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Marsh:water Column Interactions in Two Louisiana Estuaries. Flux Measurements and Conceptual Implications.

Daniel Louis Childers

Louisiana State University and Agricultural & Mechanical College

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**Marshwater column interactions in two Louisiana estuaries.
Flux measurements and conceptual implications**

Childers, Daniel Louis, Ph.D.

The Louisiana State University and Agricultural and Mechanical Col., 1989

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300 N. Zeeb Rd.
Ann Arbor, MI 48106

**Marsh:Water Column Interactions in Two Louisiana Estuaries.
Flux Measurements and Conceptual Implications**

A Dissertation

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

in

The Department of Marine Sciences

by

**Daniel Louis Childers
B.A., University of Virginia, 1983
M.S., University of South Carolina, 1985
August, 1989**

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It has been said that "nobody is given a wish without also being given the power to make it come true...sometimes, however, you have to work for it" (R.Bach, Illusions, 1976). Well, work I did, but many people played instrumental roles in giving me that power to make my wish--my dissertation--come true. I am very grateful to Dr. John W.Day, Jr., my major advisor, for being there these past 4 years. He generously provided me with the means to pursue the projects of my choice, and he provided guidance and assurance when I needed it while providing independence and freedom when I wanted it. I also wish to thank Drs. James Gosselink, Robert Muller, William Patrick, and Robert Twilley, my Dissertation Advisory Committee members, for their assistance with my work and perseverance through preparation of this dissertation. The financial support which made my degree work at Louisiana State University possible was provided by the Louisiana State University Alumni Federation, and Louisiana Land and Exploration granted me the leases necessary to conduct my field work.

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ABSTRACT

Many important ecosystem-level processes are integrated in hydrologically-forced marsh:water column interactions in estuaries. In this work, I quantified nutrient and sediment fluxes in two Louisiana estuaries using throughflow marsh flumes. The Barataria Basin estuary is in a later, deteriorating stage of the deltaic cycle. Brackish and saline marshes here exported dissolved inorganic nitrogen (DIN), total nitrogen, and dissolved organics (DOM); phosphorus fluxes were low and variable. Fourleague Bay marshes, in an earlier developmental stage of the deltaic cycle, imported DIN in the spring and released DIN in summer and fall. In this way, marshes here buffer open-bay N concentrations from high riverine N inputs. Fourleague Bay marshes exported phosphorus, but took up DOM on all tides sampled. Suspended sediment fluxes measured over individual tidal cycles were low and variable in both estuaries. Sediment accretion rates calculated from these fluxes were lower than actual measured accumulations. Apparently, most TSS flux onto marshes occurred during episodic climatological events.

Wetland loss, a pervasive problem in Louisiana, has numerous implications to marsh:water column interactions. An extended period of low water levels in 1987-88 isolated marshes from the water column; a phenomenon I refer to as "ephemeral wetland loss". Multivariate analyses indicated that 37% (intermonthly) and 46% (interannual) of the variability in historical (1963-87) coastal water levels was explained by climatological parameters. Further, mean annual water levels explained 26% of the interannual variability in inshore shrimp harvest (a measure of marsh-dependent estuarine productivity). The relationship was nonlinear, with low shrimp harvest at low and high water levels. In both cases, low catch corresponded to El Niño events.

Wetland loss represents a permanent removal of marsh area from the estuarine ecosystem. In Louisiana, most of this loss is a habitat change to open water. As marsh is converted to open water, the marsh:open water ratio changes and remaining marsh is "diluted" by increasing open water area. The functional loss of marsh, related to dilution,

is greater than actual areal loss of marsh. The magnitude of functional loss depends on estuarine morphology, and is highest in marsh-dominated estuaries with high initial marsh:open water ratios.

INTRODUCTION

Estuarine ecosystems are comprised of several closely interacting habitats, or subecosystems (referred to as microecosystems by Kitchens et al., 1979), including the vegetated marsh, water column, and subtidal benthos (Dame et al., 1977; Summers and McKellar, 1980; Childers and McKellar, 1987). An hierarchical approach to this conceptual scheme greatly facilitates ecosystem-level analysis (O'Neill et al., 1977). In such an approach the lowest hierarchical level is, for example, the individual processes unique in rate and function to each subecosystem, but common to all. These include individual steps of the nitrogen and phosphorus cycles (eg: N-fixation, denitrification, P-mobilization), primary productivity, decomposition, and trophic [material and energy] transfer (Figure 1). The interactions between these habitats, within the defined ecosystem boundaries of an estuary, form the second and more encompassing hierarchical level. At this level, the water column integrates exchanges of nutrients and materials with both the benthic and marsh subecosystems (Figure 1). The water column also receives inputs from rivers and adjacent uplands, and interacts with the coastal ocean. These extraecosystem, or landscape-scale interactions cross ecosystem boundaries and represent the apex of this hierarchical pyramid (Figure 1). In this work, I will specifically consider ecosystem-scale interactions between intertidal marshes and the estuarine water column.

Flat, expansive intertidal marshes are probably the most conspicuous morphological feature of Mississippi River Deltaic Plain (MRDP) estuaries. By recent estimates, these fresh, brackish, and saline marshes make up 25 to 40% of all coastal wetlands in the contiguous U.S. (Gagliano et al., 1981; Turner, 1982; Templet and Meyer-Arendt, 1989). In the context of the estuarine ecosystem, these marshes interact closely with the water column and, less directly, with other subecosystems (such as the subtidal benthos). Exchanges of nutrients and materials between marshes and the associated water column have been a focal point of estuarine ecosystems research for many years. Many studies quantified these exchanges by measuring nutrient fluxes at the mouths of narrow, shallow

channels connecting small marsh embayments with larger basins. Because this method does not uncouple, or separate, the marsh and subtidal benthic subecosystems, recent exchange work has been conducted with marsh flumes, allowing direct measurement of marsh:water column fluxes. In this study, I quantified marsh:water column interactions in the brackish and saline marshes of the Barataria Basin estuary and in brackish marshes of Fourleague Bay, Louisiana, with throughflow marsh flumes (Chapter 1).

The expansive marshes of the MRDP lack significant topography and are interlaced with numerous creeks and ponds. As a result, there is no hydrologic certainty that water flowing from the bayou on the flooding tide will return through the flume on the ebbing tide. To account for potentially independent flood and ebb tide water masses, I developed a throughflow flume technique--modified from past flume methodologies. In these throughflow flumes, concentrations measured simultaneously at the two ends of the flumes were compared, with "upstream" samples taken before exposure to the marsh within the flume and "downstream" samples after "treatment" by the marsh.

Measured marsh exchanges between Barataria Basin and Fourleague Bay marshes and the inundating water column seem to reflect the deltaic successional stage of each estuary (Chapter 2). The Barataria Basin estuary is in a later, deteriorating stage of the deltaic cycle, and is characterized by low freshwater inputs and subsiding, deteriorating marshes. Brackish and saline marshes here exported dissolved inorganic nitrogen (DIN, as ammonia= NH_4^+ and nitrate-nitrite= NN), total nitrogen (TKN), and dissolved organic (DOC) and nitrogen (DON); phosphorus (P) fluxes were low and variable. This export of DIN and DOM may represent a release of N and OM which accumulated during earlier stages of the deltaic succession. Fourleague Bay, on the other hand, is in an early, developmental stage of deltaic cycle, and the brackish marsh site here showed a seasonal pattern of DIN flux, with uptake in the spring [when riverine DIN inputs to Fourleague Bay are high] to export in the fall [when riverine N inputs are low]. These marshes appear to buffer the bay from high N levels during high river flow and low N levels during low

river flow. During all samplings, these marshes released soluble reactive (SRP) and total phosphorus (TP) to Fourleague Bay, where [open bay] subtidal SRP regeneration, mobilization, and release rates are low. These P releases by the marsh are probably important seasonal regulators of P levels in a Fourleague Bay water column characterized by potential P-limitation to aquatic primary productivity over much of the year.

Particulates flux patterns also differed between Barataria Basin and Fourleague Bay (Chapter 3). The Barataria Basin estuary is isolated from significant riverine sediment inputs, as demonstrated by low-magnitude TSS fluxes at the saltmarsh site (in fact, 2 of 4 samplings showed TSS export). Particulate organics (POM, as POC and PON) were taken up at both brackish and saline Barataria Basin marsh sites. In Fourleague Bay, the supply of large volumes of sediments to the water column during the spring freshet of the Atchafalaya River was reflected in TSS uptake at the brackish marsh site. The Fourleague Bay marsh exported POM during all samplings, though.

