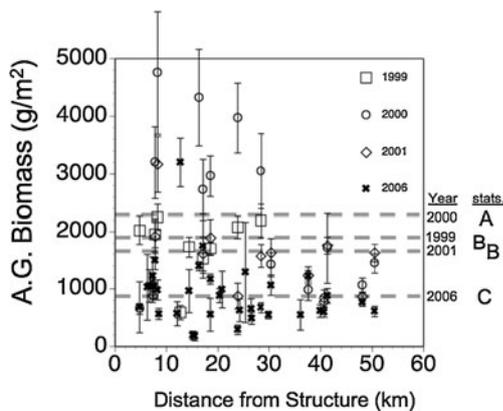
Fig. 8 Aboveground EOSL biomass versus hydrological distance from the Caernarvon diversion structure. Data are means ± 1 standard error, dashed lines are yearly means, and letters indicate significant statistical differences among overall means ($p<0.05$)

Species Composition

Site N_1 had the greatest variety of floral species of all the study sites, with *Panicum virgatum*, *Polygonum* sp., and *Alternanthera philoxeroides* as the dominant species, while *Spartina patens* was present in minor abundance. Site N_2 was highly impacted by hurricane Katrina, with accordion-like lateral folding that increased surface elevation and hydrologically isolated the site, which was exasperated by its proximity to a spoil bank along the southern edge of Big Mar (Fig. 2). Standing water was never observed at N_2, which was dominated by a thick mat of *Vigna luteola* and *Ipomoea* sp. The Mid site wetlands were dominated by *S. patens* and *Schoenoplectus americanus*. The Far sites were dominated by *Spartina alterniflora*, *S. patens*, and *Juncus roemerianus*. *S. americanus* and *S. patens* were co-dominant at the Ref sites, with *Distichlis spicata* being a minor species.

Belowground Biomass

Live belowground biomass (to 30 cm depth) ranged from 1,920±592 to 17,890±1,625 g/m² (Fig. 6), while belowground biomass had a mean of 11,234±1,239 g/m². Linear regression analysis indicates belowground biomass increased through time at all sites ($p<0.05$), with peak belowground biomass occurring during April or June 2007 at all sites.

Belowground NAPP ranged from 4,776 g/m²/year at site F_1 to 14,485 g/m²/year at site N_1 using the Milner–Hughes and Max–Min methods (Fig. 7). These methods produced similar results for all sites except the Mid sites, where the Milner–Hughes method gave higher results than the Max–Min method. The overall mean using the Milner–Hughes and Max–Min methods was 8,157±1,130 and 7,813±1,121 g/m²/year, respectively. End of season live belowground biomass

collected in late September 2006 ranged from 3,014±291 to 6,433±113 g/m²/year, with a mean of 4,615±411 g/m²/year, and was not correlated to NAPP. Peak belowground biomass in April and June, however, was linearly correlated to NAPP using the Milner–Hughes ($R^2=0.89$) and Max–Min ($R^2=0.95$) methods.

Accretion

Wetland surface accretion averaged 0.97±0.09 cm/year, with the highest accretion measured at site N_1 (1.24± 0.09 cm/year), followed by R_2, F_1, and M_2 (Fig. 9). The lowest accretion was measured at site N_2 (0.47±0.02 cm/year), followed by R_1, F_2, and M_1. Site N_1 was significantly greater than sites N_2, M_1, F_2, and R_1, while site N_2 was significantly lower than all other sites except site R_1 ($p<0.0001$). There was not a significant correlation between accretion and above- or belowground biomass or NAPP.

Decomposition

There was a significant location×depth interaction in cotton strip decomposition rates ($F=1.04$, $p=0.0018$). Cotton strip decomposition was highest near the wetland surface and decreased with depth in all locations to varying degrees (Fig. 10). All locations had similar decomposition rates in the top 5 cm, with the exception of the Far sites, which had lower surface decomposition rates. The Ref site had the lowest decomposition rates from 5 to 23 cm. The Far, Near, and Mid sites had similar decomposition rates between 7 and 17 cm. Deeper than that the decomposition rates at the Mid and Far sites declined roughly 20 %, while decomposition at the Near sites remained constant. The protected LSD test indicated that decomposition rates at the Near sites were higher than the other sites at the lowest depth.
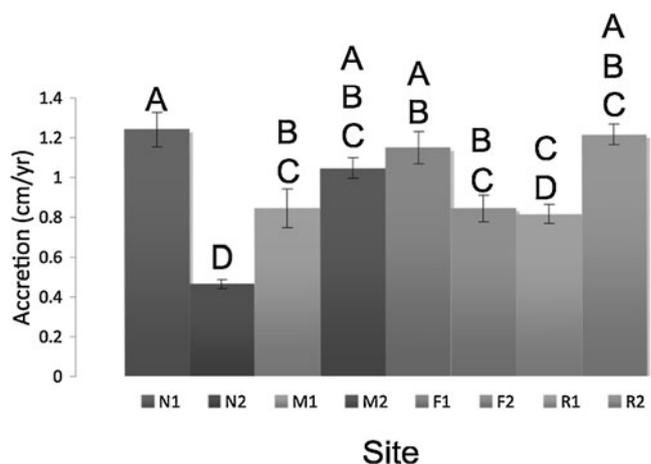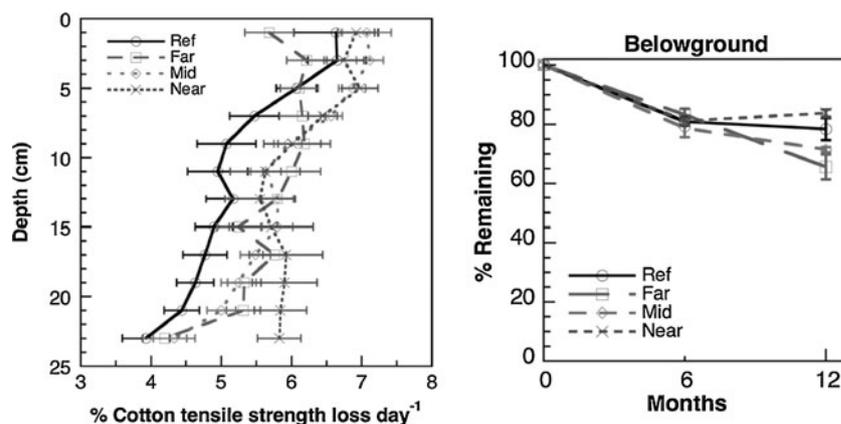

Fig. 9 Wetland surface vertical accretion. Data are means±1 standard error, and *letters* indicate significant statistical differences ($p<0.05$)

Fig. 10 Depth profile of cotton strip decomposition averaged over time (*left*) and belowground litterbag decomposition averaged over material type (*right*). Data are means±1 standard error

Discussion

Litterbag decomposition increased from the 6-month to the 12-month sampling both aboveground (F=122.3, p<0.001) and belowground (F=8.94, p=0.004). There was a significant time×location interaction belowground (F=9.24, p<0.001), but not aboveground (F=1.24, p=0.3004). Belowground, the Near and Ref sites showed little to no decomposition between 6 and 12 months, while the Far and Mid sites showed an increase in decomposition over this same time period (Fig. 10).

Aboveground there was a significant location×material interaction in litterbag decomposition (F=5.66, p=0.0016). There were no differences in decomposition between the local and mixture materials in any individual site, but there were differences between the sites (Fig. 11). The local and mixture materials at the Near sites and the local material at the Ref sites had significantly lower surface decomposition than both materials at the Far and Mid sites. Belowground there was also a significant location×material interaction in litterbag decomposition (F=6.63, p=0.0006). At the Near, Mid, and Ref sites, the local and mixture materials decomposed at the same rates, but at the Far sites, the local material decomposed faster than the mixture material.

The results of this study showed that belowground marsh biomass increased over time following the hurricanes. Aboveground biomass was lower than earlier studies carried out before the hurricanes, and there was no consistent trend of increasing biomass over time following the hurricanes. Differential flooding at the sites explained some of the variability among sites. Porewater salinity and sulfide concentrations were higher at the beginning of the study at all sites likely reflecting high salinity water brought in by the hurricanes and the flushing by the diversion. There were small to no changes in above and belowground decomposition among sites. Accretion (9.7±0.9 mm/year) was generally sufficient at all sites to offset local relative sea-level rise (2–12 mm/year; Gornitz et al. 1982; Penland and Ramsey 1990).

Belowground NAPP in the outfall area of the Caernarvon river diversion was highly variable, ranging from 4,776 to 14,485 g/m²/year. A net increase in belowground biomass occurred during this study at all eight of the intensive sites, suggesting vegetation recovery through increased root growth following the disturbance caused by hurricane

Fig. 11 Decomposition of local and mixed litterbag materials aboveground and belowground averaged over time. Data are means±1 standard error, and *means sharing a letter* were not significantly different
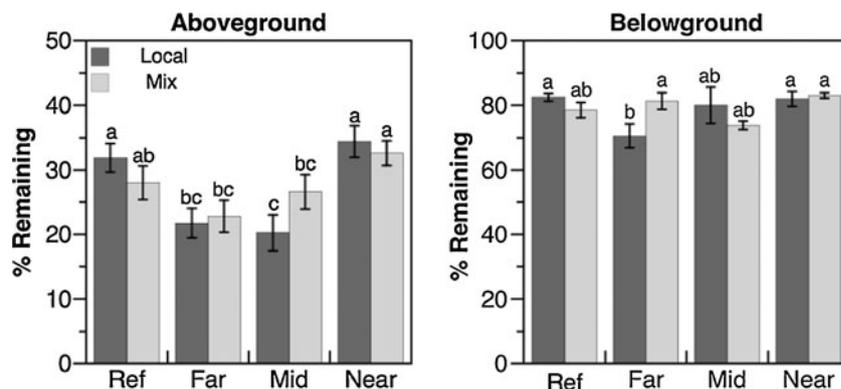
Katrina. Such recovery of belowground biomass has been reported for mangroves affected by hurricane Wilma in south Florida where belowground biomass tripled after the storm, mostly due to high production of fine roots (Rivera-Monroy et al. 2011). Although we believe that the increase in belowground biomass was recovery after the storm, the lack of pre-storm data on below ground biomass precludes a definitive conclusion. Previous studies in coastal salt marshes have reported belowground NAPP measured using the Milner–Hughes method ranging from 384 (Brazil; Lana et al. 1991) to 11,676 $g/m^2/year$ (Louisiana; Darby and Turner 2008a). Belowground NAPP at all sites besides N_1 of this study were in the upper range of previously published data (Table 2). Site N_1, however, had higher belowground NAPP (14,485 $g/m^2/year$) than previously published. Several of the belowground biomass values at N1 were among the highest reported in the literature, and *P. virgatum* was the most important root-forming species at that site. *P. virgatum* has been reported to do well at high nutrient concentrations and has high belowground biomass (George et al. 1995; McLauglin and Kszos 2005; Izdepski et al. 2009). In addition, the prostate nature of *Polygonum* and *Alternanthera* may have led to stems laying across the soil and thus being incorporated into belowground biomass

Aboveground NAPP for this study using the Milner–Hughes method ranged from 392 $g/m^2/year$ at site F_2 to 1,265 $g/m^2/year$ at site M_2. The lower NAPP values including those at sites F_2, M_2 (449 $g/m^2/year$), and N_2 (539 $g/m^2/year$) are more similar to estimates made in northeastern coastal salt marshes, such as Massachusetts (420 $g/m^2/year$) and New Jersey (500 $g/m^2/year$; Valiela et al. 1976; Smith et al. 1979). EOSL aboveground biomass was as high as 4,749 g/m^2 during sampling carried out from 1999 to 2001 (Fig. 8), while the highest EOSL aboveground biomass measured in this study (2006) was 3,200 g/m^2, suggesting that the hurricane may have suppressed aboveground marsh growth in the 2 years after the storm. Also, we observed that species composition shifted in the upper basin from primarily *S. patens* in 2000 to *Polygonum*/*Alternanthera* in 2006/2007. This shift in composition could have been a strong contributing factor in the observed change in aboveground biomass.

Numerous studies have shown that wetland productivity varies depending on species tolerance or adaptations to various environmental stressors, such as salinity and flooding (Spalding and Hester 2007). For example, *S. patens* production is more negatively affected by increased flooding depth than salinity below 6 psu (Burdick et al. 1989; Pezeshki and DeLaune 1991; Hester et al. 1996; Spalding and Hester 2007), while *Panicum hemitomon* productivity is stimulated to a degree by increased flooding (Fisher 2003; Willis and Hester 2004; Spalding and Hester 2007). The interactive effects of increased salinity and flooding also affect individual plant species production in a highly variable manner based on species adaptations to these common estuarine stressors. Increased salinity and increased flooding have been shown to cause dieback of the oligohaline (<5 psu) wetland species *Sagittaria lancifolia* (Webb and Mendelssohn 1996). Plant species composition is altered due to storm events such as hurricane Katrina, when pulses of high salinity water reach normally freshwater and low salinity wetlands, causing declines of salt-intolerant species. For example, porewater salinity >1.5 psu has been shown to lead to the localized decline of *P. hemitomon* (Willis and Hester 2004).

The effects of hurricanes Katrina and Rita on wetland hydrology appear to be a major factor influencing wetland production along the estuarine gradient. For example, the accordion-like folding of the wetland surface at site N_2 greatly altered the hydrology of the site by increasing elevation by up to a meter, with the height and frequency of the folds directly related to the force the storm surge had against the root mat (Cahoon 2006). The effects of this change are evident not only when comparing productivity levels at the two Near sites but also when assessing species composition among sites.

Following widespread disturbance of wetlands in the upper Breton Sound estuary by hurricanes Katrina and Rita in 2005, the diversion structure was operated at high flow to encourage re-growth of wetlands degraded during the storms, with a goal to reversing this loss. However, after the passage of hurricanes Ike and Gustav in 2008, there was further wetlands loss. Barras (2007a, b) and Barras et al. (2008) used historic aerial photos to show that hurricanes have commonly caused wetland disturbance and loss in the Mississippi delta, especially in low salinity and fresh wetlands. Hurricane Betsy in 1965, which followed a similar path as Katrina, caused wetland disturbance in upper Breton Sound in some of the same areas that Katrina did in 2005 (Morton and Barras 2011). Also, low salinity and fresh wetlands were impacted along the path of hurricane Katrina from the mouth of the Mississippi River to the north shore of Lake Pontchartrain. Wetlands can recover if an area is not disturbed too frequently by subsequent hurricanes (i.e., western Terrebonne—Hurricane Andrew). The strong hurricane impacts in the upper Breton Sound estuary during 2005–2008 led to widespread wetland disturbance with little time for wetland recovery. Thus, wetlands may slowly recover if hurricane frequency is not excessive in the next decade or two. The low aboveground biomass after the storms suggests that the repeated hurricanes inhibited recovery. However, a definitive conclusion will have to wait for more study.