The TSS flux data measured in the marsh flumes were used to calculate sediment accretion rates at each site, given site-specific sedimentological characteristics and flooding regimes (Chapter 3). Actual sediment deposition was also measured with marker horizons at the Barataria Basin saltmarsh and Fourleague Bay brackish marsh sites. Sediment accumulation rates calculated from TSS fluxes measured over individual tidal cycles were markedly lower than measured accretion, for several reasons: 1) most sedimentation on MRDP marshes appears to be episodic and event-related (for example, as much as 1.5 cm of sediment was deposited at the Barataria Basin saltmarsh site during Hurricane Gilbert, September, 1988); 2) much of the vertical growth of MRDP marshes may be associated with accumulation of autochthonous organic matter not measured by flume flux techniques.

Between summer, 1987 and winter, 1988, MRDP coastal water levels and marsh inundation regimes were anomalously low for an extended period of time. Water level records from the BB saltmarsh site indicated that marsh flooding was reduced 32-92%,

compared to a 5-year mean; the BB brackish marsh was flooded even less frequently during this time (Chapter 4). This phenomenon, which I referred to as an "ephemeral wetland loss" because of its inhibitory effect on marsh:water column exchanges (Chapter 4), made the flume sampling difficult and resulted in a low number of successful sampling trips. Faced with a detrimental situation, I searched for new, somewhat conceptual approaches to quantifying marsh:water column interactions. As the late Albert Szent-Gyorgyi (Nobel Laureate and discoverer of Vitamin C) once noted: "Research is to see what everybody else has seen, and to think what nobody else has thought." To that end, I sought to determine what caused this low water event, and use its peculiarity to my advantage.

To explain this extended period of low coastal water levels, I examined possible climatological causes (Chapter 4). Coincidentally, a strong and unusual El Niño-Southern Oscillation (ENSO) event, known as a "La Niña" event, and drought conditions in much of North America also characterized 1987-88. I analyzed 28 years of historical water level, climatological, and biological data to test my hypothesis that these low water level events have occurred before, have physically inhibited marsh:water column interactions, and have coincided with strong ENSO events. Inshore shrimp harvest was chosen to represent commercially-important estuarine productivity that is dependent on marsh flooding. Multivariate analysis showed that wind direction, local precipitation and temperature, and Mississippi River discharge together explained 37 and 46% of the intermonthly and interannual variability, respectively, observed in historical coastal water levels. Interestingly, strong ENSO events in 1972-73 and 1982-83 coincided with high water levels, high precipitation, and high Mississippi River discharge, while the the lowest observed water levels and drought conditions were observed during the 1987-88 La Niña event. Historical water levels explained a significant 26% of the variability in inshore shrimp harvests during the same time period, but the relationship was not linear. Shrimp catches were low when coastal water levels were both low and high. Many of these low-

catch endpoints coincided with ENSO events and higher than normal estuarine marsh flooding (primarily through large freshwater inputs to the coastal zone) or with the 1987-88 La Niña event and lower than normal flooding.

The type of wetland loss I describe above as "ephemeral" is a temporary, climatologically-forced phenomenon. Louisiana is experiencing very high rates of permanent marsh loss as well, with a permanent loss of their ecological function from the estuarine ecosystem. The magnitude of this functional loss, manifested by increased open water area which further dilutes the remaining marsh, is strongly dependent on the marsh:water column ratio of the estuary (Chapter 5). Wetland-dominated estuaries (such as those of the MRDP) have large marsh areas and high marsh:open water ratios. Even a small areal conversion of marsh to open water in these estuaries greatly alters the marsh:open water ratio. For example, in an estuary with a marsh:open water ratio of 10:3, a 20% loss of marsh (by conversion to open water) actually decreases marsh release or uptake of critical nutrients and organic matter to/from the water column by over 50% as the remaining area of marsh is diluted by an increased area of open water (Chapter 5). In any examination of the importance of intertidal marshes to the estuarine ecosystem, it is important to quantify routine exchange with the inundating water column (particularly with respect to dissolved nutrients) and be able to relate these tidal fluxes to longer term processes (such as net sediment accretion). The various environmental factors which affect marsh:water column interactions will be manifested primarily by changes in inundation regimes—the hydrologic mechanism of that interaction. It is also important to understand these exogenous forces on the ecosystem, whether temporary or permanent, and their effects on short and long-term trends in marsh:water column interactions.

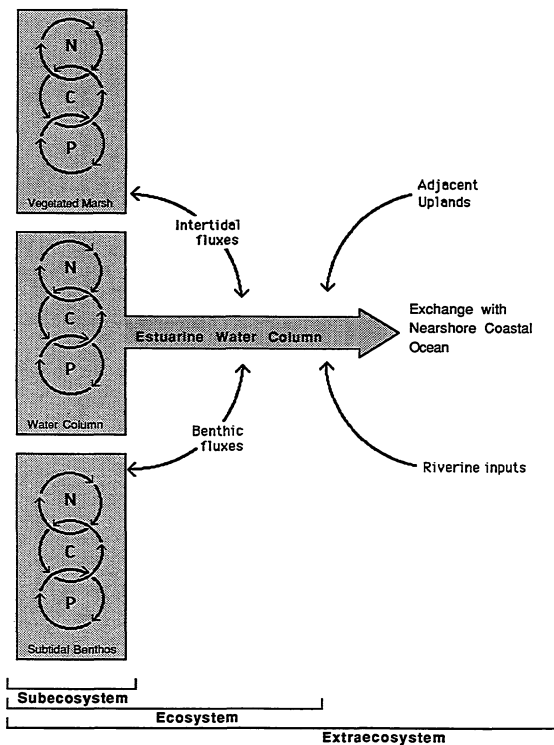


Figure I-1: Conceptual diagram of the hierarchical organization of an estuarine ecosystem. Note the central position of the water column as the integrator of interactions through all three levels.

Chapter 1

A Flow-through Flume Technique for Quantifying Nutrient and Materials Fluxes in Microtidal Estuaries

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A Flow-through Flume Technique for Quantifying Nutrient and Materials Fluxes in Microtidal Estuaries

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The marsh flume methodology has been modified for use in northern Gulf Coast estuaries, where tidal ranges are small and irregular and where wetlands are flat and expansive. In this technique, two key changes have been made: (i) flumes are open to water exchange at both ends, and; (ii) samples are taken simultaneously at both ends throughout a tidal cycle. Thus, the flumes are conceptually through-flow systems. Nutrient fluxes are calculated volumetrically, from the microtopography of the marsh and changes in water height over a tidal cycle. Instantaneous fluxes across each end of the flume are pooled for flooding or ebbing flow, then subtracted to give total net flux. Data from three flumes built in fresh, brackish, and saline marshes of the Barataria Basin estuary, LA, show that significant concentration differences (and hence significant fluxes) are detectable using our modified flume technique. Net areal flux values measured using the modified technique are in close agreement with values reported from past studies using flumes where concentrations are sampled only at the mouth end. The similarities indicate that our modifications to the flume methodology allowed us to quantify adequately the nutrient and material fluxes between Gulf Coast marshes and their inundating water column. This technique has applications in any estuary where marshes are microtidal, expansive, or irregularly flooded.

Introduction

One of the most exciting issues in estuarine ecology over the past three decades has been the 'outwelling' paradigm—do estuaries in general, and saltmarshes in particular, in fact export energy to coastal ecosystems? A number of studies and reviews have addressed this question (Odum & de la Cruz, 1967; Reimold, 1972; Kjerfve & McKellar, 1980; Nixon, 1980; Dame *et al.*, 1986; Whiting *et al.*, in press; and many others). This issue of estuarine outwelling has spurred investigations into a tangential question of the importance of intertidal marshes to estuarine ecosystem dynamics. A number of perspectives may be taken when studying exchange processes in marshes. In one, the marsh itself is the point of reference and specific processes are quantified. Another considers the intertidal marsh relative to the inundating estuarine waters and exchanges of various constituents between the marsh and the estuarine water column are measured directly. A number of studies quantifying fluxes between marsh subecosystems and adjacent water bodies have been

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conducted, but most of these have measured fluxes at the mouths of narrow, shallow channels connecting small marsh embayments to larger basins (Axelrad *et al.*, 1976; Heinle & Flemer, 1976; Woodwell *et al.*, 1977, 1979; Armstrong & Hinson, 1978; Valiela *et al.*, 1978; Odum *et al.*, 1979; Welsh, 1980; Daly & Mathieson, 1981; Chrzanowski *et al.*, 1982a,b, 1983; Simpson *et al.*, 1983; Dankers *et al.*, 1984; Stern *et al.*, 1986; Whiting *et al.*, 1987). This methodology actually measures processes occurring in both the wetland and the channel, and makes the strict separation of intertidal and subtidal subecosystems difficult. Other investigators have chosen to analyse marsh/water-column interactions directly by measuring nutrient fluxes through flumes built on the marsh surface (Lee, 1979; Wolaver *et al.*, 1980, 1985; Wolaver & Ziemann, 1983a,b; Chalmers *et al.*, 1985; Bowden, 1986; McIvor & Odum, 1986; Whiting *et al.*, in press) or in small tidal creeks (Ruble *et al.*, 1983a,b; Jordan *et al.*, 1983).