There were measurable concentrations of soil porewater nutrients at most sites throughout the study. The results of porewater chemistry from this study are consistent with

Table 2 Reported aboveground and belowground productivity, grams per square meter per year values for *Spartina alterniflora* marshes (unless other species specified)

Sampling location	Height form/species	Above NPP	Below NPP	Source
North Carolina	Short	650 (Max–Min)	460 (Max–Min)	Stroud and Cooper 1968
North Carolina	Tall	1,300 (Max–Min)	500 (Max–Min)	Stroud 1976
Massachusetts	NR	420 (Valiela)	3,500	Valiela et al. 1976
Georgia	Short	1,350	2,020	Gallagher 1978
	Tall	3,700	2,110	
New Jersey	Short	500	2,300	Smith et al. 1979
Nova Scotia	NR	803	1,051	Livingstone and Patriquin 1981
Georgia	Medium *S. alterniflora*	2,840	4,780 (Smalley)	Schubauer and Hopkinson 1984
	Tall *S. cynosuroides*	3,080	4,628	
South Carolina	Short	1,272 (Shew et al. 1981)	5,445 (Smalley)	Dame and Kenny 1986
	Medium	775	–	
	Tall	2,460	2,363	
Brazil	NR	–	569	Lana et al. 1991
Louisiana	Short	1,821 (Smalley)	11,676	Darby and Turner 2008a
Louisiana	NR	831 (PSC)	–	Kaswadji et al. 1990
		831 (Milner–Hughes)		
		1,231 (Smalley)		
		1,873 (Wiegert-Evans)		
		1,437 (Lomnick)		
Michigan	*Typha latifolia*	1,604 (1978) (Multiple)	–	Dickerman et al. 1986
		1,284 (1979)		
South Carolina	NR	402 (Dickerman)	–	Morris and Haskin 1990
		1,042 (1985)		
		700 (1986)		
		621 (1987)		
		408 (1988)		
Louisiana	*Distichlis* s.	3,237 (Wiegert-Evans)	–	Hopkinson et al. 1978
	Juncus r.	3,416		
	Phragmites c.	2,318		
	Sagittaria f.	1,501		
	Spartina a.	2,658		
	Spartina c.	1,355		
	Spartina p.	6,043		
Louisiana	Tall	2,645 (Wiegert-Evans)	–	Kirby and Gosselink 1976
Louisiana	NR	1,527 (Smalley)	–	White et al. 1978
	NR	2,895 (Wiegert-Evans)		
	Spartina p.	1,342 (Smalley)		
	Spartina p.	1,428 (Wiegert-Evans)		
Mississippi	*Distichlis* s.	1,484	–	de la Cruz 1974
	Spartina p.	1,922		
	Juncus r.	1,697		
	Spartina a.	1,964		
Georgia	NR	2,883 (PSC)	–	Odum and Fanning 1973
Georgia	NR	973 (Smalley)	–	Smalley 1959
Louisiana	*Scirpus americanus*	561 (R1) (Max–Min)	6,506	This study
	Spartina patens	1,158 (R2)	5,776	
	Distichlis spicata			
	Panicum virgatum,	872 (N1)	14,485	

Table 2 (continued)

Sampling location	Height form/species	Above NPP	Below NPP	Source
	Polygonum sp.			
	Alternanthera philoxeroides			
	Sparnita patens			
	Vigna luteola	964 (N2)	6,028	
	Ipomoea sp.			
	Spartina patens	1,236 (M1)	7,453	
	Scirpus americanus	326 (M2)	7,346	
	Spartina alterniflora	1,123 (F1)	4,776	
	Spartina patens	392 (F2)	10,131	
	Juncus reomerianus			

Estimation method appears in parentheses next to reported value. Max–Min estimate reported for this study

PSC peak standing crop method, *NR* height form not reported

previous studies that found decreased nutrient concentrations with distance from the diversion structure (Lane et al. 1999, 2004), and are sometimes higher at the outer estuary due to input of Mississippi River water from the south (Lane et al. 1999). Soil porewater nutrient concentrations fluctuated greatly at the Near sites, while concentrations at the other sites remained fairly constant, reflecting the Caernarvon river diversion as a major, albeit variable, source of nutrients to the upper estuary (Lane et al. 1999, 2004; Mitsch et al. 2001; Wissel and Fry 2005). However, these nutrient amendments are not completely assimilated possibly due to variability in site-specific hydrology as well as differences in species composition. The lower production levels measured at site N_2 in comparison to site N_1 may be due to a number of factors, including limited contact of the wetland with the diverted water at site N_2 due to the hurricane-induced folding that raised the wetland surface to a higher elevation. The generally higher porewater salinity and sulfide concentrations measured early in the study likely reflect high salinity water brought in by the hurricanes that was then flushed out by the diversion.

Lower biomass at the Ref and Mid sites may be related to higher sulfide concentrations during the growing season. *S. alterniflora* biomass growth is negatively impacted by soluble sulfide levels greater than 1 mM (or 32 ppm) total soluble sulfide (Bradley and Dunn 1989; Koch and Mendelssohn 1989; Koch et al. 1990), while fresh marsh species are less tolerant of sulfide (Koch and Mendelssohn 1989). Sulfide concentrations exceeded 1 mM at sites R_2 and M_2 during March 2006, while concentrations were below the 1 mM threshold level at all other sites during this study. Mean sulfide concentrations were highest at sites R_2 (0.58 mM) and F_1 (0.45 mM), and both the Mid sites (0.46 and 0.34 mM). These same four sites also had the lowest redox potentials. These results suggest that the influence of saltwater from the Gulf combined with more reduced soil

conditions led to the production of sulfides at these sites. However, the sulfide concentrations present at each site were not so high as to exceed the 1 mM threshold for *S. alterniflora*.

In general, few of the response variables (redox, sulfide, nutrients, biomass productivity, and accretion) varied consistently along the gradient away from the diversion. It is interesting to note that the inputs of freshwater, nutrients, and sediments did not translate into a difference in variables that affected marsh health and survival. It may be that environmental factors affecting streamside marshes are not extreme enough to negatively impact marsh health.

The key to wetland recovery and stability is the net accumulation of organic matter and mineral sediments in wetland soils, which contribute to increased wetland surface elevation relative to local water level rise. We investigated site-specific above- and belowground decomposition, and found no strong impact of the Caernarvon diversion on decomposition. Numerous factors determine decomposition rates in wetland environments (Gallagher 1978; Rybczyk et al. 1996; Mendelssohn et al. 1999), but for this study, nutrient availability, flooding frequency, and redox potential were likely the main drivers. Cotton strip decomposition rates at the Ref sites were slightly lower through most of the depth profile, but this difference was only significant at the lowest measured depth. At this depth the decomposition rate at the Ref sites was less than the Near site, but not the Far and Mid sites. Previous studies have shown high levels of interstitial nutrients can increase cotton strip decomposition rates (Mendelssohn et al. 1999; Laursen 2004). Interstitial phosphate and ammonium concentrations were higher at the Near sites, but other factors may have been at play as well. The Near sites also had the highest redox potentials that can also increase decomposition (Neckles and Neill 1994). The combination of increased interstitial nutrient concentrations and a high redox potential could

have accounted for the higher cotton strip decomposition rates at depth at the Near sites; however, these rates were only slightly higher than the other non-reference sites.

As with cotton strip decomposition, belowground litter-bag decomposition did not differ greatly between locations when looking at local and mixed materials separately. The elevated interstitial nutrient concentrations did not have the same impact on belowground litterbag decomposition as it did on the cotton strip decomposition. This agrees with recent studies that found no significant difference in below-ground litterbag decomposition between reference sites and sites receiving nitrogen fertilization (Ibañez et al. 1999; Laursen 2004). However, the local material at the Far sites did decompose more rapidly than the mixed material, indicating that material has some inherent trait that leads to increased decomposition. Pooling of the local and mixed materials showed that the first 6 months of decomposition did not yield any noticeable differences between the locations, but the second 6-month period did. Decomposition rates decreased at the Ref, Near and Mid sites, but not at the Far sites. The higher levels of decomposition of the local material at the Far sites may have accounted for this.

Aboveground litterbag decomposition showed no differences between local and mixed materials at any site. However, the Ref and Near sites had lower decomposition rates when compared to the Far and Mid sites. Water level data showed that the Near, Far, and Ref sites were flooded much less than the Mid site during the growing season. This agrees with earlier research that showed aboveground litter decomposes faster when flooded than dry (Davis and van der Valk 1978; Neckles and Neill 1994; Flores et al. 1987). The balance of net primary production, both aboveground and belowground, is an important indicator of net organic matter accumulation given that there was little difference in decomposition among sites.

Our results and findings of others provide insights into the factors that led to marsh loss during hurricane Katrina. Nutrient addition studies showing reduced belowground productivity and enhanced decomposition at high nutrient loading rates (Darby and Turner 2008b, c; Turner 2010) have been cited to suggest that excess nutrients from the Caernarvon river diversion made wetlands in the upper Breton Sound estuary more susceptible to hurricane damage (Kearney et al. 2011). Floating *P. hemitomon* wetlands that received seasonal inputs from the Atchafalaya River had more reduced soils, increased porewater sulfide and inorganic nutrient levels, and internally generated alkalinity, and Swarzenski et al. (2008) suggested that this led to decreased root production and increased decomposition. However, Sasser et al. (2004) reported that fertilized *P. hemitomon* marshes in the same area had higher above- and below-ground biomass when they were protected from grazing by nutria.

Our results indicate that above- and belowground biomass and productivity in streamside marshes were high in the Breton Sound estuary. Belowground litter decomposition did not substantially vary and belowground cellulose (cotton strip) decomposition, although slightly greater at the stations receiving Mississippi River water than reference sites, showed considerable variation and only significantly differed at specific depths. It has been suggested that high nutrient levels made marshes more susceptible to hurricanes (Kearney et al. 2011) based on results showing high nutrient loadings may be harmful to wetlands (Darby and Turner 2008b) when nitrogen loadings were greater than 350 gN/m^2/year. Such high loadings are rare, however, even for wetlands receiving nutrient-rich municipal effluent (Day et al. 2004; Hunter et al. 2008). Nutrient loadings typical of natural systems are an order of magnitude lower (Boynton et al. 1995; Faulkner and Richardson 1989; Josefson and Rasmussen 2000), including the Caernarvon river diversion, which loads the upper Breton Sound estuary, the area that included all our sites, with <25 g N/m^2/year (Lane et al. 1999; Hyfield et al. 2008). Some studies have shown that in marshes with high nutrient loading, belowground production or accretion is not affected (Anisfeld and Hill 2012; Fox et al. 2012). In addition, inorganic nutrients in surface water of interior marshes in the upper estuary, which included our near sites, were extremely low (Day et al. 2009). These findings indicate that nitrogen levels were low, especially in interior marshes, over much of the area that experienced wetland loss during the hurricane.

The high belowground biomass we measured at streamside marshes is likely related to higher sediment input to these marshes. Sediment input to marshes affected by the Caernarvon diversion was highest close to the head of the estuary and on the edge of the channels (Wheelock 2003), a finding consistent with other studies (Leonard 1997; Reed 1992; Day et al. 2011a, b). More than 4 km^2 of new wetlands have developed in upper Breton Sound since 2005 due to sediment input from the river (Henkel et al. 2011).

Our results in conjunction with other observations and findings at Caernarvon provide insights on the future management of the diversion. Snedden et al. (2007) reported that the Mississippi River carries substantially more suspended sediment during increasing and peak river stages compared to falling stage, and thus, the diversion structure should be operated during these periods to optimize sediment delivery to wetlands to promote wetland soil consolidation and plant growth (i.e., Day et al. 2011a). Winter cold front passages along the coast result in alternate flooding and drainage of wetlands (Moeller et al. 1993). The few days prior to a frontal passage are dominated by southerly winds that push Gulf waters into the estuary, raising water levels and flooding wetlands. As the cold front passes, northwesterly winds predominate that push water back towards the Gulf, rapidly lowering water levels (Moeller et al. 1993). Diversions

could be coordinated with frontal passages by pulsing large volumes of river water into the estuary during the frontal setup when there are optimal conditions for overland flow and the transport of nutrients and sediments to interior wetlands. Closing the diversion following frontal passage would lead to better drainage of the marshes reducing flooding stress. And finally, operating the diversion at high discharge rates in the winter and early spring dormant season would lessen the impact on vegetation.

In summary, belowground biomass was generally high, variable among study sites, and increased at all sites over the course of this study, suggesting vegetation recovery following hurricane Katrina. In contrast, aboveground biomass was significantly lower than measured several years previously using the EOSL method, possibly in response to hurricane Katrina. The effect of the Caernarvon river diversion resulted in relatively high porewater nutrient concentrations, which did not appear to significantly influence decomposition rates, while high sulfide levels at the Mid and Far sites were indicative of the influence of saltwater from the Gulf of Mexico. Long duration flooding at the Mid site likely affected vegetation health. In conclusion, it seems clear that a number of factors led to wetland deterioration in the upper Breton Sound estuary during hurricanes Katrina and Rita. These possibly include high nutrients, excessive inundation, herbivory, and low mineral sediment input. It is unlikely that nitrogen input from the river alone can explain the impacts of the hurricanes. Further research is needed to clearly determine the relative importance of these factors. A pulsed discharge regime, operating mainly in the cooler month, that optimized sediment load while reducing total volume of diverted water would likely be the most beneficial management plan to reduce inundation while increasing sediment input into wetlands.

References

Anisfeld, S.C., and T.D. Hill. 2012. Fertilization effects on elevation change and belowground carbon balance in a Long Island Sound tidal marsh. *Estuaries & Coasts* 35: 201–211. doi:10.1007/s12237-011-9440-4.