In all systems in which the flume technique has been used, the marshes are regularly and predictably flooded, the distances between the tidal creek and topographic boundaries are relatively small, and elevational gradients are distinct. In Louisiana Gulf coast wetlands, however, flooding occurs on irregular diel or seasonal time scales, primarily because of the effects of climatological forcing. As one proceeds inland from the coast, the frequency of marsh flooding decreases and the duration increases (Baumann *et al.*, 1984), largely due to the increasing influence of winds as tidal forcing decreases (Conner & Day, 1987). Furthermore, Louisiana marshes lack significant topography and are interlaced with numerous tidal creeks and ponds; there is no hydrologic certainty that water flowing from a tidal creek on the flood tide will return on the ebb tide. These characteristics required significant modifications to the flume techniques used in the studies cited above. We describe our modifications in this paper.

We are using this modified flume technique to quantify exchanges of carbon and nitrogen between the vegetated marshes and the water column along the estuarine salinity gradient of the Barataria Basin estuary, LA [see Conner & Day (1987) for a detailed site description]. This is the first time a flume method has been used in a northern Gulf Coast estuary, and the first time it has been used along the full salinity gradient of any estuary. Preliminary sampling at three salinity stations confirms the efficacy of the modified method for directly measuring nutrient and materials fluxes between microtidal marshes and their inundating water column.

Methods

Flumes were constructed in the fresh, brackish, and saline zones of the Barataria Basin estuary. Each flume consists of clear, corrugated fiberglass panels (each 0.66 m high and 2.70 m long) which form vertical walls 2 m apart. Boardwalks on either side prevent disturbance of the marsh surface (see Figure 1). The flumes are approximately 30 m long, with the exact length determined by site-specific conditions; they serve to prevent lateral water movement as the flooding tide inundates the marsh, without altering normal flow. The panels are removed after each sampling to prevent such long-term panel effects as shading, edge scouring, and wrack accumulation. Water levels at the brackish and saline marsh flumes are monitored continuously with Richards-type water-level recorders (Weathertronics Inc., Model 6510) installed at each site. The freshwater flume is built within the boundaries of the Jean Lafitte National Historical Park, where park personnel monitor water levels daily.

In virtually all flux studies utilizing marsh flumes, investigators constructed the flume to or beyond the high marsh. This permitted the assumption that all water which entered

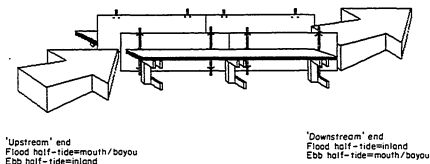


Figure 1. Conceptual representation of the throughflow flume design and schematic diagram of the calculations used to determine instantaneous fluxes of constituents into and out of the flumes. On the flood half-tide, the open-water body is on the left in the conceptual diagram, and water flows (onto the marsh) to the right. On the ebb half-tide, flow direction reverses and the open-water end is 'downstream'—on the right in the diagram. $[X]_t$, nutrient concentration at time t ; mouth and inland refer to the sampling locations at either end of the flume.

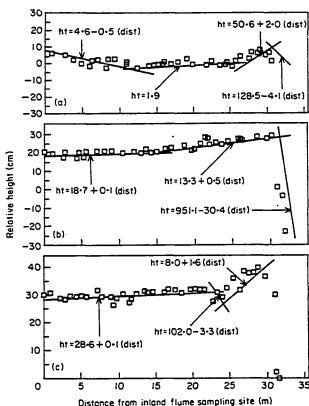


Figure 2. Microtopographic profiles of each marsh flume, including the equations used to generate geometrically the water volume based on water height (see explanation in text): (a) Profile from freshwater flume, Lake Salvador, LA. (b) Profile from brackish marsh flume, Long Bay, LA. (c) Profile from saltmarsh flume, Airplane Lake, LA.

the flume on the flooding tide left on the ebbing tide. These investigators also measured fluxes over complete exposed-inundated-exposed tidal cycles. Both these conditions generate problems when using the flume method in coastal Louisiana. First, the marshes are very flat and broad, and are frequently intersected by open water bodies. Thus, a classical low-marsh high-marsh transect is seldom seen, and it is unrealistic to try to build a flume long enough such that the 'flooding water in = ebbing water out' assumption will hold. Second, astronomical tidal amplitudes in the Barataria estuary range from 0 to less than 0.5 m. Meteorological effects on the flooding regime are significant, particularly further from the coast. At the salt marsh station, it is generally possible to sample a complete exposed-inundated-exposed cycle, but at the other stations it often is not. Certainly, the flooding regime is difficult to predict. For these reasons, we have made two key modifications to the flume design: (i) the flumes are open to water exchange at both ends (but with one end immediately adjacent to an open water body), and throughout the sampling period the water height and direction of throughflow are monitored, and (ii) water samples for nutrient analysis are taken simultaneously at both ends of the flume, as long as throughflow or water height changes are detectable (i.e. as long as there is a measurable flux of water through the flume). Theoretically, water flux may occur by throughflow only, water-level change only, or a combination of the two. We have never observed water flowing through a flume without a water-level change and horizontal velocities through our flumes are extremely slow. Thus, all water fluxes are based on volumetric changes determined from water-level data.

Past flume studies have determined the effect of the marsh on the overlying water by comparing nutrient fluxes into the flume at flood tide to those out of the flume at ebb tide. Our method is different in that we compare nutrient fluxes into the flume at one end to those out of the flume at the other end, noting the direction of throughflow (toward the marsh or toward the creek; Figure 1). This is the most important difference between our method and all other flume techniques used before. Every 30–60 min, duplicate water samples are drawn from both ends of the flume. Water is pumped from within the flume at mid-depth in the water column, using a handpump at one end of the flume and an auto-sampler at the other (ISCO). From each sample, replicated subsamples for NH_4^+ , $\text{NO}_3^- + \text{NO}_2^-$, and DOC are immediately filtered through pre-ashed, pre-weighed 0.45 μm glass-fiber filters. Samples for TN are also taken; subsamples and filter papers are frozen in the field. Water levels within the flume are monitored continuously. At each sampling, above-ground macrophyte biomass samples (0.25 m^2) are taken at randomly chosen sites near the creekside, central, and inland areas of the flume.

In the laboratory, Technicon Autoanalyzer techniques are used to determine NH_4^+ (Method 154-71W/B), $\text{NO}_3^- + \text{NO}_2^-$ (Method 158-71W), and DOC (Method 451-76W) concentrations and Keldahl digestions of unfiltered samples generate TN values (Method 329-74W/B; U.S. E.P.A., 1979). Filter papers are dried and weighed for suspended sediment data, then acidified and ashed using a Perkin-Elmer Model 240-B elemental analyzer to determine POC and PON concentrations. DON is derived by subtraction. Biomass samples are separated into live and dead components, dried at 60°C, and weighed.

The volume of water within the flume is calculated hypsometrically using the marsh microtopography within the flume; this volumetric model assumes a horizontal flux of water through the flume (Wolaver *et al.*, 1985). Water level changes, coupled with measurements of water flow through the flume when detectable, thus represent water-volume changes. For each sampling, we generate a polynomial regression of the

water-volume change pattern over time sampled (r^2 is typically 0.95–0.99). Instantaneous water flux is then calculated as the derivative of this volume versus time curve (dV/dT) at each sampling time. Positive water flux occurs during the flooding tide (equivalent to uptake of water by the marsh) while negative water flux occurs when the tide ebbs. Instantaneous nutrient fluxes across each end of the flume (in mg X s^{-1}) are calculated as products of sample concentration (mg X m^{-3}) and the instantaneous water flux ($\text{m}^3 \text{s}^{-1}$):

$$\text{instantaneous flux} = [X]_i (\text{instantaneous flux } \text{H}_2\text{O})_i$$

For each half-tide (flood and ebb), we calculate a total mass of each constituent passing across each end of the flume (= total flux, in mg X). This total flux is derived by averaging instantaneous flux from consecutive samplings, multiplying this average by the time interval between the samples, and summing these values for each half-tide:

$$\text{total flux} = \sum_{i=1}^n \left[\frac{(\text{inst. flux})_i + (\text{inst. flux})_{i+1}}{2} (\text{time interval})_{i,i+1} \right]$$

where n is the number of samples in the flood or ebb half-tide.