APHA, 1985. Greenberg, A.E., R.R. Trussell, L.S. Clesceri & M.A.H. Franson (eds.), Standard Methods for the Examination of Water and Wastewater, 16th Ed., American Public Health Association, Washington, D.C.

Barras, J.A. 2007a. Land area changes in coastal Louisiana after Hurricanes Katrina and Rita, in Farris, G.S., Smith, G.J., Crane, M. P., Demas, C.R., Robbins, L.L., and Lavoie, D.L., eds., Science and the storms: the USGS response to the hurricanes of 2005: U.S. Geological Survey Circular 1306., pp. 98–113, http://pubs.usgs.gov/circ/1306/pdf/c1306.

Barras, J.A., 2007b, Satellite images and aerial photographs of the effects of Hurricanes Katrina and Rita on coastal Louisiana: U.S. Geological Survey Data Series 281, http://pubs.usgs.gov/ds/2007/281.

Barras, J.A., Bernier, J.C., and Morton, R.A., 2008, Land area change in coastal Louisiana—a multidecadal perspective (from 1956 to 2006): U.S. Geological Survey Scientific Investigations Map 3019, scale 1:250,000, 14 p. pamphlet, available online at http://pubs.usgs.gov/sim/3019.

Bass, A.S., and R.E. Turner. 1977. Relationships between salt marsh loss and dredged canals in three Louisiana Estuaries. *Journal of Coastal Research* 13: 895–903.

Barry, J. 1998. *Rising tide: the great Mississippi flood of 1927 and how it changed America*. New York: Simon and Schuster.

Bazzaz, F.A. 1997. Allocation of resources in plants: State of the science and critical questions. In *Plant resource allocation*, ed. F.A. Bazzaz and J. Grace, 1–38. San Diego: Academic Press.

Blum, M., and H. Roberts. 2009. Drowning of the Mississippi delta due to insufficient sediment supply and global sea-level rise. *Nature Geoscience* 2: 488–491.

Blum, M., and H. Roberts. 2012. The Mississippi delta region: past, present, and future. *Annual Review of Earth and Planetary Sciences* 40: 655–683.

Boesch, D.F., L. Shabman, L.G. Antle, J.W. Day Jr., R.G. Dean, G.E. Galloway, C.G. Groat, S.B. Laska, R.A. Luettich Jr., W.J. Mitsch, N.N. Rabalais, D.J. Reed, C.A. Simenstad, B.J. Streever, R.B. Taylor, R.R. Twilley, C.C. Watson, J.T. Wells, D.F. Whigham. 2006. *A new framework for planning the future of coastal Louisiana after the hurricanes of 2005*. Cambridge: University of Maryland Center for Environmental Science. by Working Group for Post-Hurricane Planning for the Louisiana Coast.

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. Journal of Coastal Research Special Issue No. 20.

Boesch, D.F. 1996. Science and management in four U.S. coastal ecosystems dominated by land–ocean interactions. *Journal of Coastal Conservation* 2: 103–114.

Boynton, W.R., J.H. Garber, R. Summers, and W.M. Kemp. 1995. Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake bay and selected tributaries. *Estuaries* 18: 285–314.

Bradley, P.M., and E.L. Dunn. 1989. Effects of sulfide on the growth of 3 salt marsh halophytes of the southeastern United States. *American Journal of Botany* 76: 1707–1713.

Burdick, D.M., I.A. Mendelssohn, and K.L. McKee. 1989. Live standing crop and metabolism of the marsh grass *Spartina patens* as related to edaphic factors in a brackish, mixed marsh community in Louisiana. *Estuaries* 12: 195–204.

Cahoon, D.R. 2006. A review of major storm impacts on coastal wetland elevations. *Estuaries & Coasts* 29: 889–898.

Cahoon, D.R., and R.E. Turner. 1989. Accretion and canal impacts in a rapidly subsiding wetland II. Feldspar marker horizon technique. *Estuaries* 12: 260–268.

Chapin, F.S. 1980. The mineral nutrition of wild plants. *Annual Review of Ecology & Systematics* 11: 233–260.

Chatry, M., and D. Chew. 1985. Freshwater diversion in coastal Louisiana: recommendations for development of management criteria. 4th Coastal Marsh and Estuary Mgt. Symposium, pp. 71–84.

Colten, C. (ed.). 2000. *Transforming New Orleans and its Environs*, 272. Pittsburgh: University of Pittsburgh Press.

Dame, R.F., and P.D. Kenny. 1986. Variability of *Spartina alterniflora* primary production in the euhaline North Inlet estuary. *Marine Ecology Progress Series* 32: 71–80.

Darby, F.A., and R.E. Turner. 2008a. Below- and aboveground *Spartina alterniflora* production in a Louisiana salt marsh. *Estuaries & Coasts* 31: 223–231.

Darby, F.A., and R.E. Turner. 2008b. Below- and aboveground biomass of *Spartina alterniflora*: response to nutrient addition in a Louisiana salt marsh. *Estuaries & Coasts* 31: 326–334.

Darby, F.A., and R.E. Turner. 2008c. Effects of eutrophication on salt marsh root and rhizome accumulation. *Marine Ecology Progress Series* 363: 63–70.

Davis, C.B., and A.G. van der Valk. 1978. Litter decomposition in prairie glacial marshes. In *Freshwater wetlands*, ed. R.E. Good, D.F. Whigham, and R.L. Simpson, 99–113. New York: Academic Press Inc.

Davis, D.W. 2000. Historical perspective on crevasses, levees, and the Mississippi River. In *Transforming New Orleans and its Environs*, ed. C.E. Colten, 84–106. Pittsburgh: University of Pittsburgh Press.

Day, J.W., 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, T.J. Wells, and D.F. Whigham. 2007. Restoration of the Mississippi Delta: lessons from hurricanes Katrina and Rita. *Science* 315: 1679–1684.

Day, J.W., J. Martin, L. Cardoch, and P. Templet. 1997. System functioning as a basis for sustainable management of deltaic ecosystems. *Coastal Management* 25: 115–153.

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. *Estuaries* 23: 425–438.

Day, J.W., J. Jae-Young Ko, D. Rybczyk, R. Sabins, G. Bean, C. Berthelot, L. Brantley, W. Cardoch, J.N. Conner, A.J. Day, S. Englande, E. Feagley, R. Hyfield, J. Lane, J. Lindsey, E.Reyes Mitsch, and R. Twilley. 2004. The use of wetlands in the Mississippi delta for wastewater assimilation: a review. *Ocean & Coastal Management* 47: 671–691.

Day, J.W., J.E. Cable, J.H. Cowan, R. DeLaune, B. Fry, H. Mashriqui, D. Justic, P. Kemp, R.R. Lane, J. Rick, S. Rick, L.P. Rozas, G. Snedden, E. Swenson, R.R. Twilley, and B. Wissel. 2009. The impact of pulsed re-introduction of river water on a Mississippi delta basin. *Journal of Coastal Research* 54: 225–243.