Net flux (in mg X s^{-1}) for each half-tide is calculated as: $(\text{total flux}_{\text{mouth}}) - (\text{total flux}_{\text{inland}}) / \text{total time that tide was sampled}$:

$$\text{net flux} = \frac{(\text{total flux})_{\text{mouth}} - (\text{total flux})_{\text{inland}}}{\text{total time}_{(i=1-n)}}$$

Net areal flux (in $\text{mg X m}^{-2} \text{h}^{-1}$) is simply the net flux divided by the time-weighted area of marsh inundated during that tide:

$$\text{net areal flux} = \left[\frac{(\text{total flux})_{\text{mouth}} - (\text{total flux})_{\text{inland}}}{\left(\sum_{i=1}^n \frac{[\text{mean flume length}]_i [\text{time interval}]_i}{\text{total time}_{(i=1-n)}} \right) \frac{\text{total time}}{3600}} \right]$$

Subtracting total flux across the inland end from that across the mouth end generates positive flux values for uptake by the marsh and negative values for marsh export, regardless of the direction of water movement.

We use a simple randomized block design to test for flux significance. Nutrient and materials concentrations are pooled into upstream and downstream values and compared using a paired t -test. Upstream concentrations are values measured at the mouth end of the flume on the flood half-tide and the inland end of the ebb half-tide; they are (essentially) 'before treatment' samples. Downstream concentrations, as sample values from the other end of the flume, are 'after treatment' measurements. The water flowing through the flume on the two half-tides is likely to be different, even independent, and fluxes may reflect this. To allow the determination of these possible differences, we separate our concentration data into flood half-tide and ebb half-tide subsets and compare upstream and downstream values for each separately, again using paired t -tests. This additional statistical treatment will make it possible to determine whether any flux is weighted to activity early or late in a tidal cycle, or if time or direction of water flow have any effect on overall sampled flux.

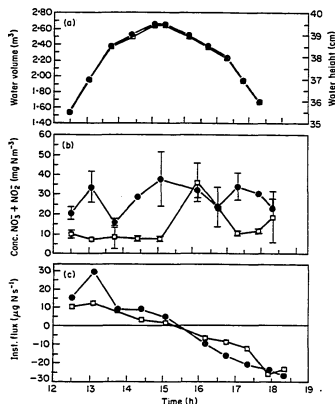


Figure 3. Dissolved inorganic nutrients as $\text{NO}_3 + \text{NO}_2$, saltmarsh flume, 2 June 1987: (a) water height (●) and volume within the flume (□); (b) concentration at the mouth (□) and inland (●); (c) instantaneous flux at the mouth (□) and inland (●).

The throughflow method we use here is (essentially) a modified upstream-downstream system. In a throughflow system, concentration change between the upstream and downstream sampling stations is a function both of processes within the system and of the initial upstream concentration. Thus, temporal variation in concentrations at the upstream sampling site will be manifested in the patterns seen at the downstream site, after a time lag (Odum, 1956; Rogers, 1979). Strictly speaking, this should not generate problems with our calculations when water height is constantly changing; a water mass having passed through the flume will be volumetrically different from when it entered the flume. Nonetheless, we circumvent these possible problems by pooling the instantaneous fluxes measured during each half-tide (flood and ebb) at the creekside (mouth) and the inland ends of the flume separately, and generating a total mass passing across each end during that tide.

Results and discussion

Preliminary field project

To demonstrate the feasibility of using marsh flumes in a microtidal Gulf Coast system, we built three flumes in the Barataria Basin estuary; one each at a freshmarsh site (FM), a

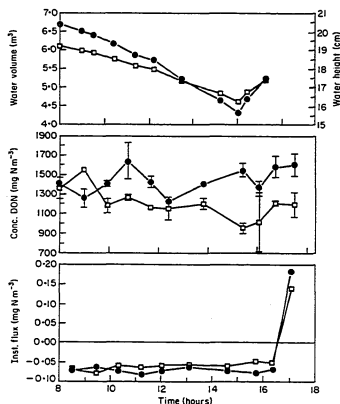


Figure 4. Dissolved organic nutrients as DON, freshwater flume, 5 October 1986: (a) water height (●) and volume (□) within the flume; (b) concentration at the mouth (□) and inland (●); (c) instantaneous flux at the mouth (□) and inland (●).

brackish-marsh site (BM), and a saltmarsh site (SM). The freshwater flume was sampled 5 October, 1986, the brackish marsh flume 23 June, 1987, and the saltmarsh flume on 2 June, 1987. Figure 2 shows the microtopographic data and the linear approximations used to calculate water volume. The temporal water-height patterns observed during the tidal cycles sampled are plotted in Figures 3a–5a. Note that the freshwater tidal pattern for 5 October, is ebb dominated. Asymmetrical tidal patterns are common in these marshes.

Figures 3–5 show examples of the temporal patterns of concentration (b) and flux (c) for each nutrient type and each site: $\text{NO}_3^- + \text{NO}_2^-$ at the SM site as dissolved inorganic [Figure 3(b,c)]; DON at the FM site as dissolved organic [Figure 4(b,c)]; and POC at the BM site as particulate organic [Figure 5(b,c)]. In most cases, there are distinct and significant differences between samples taken simultaneously at the two ends of the flumes. Often, as the tide turns, these differences diminish and concentrations at both ends of the flume increase (for dissolved constituents) or decrease (for particulate constituents). This observation may be related to one or more of the factors discussed above, but a more complete data set is needed before this can be clarified and tested.

Information on the concentrations, fluxes, and tests of significance for dissolved inorganic, dissolved organic, and particulate nutrients is summarized in Table 1. Data

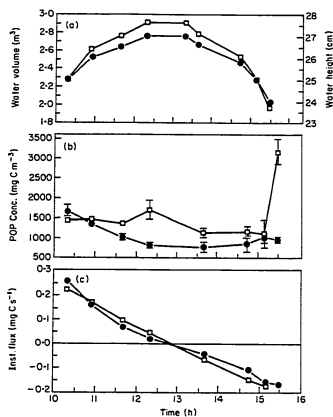


Figure 5. Particulate nutrients as POC, brackish marsh, 23 June 1987: (a) water height (\bullet) and volume (\square) within the flume (m^3); (b) concentration at the mouth (\square) and inland (\bullet); (c) instantaneous flux at the mouth (\square) and inland (\bullet).

from the FM flume sampling indicate insignificant fluxes of NH_4^+ , $\text{NO}_3^- + \text{NO}_2^-$, and DOC, yet total nitrogen (TN) shows a significant export during the flooding tide and a total export of $360 \mu\text{g N m}^{-2} \text{h}^{-1}$. In late June 1987, the brackish marsh exported a significant mass of oxidized inorganic nitrogen ($100 \mu\text{g N m}^{-2} \text{h}^{-1}$ as $\text{NO}_3^- + \text{NO}_2^-$) while NH_4^+ exchange was insignificant. Significant exports were also seen in DOC ($224 \text{ mg C m}^{-2} \text{h}^{-1}$), DON ($2.8 \text{ mg N m}^{-2} \text{h}^{-1}$), and TN ($9.2 \text{ mg N m}^{-2} \text{h}^{-1}$). Our modified flume technique detected significant dissolved nutrient exports from the saltmarsh in June 1987, particularly $\text{NO}_3^- + \text{NO}_2^-$ ($190 \mu\text{g N m}^{-2} \text{h}^{-1}$), DOC ($9.2 \text{ mg C m}^{-2} \text{h}^{-1}$), and total nitrogen ($5.2 \text{ mg N m}^{-2} \text{h}^{-1}$). POC, PON, and TSS fluxes at the SM flume switched direction in the half-tide fluxes measured over the same tidal cycle. POC release was significant during the ebbing tide only, but the magnitude of this export was sufficient to make the overall export significant as well. PON export occurred on the flooding tide but the flux reversed direction, to marsh uptake, when the tide switched. TSS exchange followed the opposite pattern (Table 1); neither the PON nor the TSS whole-tide flux was significant.

TABLE 1. Summary of preliminary results from marsh flume flux study

Constituent	Conc. range (mg m ⁻³)	Whole-tide flux		Half-tide flux significance ^a	
		(mg m ⁻² h ⁻¹)	Sig. ^a	Flood	Ebb
<i>Freshwater flume (Lake Salvador, LA) 5 October 1986</i>					
NH ₄ ⁺	11.2-134.4	0.52	NS	NS	NS
NO ₃ ⁻ + NO ₂ ⁻	1.4-23.8	-0.05	NS	NS	NS
DOC	10 800-29 100	-44.98	NS	NS	NS
DON	808-1679	-2.28	NS	E**	I**
POC	1387-4383	8.94	I**	NS	I**
PON	153-740	1.22	I**	NS	I**
TSS	1700-40 000	-169.8	NS	NS	NS
TN	1022-2128	0.36	E*	E**	NS
<i>Brackish marsh flume, 23 June 1987</i>					
NH ₄ ⁺	19.6-107.8	0.08	NS	NS	NS
NO ₃ ⁻ + NO ₂ ⁻	2.8-36.4	-0.10	E**	E**	E*
DOC	6800-48 700	-223.8	E**	E*	E*
DON	91.0-626.4	-2.77	E**	E*	E*
POC	685-1875	-0.45	NS	I*	E*
PON	78.3-195.0	0.35	I**	I*	NS
TSS	1100-13 300	33.87	NS	I*	E*
TN	336-826	-2.50	E**	E*	E**
<i>Saltmarsh flume (Airplane Lake, LA), 2 June 1987</i>					
NH ₄ ⁺	29.4-204.4	-0.32	NS	E**	I**
NO ₃ ⁻ + NO ₂ ⁻	4.2-47.6	-0.19	E**	E**	E**
DOC	2500-12 600	-9.24	NS	E**	I**
DON	370-1717	-3.42	NS	E**	I**
POC	496-1847.9	-3.93	E*	NS	E*
PON	41.7-266.7	-0.99	NS	E**	I*
TSS	19 000-125 000	-78.06	NS	I*	E**
TN	602-2058	-5.24	NS	E**	I**

^aSignificant results ($\alpha=0.05$) are marked with one asterisk (*); highly significant results ($\alpha=0.01$) are marked with two asterisks (**); NS, not significant. I, +, Import to, or uptake by the marsh; E, -, export from, or release by the marsh.

In a number of cases, the data from Table 1 show a significant exchange during the flooding tide, then a significant exchange in the other direction during the ebbing tide (DON flood export and ebb import at the FM site; POC, PON, and TSS flood import and ebb export at the BM site on 15 September; nearly all constituents from the June saltmarsh sampling). This may be related to the fact that the creekside/mouth sampler was placed directly at the marsh-bayou interface rather than 1-2 m inland, causing ebbing-tide mouth samples to represent bayou water rather than water flowing through the flume toward the bayou. In more recent samplings, the creekside sampler has been placed over the marsh proper, within 1-2 m of the creekbank. This phenomenon may also be an artifact of differences in the duration which different parcels of water are exposed to marsh and (in a related sense) differences in the total area of marsh to which a water parcel has been exposed. The half-tide flood and ebb fluxes measured with our throughflow flume technique are somewhat independent, as they are based on samples from different water parcels. With more data, we will be able to test hypotheses about residence time of, and marsh area affected by, these water parcels by comparing half-tide fluxes to the whole-tide flux data.

TABLE 2. A comparison of net areal flux values generated by this preliminary study, from the Barataria Basin estuary, Louisiana, and values reported in three past studies: where only month and year are shown, the fluxes are means of all reported values for that month; all fluxes shown are in units of $\text{mg N/m}^{-2}/\text{h}^{-1}$

Location	NH_4^+	$\text{NO}_3^- + \text{NO}_2^-$	DON	PON/PN*
<i>Freshmarsh</i>				
North River, MA, 10/7/82†	-3.7	-4.06		
North River, MA, 8/4/82†	-2.12	-1.13		Not presented
Lake Salvador, LA, 10/5/86	0.52	-0.05		
<i>Brackish marsh</i>				
Carter's Creek, VA, 6/78†	0.16	-0.02	1.40	8.72
Carter's Creek, VA, 6/79†	0.0	0.02		Not presented
Carter's Creek, VA, 9/78†	0.32	-0.41	4.23	-5.64
Long Bay, LA, 9/15/86	-3.25	-0.19	-9.86	2.81
Long Bay, LA, 6/23/87	0.08	-0.10	-2.74	0.35
<i>Saltmarsh</i>				
North Inlet, SC, 6/83†	-0.10	0.23	-4.67	-2.33
North Inlet, SC, 6/84†	-0.25	0.08	-2.00	-1.50
Airplane Lake, LA, 6/2/87	-0.32	-0.19	-3.42	-0.99

*Particulate nitrogen fractions presented as PN in the reported studies and as PON in our work.

†From Bowden (1986).

‡From Wolaver *et al.* (1983). Fluxes calculated as a mean of more than one value presented for that month, and based on an average 12-42-h tidal cycle.

§From Whiting (1985). Fluxes calculated as a mean of more than one value presented for that month, and based on an average 12-42-h tidal cycle.

-, Export from the marsh; +, import to the marsh.

Comparison of techniques

In every study utilizing the flume methodology to quantify directly the material exchanges on vegetated marshes, fluxes have been calculated given the 'flooding water in = ebbing water out' assumption. This allows investigators to determine import or export by sampling only at the mouth of the flume. Thus, export occurs when the measured flux of a constituent during ebb tide is greater than during flood tide. Our modified method is unique because our flumes are throughflow structures and we calculate fluxes across both ends of the flume for each half-tide. Thus, flood-tide export occurs when the flux of a constituent across the inland end of the flume is greater than across the mouth end. Ebb tide export is the opposite. The final product of both flume methods is an areal flux of nutrients and materials between marshes and the inundating estuarine water column. Though the methods are different, the resultant exchanges should be (at least) reasonably comparable. In Table 2 we compare net areal fluxes (in $\text{mg X m}^{-2} \text{ marsh h}^{-1}$ inundated) from our preliminary study to those reported for the same time of year in three past flume studies: Bowden (1986) for the freshmarsh, Wolaver *et al.* (1983a,b) for the brackish marsh, and Whiting (1985) for the saltmarsh. In most cases, our flux values are comparable to those measured in flume studies using the traditional method. The similarities are particularly striking in the saltmarsh data and the June brackish marsh data. [We note that the riverine freshwater flume used in Bowden's study (1986) was artificially flooded for most samplings.] Thus, while our modified flume technique is clearly different from past methods, Table 2 exemplifies the fact that our modifications

allowed us to apply the flume technology successfully to the microtidal, expansive marshes characterizing the Gulf Coast.

Summary

In this paper we report on modifications which make the flume methodology feasible for directly quantifying nutrient and material exchanges in microtidal wetlands. The most important of these changes treats the flume as a throughflow system, both in methods of sampling and in computation of fluxes. Preliminary data from three marsh flumes built along the salinity gradient of the Barataria Basin estuary, LA, demonstrate that this modified technique is an effective method of directly measuring how Gulf Coast wetlands interact with their associated estuarine water column. The methods comparison shows that our flume technique, designed for the expansive, microtidal wetlands of the Louisiana coast, is a unique and viable method for quantifying intertidal exchanges in these systems. We plan to continue using this technique to quantify, for the first time, temporal patterns in marsh-estuarine exchanges along the salinity gradient of a microtidal Gulf coast estuary.