Day, J.W., G.P. Kemp, D.J. Reed, D.R. Cahoon, R.M. Boumans, J.M. Suhayda, and R. Gambrell. 2011a. Vegetation death and rapid loss of surface elevation in two contrasting Mississippi delta salt marshes: the role of sedimentation, autocompaction and sea-level rise. *Ecological Engineering* 37: 229–240. doi:10.1016/j.ecoleng.2010.11.021.

Day, J.W., C. Ibanez, F. Scarton, D. Pont, P. Hensel, J.N. Day, and R. Lane. 2011b. Sustainability of Mediterranean deltaic and lagoon wetlands with sea-level rise: the importance of river input. *Estuaries & Coasts* 34: 483–493. doi:10.1007/s12237-011-9390-x.

de la Cruz, A.A. 1974. Primary productivity of coastal marshes in Mississippi. *Gulf Research Reports* 4: 351–356.

DeLaune, R.D., A. Jugsujinda, G.W. Peterson, and W.H. Patrick. 2003. Impact of Mississippi River freshwater reintroduction on enhancing marsh accretionary processes in a Louisiana estuary. *Estuarine, Coastal & Shelf Science* 58: 653–662.

Delaune, R.D., and S.R. Pezeshki. 2003. The role of soil organic carbon in maintaining surface elevation in rapidly subsiding U.S. Gulf of Mexico coastal marshes. *Water Air Soil Pollution* 3: 167–179.

DeLaune, R.D., C.J. Smith, and W.H. Patrick. 1983. Relationship of marsh elevation, redox potential, and sulfide to *Spartina alterniflora* productivity. *Journal of the Soil Science Society of America* 47: 930–935.

Dickerman, J.A., A.J. Stewart, and R.G. Wetzel. 1986. Estimates of net annual aboveground production: Sensitivity to sampling frequency. *Ecology* 67: 650–659.

Faulkner, S.P., and C.J. Richardson. 1989. Physical and chemical characteristics of freshwater wetland soils. In *Constructed wetlands for wastewater treatment*, ed. D.A. Hammer, 41–72. Chelsea: Lewis Publishers.

Fisher, K.J. 2003. Response of *Panicum hemitomon* Shultes to environmental change. M. S. thesis. Southeastern Louisiana University. Hammond, Louisiana

Fitter, A. 2002. In *Plant roots: the hidden half*, ed. Y. Waisel, A. Eshel, and U. Kafkafi. New York: Marcel Dekker, Inc.

Flores, F., J. Day, and R. Breseño. 1987. Structure, litter fall, decomposition, and detritus dynamics of mangroves in a Mexican coastal lagoon with an ephemeral inlet. *Marine Ecology Progress Series* 35: 83–90.

Fox, L., I. Valiela, and E. Kinney. 2012. Vegetation cover and elevation in long-term experimental nutrient-enrichment plots in Great Sippewissett salt marsh, Cape Cod, Massachusetts: implications for eutrophication and sea-level rise. *Estuaries & Coasts* 35: 445–458.

Gallagher, J.L. 1978. Decomposition processes: summary and recommendations. In *Freshwater wetlands*, ed. R.E. Good, D.F. Whigham, and R.L. Simpson, 145–151. New York: Academic Press Inc.

George, J.R., K.M. Blanchet, R.M. Gettle, D.R. Buxton, and K.J. Moore. 1995. Yield and botanical composition of legume-interseeded vs. nitrogen-fertilized switchgrass. *Agronomy Journal* 87: 1147–1153.

Gornitz, V., S. Lebedeff, and J. Hansen. 1982. Global sea level trend in the past century. *Science* 215: 1611–1614.

Gosselink, J.G. 2001. Comments on "Wetland loss in the northern Gulf of Mexico: multiple working hypothesis." by R.E. Turner. 1997 Estuaries 20: 1–13. *Estuaries* 24: 636–639.

Grime, J.P. 1977. Evidence for the existence of three primary strategies in plants and its relevance to ecological and evolutionary theory. *The American Naturalist* 111(982): 1169–1194.

Gross, M.F., M.A. Hardisky, P.L. Wolf, and V. Klemas. 1991. Relationship between aboveground and belowground biomass of *Spartina alterniflora* (Smooth Cordgrass). *Estuaries* 14: 180–191.

Harmon, M.E., K.J. Nadelhoffer, and J.M. Blair. 1999. Measuring decomposition, nutrient turnover, and stores in plant litter. In *Standard soil methods for long-term ecological research*, ed. G.P. Robertson, C.S. Beldsoe, D.C. Coleman, and P. Sollins, 202–240. New York: Oxford University Press.

Henkel, T.K., A. Baker, J. Lopez, and E. Boyd. 2011. *Accelerated wetland growth and cypress planting of the Caernarvon delta complex, Big Mar pond—southeast Louisiana.*, 7. New Orleans: Lake Pontchartrain Basin Foundation.

Hester, M.W., I.A. Mendessohn, and K.L. McKee. 1996. Intraspecific variation in salt tolerance and morphology in the coastal grass *Spartina patens* (Poaceae). *American Journal of Botany* 83: 1521–1527.

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

Hunter, R., R. Lane, J.W. Day, J. Lindsey, J.N. Day, and M. Hunter. 2008. Nutrient removal and loading rate analysis of Louisiana forested wetlands assimilating treated municipal effluent. *Environmental Management* 44: 865–873.

Hyfield, E., J. Day, J. Cable, and D. Justic. 2008. The impacts of re-introducing Mississippi River water on the hydrologic budget and

nutrient inputs of a deltaic estuary. *Ecological Engineering* 32: 347–359.

Ibañez, C., J. Day, and D. Pont. 1999. Primary production and decomposition of wetlands of the Rhone Delta, France: interactive impacts of human modifications and relative sea level rise. *Journal of Coastal Research* 15: 717–731.

IPCC (Intergovernmental Panel on Climate Change). 2007. *Climate Change 2001: The Scientific Basis Contribution of Working Group 1 to the Fourth Assessment Report*. Cambridge: Cambridge University Press.

Izdepski, C.W., J.W. Day, C.E. Sasser, and B. Fry. 2009. Early floating marsh establishment and growth dynamics in a nutrient amended wetland in the lower Mississippi delta. *Wetlands* 29: 1004–1013.

Josefson, A.B., and B. Rasmussen. 2000. Nutrient retention by benthic macrofaunal biomass of Danish estuaries: importance of nutrient load and residence time. *Estuarine, Coastal & Shelf Science* 50: 205–216.

Kaswadji, R.F., J.G. Gosselink, and R.E. Turner. 1990. Estimation of primary production using five different methods in a *Spartina alterniflora* salt marsh. *Wetlands Ecology and Management* 1: 57–64.

Kearney, M.S., J.C.A. Riter, and R.E. Turner. 2011. Freshwater river diversions for marsh restoration in Louisiana: twenty-six years of changing vegetative cover and marsh area. *Geophysical Research Letters* 38: L16405. doi:10.1029/2011GL047847.

Kesel, R.H. 1988. The decline in the suspended load of the Lower Mississippi River and its influence on adjacent wetlands. *Environmental & Geological Water Science* 11: 271–281.

Kesel, R.H. 1989. The role of the lower Mississippi River in wetland loss in southeastern Louisiana, USA. *Environmental & Geological Water Science* 13: 183–193.

Kirby, C.J., and J.G. Gosselink. 1976. Primary production in a Louisiana Gulf Coast *Spartina alterniflora* marsh. *Ecology* 57: 1052–1059.