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Chapter 2

Marsh:Water Column Interactions in two Louisiana Estuaries. I. Dissolved and Total Nutrients

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ABSTRACT

The exchange of dissolved nutrients between marshes and the inundating water column was measured using marsh flumes, in two microtidal Mississippi River Deltaic Plain (MRDP) estuaries: The Barataria Basin estuary and Fourleague Bay, Louisiana. The flumes are of a throughflow design, with samples taken simultaneously at either end during a tide and net fluxes determined by comparing instantaneous fluxes into and out of the flume. The flumes were sampled between September, 1986 and April, 1989, coincident with an extended period of low sealevel on the Louisiana coast, and thus marshes were flooded less frequently than normal. The magnitude and direction of dissolved nutrient exchanges in MRDP marshes seem to reflect the deltaic successional stage of that estuary. The Barataria Basin estuary is in the later, deteriorating stage of the deltaic cycle, characterized by low freshwater inputs and subsiding marshes. Both brackish and saline marshes supplied dissolved organic nitrogen (DON), inorganic nitrogen (ammonia+nitrate+nitrite=DIN), dissolved organic carbon (DOC), and total nitrogen (as total Kjeldahl nitrogen=TKN) to the water column. The export of DIN is probably related to the N accumulated in earlier stages of deltaic development and released as these marshes deteriorate. Coastal brackish marshes of Fourleague Bay, part of an accreting marsh system in an early, developmental stage of the deltaic cycle, seasonally exported NH_4^+ , NN, and TKN to the open water estuary. FLB marshes apparently acted as a short-term buffer of DIN by taking up N in spring, when baywide concentrations were high, and supplying DIN to the estuary in summer and fall, when concentrations in the bay were lower. Differences in phosphorus (P) and dissolved organic fluxes between these two estuaries were also observed. There were large exports of soluble reactive phosphorus (SRP) and total phosphorus (TP) at the Fourleague Bay site, while DOC was imported. In the Barataria Basin marshes, SRP fluxes were variable and low, while DOC was imported. The export of P from Fourleague Bay marshes may be related to the large

riverine supply of sediment-bound P deposited in the earlier [deltaic] successional history of this estuary, which may be remobilized in the reducing soils of the marshes.

Throughflow flumes are unique in that water is sampled upstream and downstream of a given area of marsh enclosed within the flume. Thus, nutrient flux at the marsh sediment:water interface can be determined on both flood and ebb tides. During flood tide, the upstream, or "source" end of the flume is adjacent to the bayou, while during ebb tide it is at the inland end of the flume. Most imports of dissolved nutrients were correlated with higher upstream [source] concentrations, and flux rates were fairly consistent throughout the tide. Nutrient exports did not correlate with upstream concentrations, though, and in many cases the flux was dominated by [early] flood tide nutrient release. This pulsed behavior may be caused by rapid diffusion from the sediments early in the tidal cycle, when the sediment:water concentration gradient is largest. NH_4^+ fluxes were not related to concentration, perhaps indicating an exchange mechanism dominated by [active] biological processes. The length of interaction (flooding) between marsh and inundating water had no apparent effect on flux as dissolved nutrient fluxes did not correlate with duration or maximum height of tides.

INTRODUCTION

A detailed understanding of marsh:water column interactions is critical to any study of estuarine ecosystem dynamics. In Carmarthen Bay, S. Wales, for example, Owens et al. (1986) observed the highest chlorophyll concentrations and rates of primary productivity, and the most rapid nutrient regeneration in the shallow periphery near marshes fringing the bay. It is difficult, however, to draw generalizations about marsh:water column exchanges because data from exchange studies are highly variable (Nixon, 1980). Work published in the 1960's and 1970's generally implicate coastal marshes as sources of organic carbon, but there is a wide range of flux values. Marshes also seem to transform nitrogen by importing oxidized inorganic species and exporting dissolved and particulate reduced forms. In addition, total phosphorus is taken up and later released as remobilized inorganic phosphate (Nixon, 1980). In most of these studies, fluxes were measured at the mouths of narrow, shallow channels connecting small marsh embayments to larger basins (Axelrad et al., 1976; Heinle and Flemer, 1976; Stevenson et al., 1977; Woodwell et al., 1977, 1979; Armstrong and Hinson, 1978; Valiela et al., 1978; Odum et al., 1979; Welsh, 1980; Daly and Mathieson, 1981; Chrzanowski et al., 1982a,b, 1983; Simpson et al., 1983; Dankers et al., 1984; Stern et al., 1986; Whiting et al., 1987). Unfortunately, accurate records of marsh flooding were rarely reported. This methodology measures processes occurring both on the marsh and in the channel, and makes the strict separation of intertidal and subtidal subecosystems difficult.

Since 1980, a number of studies have quantified marsh:water column interactions directly by measuring nutrient fluxes through flumes built on the marsh surface (Lee, 1979; Wolaver et al., 1980, 1983; Wolaver and Zieman, 1983; Chalmers et al., 1985; Wolaver et al., 1985; Bowden, 1986; Wolaver and Spurrier, 1988a,b; Whiting et al., 1989; see Table 1). Data from this technique show that most saltmarshes import virtually all dissolved nutrients, including ammonium (NH_4^+), nitrate+nitrite ($\text{NO}_3^- + \text{NO}_2^- = \text{NN}$), orthophosphate (PO_4^{3-}), and dissolved organic carbon (DOC). Total nitrogen (TN) and

total phosphorus (TP) are also taken up, but dissolved organic nitrogen fluxes are variable, with export from a South Carolina marsh (Whiting et al., 1989) and uptake by a Virginia marsh (Wolaver et al., 1983; Table 1). Bowden (1986) reported uptake of NH_4^+ and NN by a Massachusetts tidal freshwater marsh (Table 1). Significant relationships between the magnitude of fluxes measured using marsh flumes and the concentration of the overlying tidal waters have been reported for dissolved inorganic nitrogen (Wolaver et al., 1983; Whiting et al., 1989), while no such relationship has been observed for PO_4^{3-} (Wolaver and Spurrier, 1988a) or DOC (Chalmers et al., 1985; Wolaver and Spurrier, 1988b).

Direct measurements of marsh:water column nutrient fluxes have never been made in microtidal Gulf coast estuaries, and these fluxes have never been quantified in brackish marshes. In this study, we measured dissolved and total nutrient exchanges between a coastal saltmarsh, a coastal brackish marsh, and an inland brackish marsh in two Louisiana estuaries using throughflow marsh flumes (Chapter 1—Childers and Day, 1988).

MATERIALS AND METHODS

Two marsh flumes were built in the Barataria Basin estuary, LA in summer, 1986; one in a coastal saltmarsh (BB-SM), approximately 10 km from the Gulf of Mexico, and one in a brackish marsh (BB-BM), about 40 km from the coast (Figure 1a). This estuary is located on a deltaic land mass deposited over 2000 years ago, and has since been mostly isolated from riverine influence (Conner and Day, 1987). The Barataria Basin is in the "deterioration" stage of the deltaic cycle (Coleman and Gagliano, 1964; Kolb and Van Lopik, 1966), as indicated by high wetland loss rates (Gagliano et al., 1981; for a complete description of this estuary, see Conner and Day, 1987). A third flume (FLB-BM) was constructed in the coastal brackish marshes of lower Fourleague Bay, LA, also approximately 10 km from the Gulf of Mexico (Figure 1b). This estuary is adjacent to the active Atchafalaya River delta, where sediment supply is sufficient for marsh accretion (Baumann et al., 1984). Fourleague Bay is thus in the young, "depositional" stage of

deltaic succession (Coleman and Gagliano, 1964; Kolb and Van Lopik, 1966; see Madden et al., 1988, for a description of this estuary). We will refer to these flumes as BB-SM, BB-BM, and FLB-BM, respectively. Macrophyte plant communities at the BB-BM site, are dominated by *Spartina patens*, *Distichlis spicata*, and *Scirpus olneyi*. The BB-SM site is a pure stand of *Spartina alterniflora*, while the coastal brackish marsh site FLB-BM is primarily *Spartina patens* and *Distichlis spicata*, interspersed with *Juncus roemerianus* and *Spartina alterniflora*.

The flumes consisted of clear, corrugated fiberglass panels (each 0.66 m high and 2.7 m long) forming vertical walls 2 m apart. Boardwalks on either side prevented disturbance of the marsh surface. The 30 m long flumes prevented lateral water movement, without altering normal flow, as the flooding tide inundated the marsh. The panels were removed after each sampling to prevent long-term panel effects, such as shading, edge scouring, and wrack accumulation. Between May, 1987 and April, 1989, we continuously monitored water levels at the BB-SM and BB-BM sites with Richards-type water level recorders (Weathertronics Inc. Model 6510), and used this data to determine the frequency of marsh inundation at each site.

A throughflow design was used in these flumes. This modification was necessary because marshes of the Mississippi River Deltaic Plain (MRDP) lack significant topographic relief and are interlaced with numerous tidal creeks and ponds. There is no hydrologic certainty that water flowing from a tidal creek on the flooding tide will return on the ebbing tide. The throughflow technique is based on comparison of concentrations before and after passing over a specific marsh surface, regardless of the direction of tidal flow (Chapter 1--Childers and Day, 1988). Replicate water samples were taken simultaneously at each end of the flume every 30-45 minutes over a full tidal cycle. From each, replicated subsamples for NH_4^+ , NN , PO_4^{3-} (as soluble reactive P=SRP), and DOC were immediately filtered through pre-ashed, pre-weighed glass fiber filters (GF-F). Unfiltered samples for TN (as total Kjeldahl N=TKN) and TP were also taken, and all

subsamples were frozen in the field. At each sampling, aboveground macrophyte biomass samples (0.25 m²) were taken at randomly chosen sites near the creekside, central, and inland areas of the flume. Over each tidal cycle sampled, we continuously monitored water levels within the flume. Constituent fluxes (calculated as mg m⁻² marsh inundated hr⁻¹ of inundation) were based on constituent concentrations multiplied by the volumetrically-determined water flux. We determined net fluxes by comparing flux into the flume with that out of the flume, following the convention that negative fluxes are exports from the marsh surface to the inundating water column and positive fluxes are imports. Details of the flume design, sample treatments, flux computations, and data interpretation are presented in Chapter 1 (Childers and Day, 1988).

In the laboratory, Technicon Autoanalyzer techniques were used to determine concentrations of NH₄⁺ (Method 154-71W/B), NO₃⁻ + NO₂⁻ (NN, Method 158-71W), SRP (Method 155-71W), and DOC (Method 451-76W) concentrations and Kjeldahl digestions of unfiltered samples generated TKN and TP values (Method 329-74W/B; U.S. E.P.A., 1979). DON was derived by subtraction of NH₄⁺, NN, and particulate organic N (PON) concentrations from the corresponding TKN value (analysis of particulate nutrients is detailed in Chapter 3--Childers and Day, submitted). Biomass samples were separated into live and dead components, dried at 60°C, and weighed.

We used a simple randomized block design analysis of variance (RBD ANOVA) to test whether fluxes were significantly different from zero. Nutrient concentrations were pooled into upstream and downstream values and compared using a paired t-test. Upstream concentrations, measured at the bayou (mouth) end of a flume on the flood tide and the inland end on the ebb tide, are "before treatment" samples. Downstream concentrations, as sample values from the other end of the flume, are "after treatment" measurements. The water flowing through a flume on flood and ebb tides are likely to be independent, and fluxes may reflect this. To allow determination of these possible differences, we separated concentration data into flood tide and ebb tide subsets and

compared upstream and downstream values for each separately, using both paired t-tests (RBD) and 2-factor ANOVA. A 2-factor ANOVA (Statview 512+® Software Package) was used to determine whether any flux was weighted to activity early or late in a tidal cycle, or if time or direction of water flow have any effect on overall sampled flux. In this analysis, the independent variables (treatments) were tide, as flood or ebb, and flume location, as upstream or downstream and the dependent variable was constituent concentrations (pooled by flux direction into imports and exports). The 2-factor ANOVA interaction term determined whether the effect of tide and location in the flume on sample concentrations were independent.

We assessed the potential importance of tidal water concentration and hydrologic parameters to measured fluxes by correlating the following: flood and ebb tide fluxes (Y_1) against upstream (X_1) and downstream (X_2) mean concentrations and durations (X_3) for each tidal regime; net tidal fluxes (Y_2) against flood (X_4) and ebb (X_5) mean nutrient concentrations of flood and ebb tides, upstream (X_6) and downstream (X_7) mean net tidal concentrations, overall mean concentrations (X_8), slack high tide water height (X_9), and total (X_{10}), standing live (X_{11}), and standing dead (X_{12}) macrophyte biomass. Correlations were run on flood, ebb, and net tide flux results from the BB-SM and FLB-SM stations only, as there were only 2 sampling events at the BB-BM site. Since both the BB-SM and FLB-BM sites are in coastal marshes (each is about 10 km from the Gulf coast), we also pooled flux data from the two sites for testing these correlations.

RESULTS

The 2.5 years of this study corresponded with an extended period of anomalously low sealevel on the Louisiana coast which resulted in a dramatically reduced marsh inundation regime (frequency and duration) in MRDP estuaries between summer, 1987 and early 1989. Water level records from the BB-SM and BB-BM sites indicate that, during this time, the BB-SM was inundated 32-92% less frequently than expected (Baumann,

1987; Chapter 4—Childers et al., submitted). Inundation frequency was more reduced further inland, where the BB-BM was flooded less than 10% of the time between July, 1987 and August, 1988 and did not flood for 92 consecutive days between mid-December, 1987 and mid-March, 1988 (Chapter 4—Childers et al., submitted). This unusual phenomenon made sampling of marsh:water column interactions difficult. In fact, between September, 1986 and April, 1989, 10 trips were made to the BB-BM, 6 trips to the BB-SM, and 9 trips to the FLB-BM, yet we were able to sample only 2, 4, and 3 flooding tides, respectively. Only the November, 1987 and April, 1988 samplings at the BB-SM site were made after the onset of this low flooding event.

Flux Results

In both samplings at the BB-BM, the marsh exported NN for a net average release of $-0.29 \text{ mg m}^{-2} \text{ h}^{-1}$ (Figure 2b). DIN export was measured at the BB-SM site in all tides except in late summer, during the peak macrophyte biomass season, when there was a highly significant uptake of NH_4^+ and NN (Figure 2a, b). Net releases of NH_4^+ and NN averaged $-0.18 \text{ mg m}^{-2} \text{ h}^{-1}$ and $-0.22 \text{ mg m}^{-2} \text{ h}^{-1}$, respectively. At the FLB-BM site, DIN was imported in the spring ($0.14 \text{ mg m}^{-2} \text{ h}^{-1}$) and released in low quantities in mid-summer ($-0.46 \text{ mg m}^{-2} \text{ h}^{-1}$) and in greater amounts in early fall ($-0.73 \text{ mg m}^{-2} \text{ h}^{-1}$; Figure 2a, b).

There was a marked difference in dissolved inorganic phosphorus fluxes between the two estuaries. SRP flux at the BB-BM site was variable, with export observed in the fall and import, of the same magnitude, in the summer (Figure 2c). SRP was released by the BB-SM marsh in April, June, and November, but a relatively high uptake of $0.38 \text{ mg m}^{-2} \text{ h}^{-1}$ was measured in August (Figure 2c). Inorganic phosphorus was always released by the FLB-BM, though, with a net average flux two orders of magnitude greater than that measured at either Barataria Basin location ($-1.04 \text{ mg m}^{-2} \text{ h}^{-1}$; Figure 2c).

We measured releases of DOC and DON at the BB-BM site, variable flux patterns at the BB-SM site, and uptake of DOC by the FLB-BM marsh (Figure 3). Dissolved

organics were exported from the BB-BM marsh in both samplings. Barataria Basin saltmarshes appear to export dissolved organics at all times of the year except late summer, when a large uptake of DOC occurred (211 mg m⁻² h⁻¹; Figure 3a). During all three FLB-BM tides, the marsh took up DOC. These fluxes were up to an order of magnitude greater than dissolved organics exchanges in the Barataria Basin estuary (Figure 3a). The net uptake of DOC by the FLB-BM marsh, averaged over the three samplings, was 321.9 mg m⁻² h⁻¹. [Analytical problems precluded the use of DON concentration data in 2 of 3 samplings at this site].

Total N (TKN) was supplied to the inundating water column at all three sites except in the BB-SM during August and November tides (Figure 4a). The average export of TKN was -6.6, -5.7, and -25.9 mg m⁻² h⁻¹ at BB-BM, BB-SM and FLB-BM, respectively. The BB-BM released TP in the fall but imported TP in the summer (Figure 4b). TP was exported from both coastal marsh stations 6 of 7 times (Figure 4b), with average fluxes of -4.93 mg m⁻² h⁻¹ and -1.89 mg m⁻² h⁻¹ from the BB-SM and FLB-BM sites, respectively. In virtually all samplings at all sites, TKN and TP fluxes were in the same direction as DIN and SRP fluxes (compare Figures 2 and 4), in spite of the fact that DIN and SRP fluxes are often less than 20% of TKN and TP exchanges (Table 2). DIN contributes less to TKN flux in early summer (0.8-9.7%) compared to late summer (14.8-32.2%) in both estuaries (Table 2). Ratios of TKN:TP flux in the Barataria Basin did not show any discernible trend (Table 2). The anomalously high TKN:TP ratio of April BB-SM fluxes was due to a very low, non-significant TP flux (Figure 4b; Table 2). A temporal trend in TKN:TP ratios is seen in FLB-BM flux data, however. In March, the TKN:TP ratio was highest, at 41.8. By June, the ratio was 5.4 and in September the TN:TP flux ratio was only 3.1. Interestingly, June, 1987, TKN:TP ratios were similar (3-5) at all three sites (Table 2).