Koch, M.S., and I.A. Mendelssohn. 1989. Sulfide as a soil phytotoxin —differential responses in 2 marsh species. *Journal of Ecology* 77: 565–578.

Koch, M.S., I.A. Mendelssohn, and K.L. McKee. 1990. Mechanism for the hydrogen sulfide-induced growth limitation in wetland macrophytes. *Limnology & Oceanography* 35: 399–408.

Lana, P., C. Guiss, and S.T. Disaro. 1991. Seasonal variation of biomass and production dynamics for above- and belowground components of a *Spartina alterniflora* marsh in the euhaline sector of Paranaguá (SE Brazil). *Estuarine, Coastal & Shelf Science* 32: 231–241.

Lane, R.R., J.W. Day, and B. Thibodeaux. 1999. Water quality analysis of a freshwater diversion at Caernarvon, Louisiana. *Estuaries* 22: 327–336.

Lane, R.R., J.W. Day, D. Justic, E. Reyes, B. Marx, J.N. Day, and E. Hyfield. 2004. Changes in stoichiometric Si, N and P ratios of Mississippi River water diverted through coastal wetlands to the Gulf of Mexico. *Estuarine, Coastal & Shelf Science* 60: 1–10.

Lane, R.R., J.W. Day Jr., B. Marx, E. Hyfield, J. Day, and Enrique Reyes. 2007. The effects of riverine discharge on temperature, salinity, suspended sediment and chlorophyll a in a Mississippi delta estuary measured using a flow-through system. *Estuarine, Coastal & Shelf Science* 74: 145–154.

Lane, R.R., J.W. Day Jr., and J.N. Day. 2006. Marsh elevation change, accretion, and subsidence at three Louisiana coastal wetlands receiving diverted Mississippi River water. *Wetlands* 26: 1130–1142.

Laursen, K.R. 2004. The effects of nutrient enrichment on the decomposition of belowground organic matter in a *Saggitaria lancifolia* dominated oligohaline marsh. M.S. Thesis. Louisiana State University, Baton Rouge, LA. 69 p

Leonard, L. 1997. Controls of sediment transport and deposition in an incised mainland marsh basin, southeastern North Carolina. *Wetlands* 17: 263–274.

Leuliette, E.W., R.S. Nerem, and G.T. Mitchum. 2004. Calibration of TOPEX/Poseidon and Jason altimeter data to construct a continuous record. *Marine Geodesy* 27: 79–94.

LDNR. 1998. *Coast 2050: toward a sustainable coastal Louisiana*, 161. Baton Rouge: Louisiana Coastal Wetlands Conservation and Restoration Task Force and the Wetlands Conservation and Restoration Authority. Louisiana Department of Natural Resources.

Livingstone, D.C., and D.G. Patriquin. 1981. Belowground growth of *Spartina alterniflora* Loisel: Habitat, functional biomass and non-structural carbohydrates. *Estuarine, Coastal and Shelf Science* 12: 579–588.

McKee, K.L., I.A. Mendelssohn, and M.W. Hester. 1988. Reexamination of pore water sulfide concentrations and redox potentials near the aerial roots of *Rhizophora mangle* and *Avicennia germinans*. *American Journal of Botany* 75(9): 1352–1359.

McLauglin, S.B., and L.A. Kszos. 2005. Development of switchgrass (*Panicum virgatum*) as a bioenergy feedstock in the United States. *Biomass and Bioenergy* 28: 515–535.

Mendelssohn, I.A., B.K. Sorrell, H. Brix, H. Schierup, B. Lorenzen, and E. Maltby. 1999. Controls on soil cellulose decomposition along a salinity gradient in a *Phragmites australis* wetland in Denmark. *Aquatic Botany* 64: 381–398.

Milner, C., and R.E. Hughes. 1968. *Method for the measurement of the primary production of grasslands*. Oxford: Blackwell Scientific Publications. IBP Handbook No. 6.

Mitsch, W., J. Day, J. Gilliam, P. Groffman, D. Hey, G. Randall, and N. Wang. 2001. Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River basin: strategies to counter a persistent problem. *BioScience* 51(5): 373–388.

Moeller, C.C., O.K. Huh, H.H. Roberts, L.E. Gumley, and W.P. Menzel. 1993. Response of Louisiana coastal environments to a cold front passage. *Journal of Coastal Research* 9: 434–447.

Morris, J.T., and B. Haskin. 1990. A 5-yr Record of aerial Primary production and stand characteristics of *Spartina alterniflora*. *Ecology* 71: 2209–2217.

Morton, R.A., and J.A. Barras. 2011. Hurricane impacts on coastal wetlands: a half-century record of storm-generated features from southern Louisiana. *Journal of Coastal Research* 27: 27–43.

Morton, R.A., N.A. Buster, and M.D. Krohn. 2002. Subsurface controls on historical subsidence rates and associated wetland loss in southcentral Louisiana. *Gulf Coast Association of Geological Societies Transactions* 52: 767–778.

Mossa, J. 1996. Sediment dynamics in the lowermost Mississippi River. *Engineering Geology* 45: 457–479.

Neckles, H.A., and C. Neill. 1994. Hydrologic control of litter decomposition in seasonal flooded prairie marshes. *Hydrobiologia* 286: 155–165.

Odum, E.P., and M.E. Fanning. 1973. Comparison of the productivity of *Spartina alterniflora* and *Spartina cynosuroides* in Georgia coastal marshes. *Georgia Journal of Science* 31: 1–12.

Patrick Jr., W.H., R.P. Gambrell, and S.P. Faulkner. 1996. Redox measurements in soil. In *Methods of soil analysis, part 3: chemical methods*, ed. D.L. Sparks, 1255–1273. Madison: Soil Science Society of America.

Penland, S., and K.E. Ramsey. 1990. Relative sea-level rise in Louisiana and the Gulf of Mexico: 1908–1988. *Journal of Coastal Research* 6: 323–342.

Penland, S., K.E. Ramsey, R.A. McBride, J.T. Mestayer, and K.A. Westphal. 1988. *Relative sea level rise and delta plain development in the Terrebonne Parish Region. Coastal Geology Technical Report*, 121. Baton Rouge, Louisiana: Louisiana Geological Survey.

Pezeshki, S.R., and R.D. DeLaune. 1991. A comparative study of aboveground productivity of dominant United States Gulf coast marsh species. *Journal of Vegetation Science* 2: 331–338.

Pfeffer, W.T., J.T. Harper, and S. O'Neel. 2008. Kinematic constraints on glacier contributions to 21st-century sea-level rise. *Science* 321: 1340–1343.

Poore, M.E.D. 1955. The use of phytosociological methods in ecological investigations. The Braun-Blanquet System. *Journal of Ecology* 43: 226–244.

Rahmstorf, S. 2007. A semi-empirical approach to projecting future sea-level rise. *Science* 315: 368–370.

Reed, D.J. 1992. Effects of weirs on sediment deposition in Louisiana coastal marshes. *Environmental Management* 16: 55–65.

Rivera-Monroy, V., E. Castaneda, and R.R. Twilley. 2011. Mangrove forest functioning in response to hurricane disturbance. Final report. Louisiana State University. Sub award # 4126-LSU-TU-L600. National Institute for Climatic Change Research (NICCR) Coastal Center at Tulane University. Office of Biological and Environmental Research (BER), U.S. Department of Energy (DOE).

Roberts, H.H. 1997. Dynamic changes of the holocene Mississippi River delta plain: the delta cycle. *Journal of Coastal Research* 13: 605–627.

Rohling, E.J., K. Grant, Ch Hemleben, M. Siddall, B.A.A. Hoogakker, M. Bolshaw, and M. Kucera. 2008. High rates of sea-level rise during the last interglacial period. *Nature Geoscience* 1: 38–42.

Rybczyk, J.M., G. Garson, and J.W. Day Jr. 1996. Nutrient enrichment and decomposition in wetland ecosystems: models, analyses and effects. *Current Topics in Wetland Biogeochemistry* 2: 52–72.

Sall, J., L. Creighton, and A. Lehman. 2005. *JMP start statistics: a guide to statistical and data analysis using JMP and JMP IN Software*, 560. Belmont: Brooks/Cole-Thompson Learning.

SAS Institute Inc. 2003. *SAS/STAT version 9*. Cary: SAS Institute Inc.

Sasser, C.E., G.O. Holm, J.M. Visser, and E.M. Swenson. 2004. *Thin-mat floating marsh enhancement demonstration project, TE-36. Final Report to Louisiana Department of Natural Resources by Coastal Ecology Institute, School of the Coast & Environment*. Baton Rouge: Louisiana State University.

Schubauer, J.P., and C.S. Hopkinson. 1984. Above- and belowground emergent macrophyte production and turnover in a coastal marsh ecosystem, Georgia. *Limnology and Oceanography* 29: 1052–1063.

Scruton, P.C. 1960. Delta building and the deltaic sequence. Recent sediments, NW Gulf Coast of Mexico AAPG Symposium, pp. 82–102.

Shaffer, G., J. Day, S. Mack, P. Kemp, I. van Heerden, M. Poirrier, K. Westphal, D. FitzGerald, A. Milanes, C. Morris, R. Bea, and S. Penland. 2009. The MRGO navigation project: a massive human-induced environmental, economic, and storm disaster. *Journal of Coastal Research* 54: 206–224.

Shew, D.M., R.A. Linthurst, and E.D. Seneca. 1981. Comparison of production computation methods in a southeastern North Carolina *Spartina alterniflora* salt marsh. *Estuaries* 4: 97–109.

Slocum, M.G., J. Roberts, and I.A. Mendelssohn. 2009. Artist canvas as a new standard for the cotton-strip assay. *Journal of Plant Nutrition & Soil Science* 172: 71–74.

Smalley, A.E. 1959. *The role of two invertebrate populations, Littorina irrorata and Orchelimum ji dicinium in the energy flow of a salt marsh ecosystem*, 126. Athens: Ph.D. Thesis. University of Georgia.

Smith, K., R.R. Good, and N.F. Good. 1979. Production dynamics for above- and belowground components of a New Jersey *Spartina alterniflora* tidal marsh. *Estuarine, Coastal & Shelf Science* 9: 189–201.

Snedden, G.A., J.E. Cable, C. Swarzenski, and E. Swenson. 2007. Sediment discharge into a subsiding Louisiana deltaic estuary through a Mississippi River diversion. *Estuarine Coastal & Shelf Science* 71: 181–193.

Spalding, E.A., and M.W. Hester. 2007. Interactive effects of hydrology and salinity on oligohaline plant species productivity: implications of relative sea-level rise. *Estuaries & Coasts* 30: 214–225.

Steyer, G.D. 2008. Landscape analysis of vegetation change in coastal Louisiana following Hurricanes Katrina and Rita. Ph.D. Dissertation. Louisiana State University, Baton Rouge, LA, p. 145

Stroud, L.M. 1976. Net primary production of belowground material and carbohydrate patterns of two height forms of *Spartina alterniflora* in two North Carolina marshes. Ph.D. dissertation. North Carolina State University. Raleigh, NC, p. 140

Stroud, L.M., and A.W. Cooper. 1968. *Color infrared aerial photographic interpretation and net primary productivity of a regularly flooded North Carolina marsh. Report 14*, 1–86. Raleigh: Water Resource Institute.

Swarzenski, C., T. Doyle, B. Fry, and T. Hargis. 2008. Biogeochemical response of organic-rich freshwater marshes in the Louisiana delta plain to chronic river water influx. *Biogeochemistry* 90: 49–63.

Swenson, E.M., and R.E. Turner. 1987. Spoil banks: effects on a coastal marsh water-level regime. *Estuarine Coastal & Shelf Science* 24: 599–609.

Symbula, M., and F.P. Day Jr. 1988. Evaluation of two methods for estimating belowground production in a freshwater swamp. *American Midland Naturalist* 120: 405–415.

Templet, P.H., and K.J. Meyer-Arendt. 1988. Louisiana wetland loss: a regional water management approach to the problem. *Environmental Management* 12: 181–192.

Tilman, D. 1985. The resource ratio hypothesis of succession. *The American Naturalist* 125: 827–852.

Tornqvist, T.E., C. Paola, G. Parker, K. Liu, D. Mohrig, J.M. Holbrook, and R.R. Twilley. 2007. Comment on "Wetland sedimentation from Hurricanes Katrina and Rita". *Science* 316: 5822.

Turner, R.E. 2010. Beneath the salt marsh canopy: loss of soil strength with increasing nutrient loads. *Estuaries & Coasts* 34: 1–10. doi:10.1007/s12237-010-9341-y.

Turner, R.E., and N.N. Rabalais. 1991. Changes in Mississippi River water quality this century. *Bioscience* 41(3): 140–147.

USCOE. 2004. *Louisiana Coastal Area (LCA), Louisiana Ecosystem Restoration Study*. New Orleans: United States Corps of Engineers.

Valiela, I., J.M. Teal, and N.Y. Persson. 1976. Production and dynamic experimentally enriched salt marsh vegetation: belowground biomass. *Limnology & Oceanography* 21: 245–252.

Vermeer, M. and S. Rahmstorf. 2009. Global sea level linked to global temperature. Proceedings of the National Academy of Sciences 106(51): 21527–21532. doi: 10.1073_pnas.0907765106

Webb, E.C., and I.A. Mendelssohn. 1996. Factors affecting vegetation dieback of an oligohaline marsh in coastal Louisiana: field manipulation of salinity and submergence. *American Journal of Botany* 83(11): 1429–1434.

Welder, F.A. 1959. *Processes of deltaic sedimentation in the lower Mississippi River. Coastal Studies Institute Technical Report 12*. Baton Rouge: Louisiana State University.

Wheelock, K.V. 2003. Pulsed river flooding effects on sediment deposition in Breton Sound Estuary, Louisiana. Baton Rouge, Louisiana: Louisiana State University, M.S. thesis, p. 159.

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

Willis, J.M., and M.W. Hester. 2004. Interactive effects of salinity, flooding, and soil type on *Panicum hemitomon*. *Wetlands* 24: 43–50.

Wissel, B., and B. Fry. 2005. Tracing Mississippi River influences in estuarine food webs of coastal Louisiana. *Oecologia* 144: 659–672.