Relating Fluxes to Constituent Concentrations

Some investigators have used stepwise regression techniques and numerous environmental predictor variables to model measured nutrient fluxes, in an attempt to

determine what factors regulate these fluxes (Wolaver and Spurrier, 1988a,b; Whiting et al., 1989). Results of our correlative analysis of fluxes with concentrations, hydrologic data, and biomass values (see Table 3 for a summary of standing biomass data) are shown in Table 4. This analysis used NH_4^+ , NN, SRP, and DOC exchange data from the BB-SM and FLB-BM sites as dependent variables—the BB-BM flux data had only two sample points (Table 4).

The throughflow flume technique allows analysis of respective flood and ebb tides of a full tidal cycle. In these throughflow flumes, there is thus always an "upstream" end, where water is sampled before exposure to the marsh within the flume, and a "downstream" end—after passing through the flume. Flumes were oriented roughly normal to the adjacent bayou. The "upstream" end on flooding half-tides is thus at the bayou-marsh interface while the "upstream" point on ebbing half-tides is the inland end of the flume. The correlation analysis relates flood and ebb tide fluxes and net tide fluxes to their respective mean upstream and downstream concentrations (Table 4). Upstream concentrations may be viewed as a "source" to the marsh within the flume; downstream concentrations have been "influenced" by this marsh.

Downstream concentrations higher than upstream concentrations indicated nutrient export, as the marsh released nutrients along the length of the flume. Nutrient imports were related to higher upstream "supply" concentrations and flux was a result of uptake along the flume. NN and SRP flood and ebb fluxes at the BB-SM site and NH_4^+ and SRP flood and ebb fluxes at FLB-BM were all correlated with downstream concentrations (Table 4). Two-factor ANOVA testing showed significant interaction ($p \leq 0.05$) between tidal direction (flood, ebb) and the location along the flume (upstream, downstream) with NN exports at the BB-SM site and with SRP exports at both the BB-SM and FLB-BM sites. In each case, most export occurred on flood tide, and concentration differences along the flume were not independent of flow direction—hence the significant interaction. Figure 5 graphically depicts these negative relationships between flux and concentration for NN at

the BB-SM site (Figure 5a) and for NH_4^+ at the FLB-BM site (Figure 5b). When net tide fluxes (Y_2) were tested against various manipulations of the concentration data (X_4 - X_8), BB-SM NN flux (Y_2) correlated with mean ebb tide NN concentration (X_5), mean downstream NN concentration (X_7), and overall mean concentration (X_8). A similar situation appears to occur in the FLB-BM SRP full tide flux correlations, in which SRP release (Y_2) is significantly or highly significantly related to all concentration variables (X_4 - X_8 ; Table 4). The statistical significance of these correlations is an artifact of the large export measured in June (Figure 2c). The remaining correlations with full-tide fluxes were not significant. This is largely due to the small number of tidal cycles sampled ($n=4$ and 3); the effect of small sample size can be seen in the high correlation coefficients but high p-values shown in Table 4.

Full and half-tide NN fluxes, from both the BB-SM and FLB-BM sites combined, correlated with most concentration parameters, and similar relationships were noted with SRP fluxes (Table 4). This analysis of combined data uses measurements pooled from two coastal marshes. They are vegetated by different macrophyte communities and may behave differently. We found a significant ($p \leq 0.05$) interaction (ANOVA, 2-factor) between tidal direction and concentration differences within the flume for DOC export data from the BB-SM site, but not in the FLB-BM DOC uptake data. DOC uptake (FLB-BM) occurred on both flood and ebb tides, and flux was independent of the direction of tide, while most DOC export (BB-SM) occurred on the flooding tide. This phenomenon is illustrated in the combined DOC data. Figure 6 plots the combined half-tide DOC flux against upstream (Figure 6a) and downstream (Figure 6b) DOC concentrations. These overall relationships are highly significant for upstream ($p=0.001$) and significant for downstream ($p=0.032$) values (Table 4), and both have positive slopes. Decomposing the data into site-specific relationships indicates that DOC flux on the two marshes correlates with upstream "source" concentrations in a similar way, although many BB-SM fluxes are exports and all FLB-BM fluxes are imports (Figure 6a). DOC fluxes at the two marsh sites are related to

downstream DOC concentrations in very different ways, however (Figure 6b). The positive slope at the FLB-BM site indicates that, even with enhanced DOC uptake at higher concentrations, this DOC removal has little effect on the large DOC pool in the water inundating the marsh--the FLB-BM response slopes in Figures 6a and 6b are nearly identical. The negative slope at the BB-SM site is a downstream manifestation of the measured flux--as flux becomes more negative [=export], downstream concentrations become greater (Figure 6b). In general, exports were somewhat pulsed and "caused" upstream to downstream concentration gradients while imports were relatively continuous through the tidal cycle and were "driven" by concentration gradients.

Relating Fluxes to Biomass and Physical Variables

Results from other flume studies suggest that significant positive relationships among maximum tidal height, time of year, and measured flux of dissolved nutrients indicate that biological mediation of uptake or release is responsible for observed exchanges (Wolaver and Zieman, 1983; Wolaver and Spurrier, 1988a,b). We found no significant relationships between any nutrient flux at either site and the duration of the associated flood or ebb tide (X_3), or the maximum water depth of tidal inundation (X_9 ; Table 4). Macrophyte biomass, as total standing crop (X_{10}), standing live (X_{11}), and standing dead (X_{10}), failed to correlate with dissolved nutrient fluxes (Table 4). Combined full and half-tide fluxes were also independent of these physical variables.

DISCUSSION

Comparison with Other Flume Studies

Past flume work implicates saline and fresh marshes as sinks for dissolved inorganic forms of N as well as TKN (Table 1). Our flume data show that estuarine marshes in the Barataria Basin and Fourleague Bay nearly always exported NN and, while NH_4^+ fluxes were seasonally variable, overall average fluxes were also net exports (Figure

2a, b). TKN was released by the brackish marsh sites (BB-BM and FLB-BM) in all tides; there was a net export of TKN at the saltmarsh site (BB-SM; Figure 4a). N budgets from these two estuaries, based on denitrification losses (BB=Smith and DeLaune, 1983; FLB=Smith et al., 1985), N-fixation (BB=Casselmann et al., 1981; FLB=Miller, 1983), and deposition of N associated with suspended sediments (BB=DeLaune et al., 1983; FLB=Smith et al., 1985), appear to implicate marshes in both estuaries as N sinks (DeLaune and Patrick, 1980; Madden et al., 1988), however. These marshes may thus be transforming sediment-bound N and atmospheric N fixed into organic matter--both are N inputs to the marsh--into DIN (and TKN, which includes particulate and dissolved organic N) that is subsequently released to the inundating water column.

Virginia and South Carolina saltmarshes take up both SRP and TP on an annual basis (Table 1). Again, MRDP marshes differ, showing export of P. SRP fluxes in the Barataria Basin marshes were variable and low, but averaged to net releases. A relatively large and consistent SRP release was observed in the FLB-BM (Figure 2c). As with TKN, TP fluxes showed patterns similar to SRP, although the magnitude of TP fluxes in the two estuaries we studied were similar (Figure 4b). SRP flux is a much greater fraction of TP flux at the FLB-BM site than at either BB site (Table 2). Only the FLB-BM data showed any discernible pattern in TKN:TP flux ratios, with a high spring value (41.8) decreasing to a ratio of 3.1 in fall (Table 3). This relationship between TN and TP exchange with the marsh closely follows the seasonal pattern of high spring and low fall N:P ratios in Fourleague Bay (Madden, 1986; Madden et al. 1988). The high spring N:P ratios in FLB indicate potential P limitation to aquatic productivity where light is not limiting (Randall and Day, 1987; Madden et al., 1988). At this time, marsh uptake of excess DIN and concurrent release of SRP may moderate excess N conditions in the bay. In the fall, when baywide N:P ratios drop below 16 (Madden et al., 1988), marsh TKN:TP flux ratios are also low (Table 2) although the marsh appears to release large amounts of DIN and SRP to the water column. In fact, SRP regeneration, mobilization, and flux from subtidal FLB

sediments is quite low (Teague et al., 1988; R.Twilley, Univ. Southwestern LA, pers. comm.) while water column concentrations are fairly constant on an annual basis (Madden, 1986). This SRP supply by marsh may be an important seasonal regulator of nutrient limitation on phytoplankton productivity in the bay (Childers, 1989).

Past flume studies have shown uptake of DOC (Chalmers et al., 1985; Wolaver and Spurrier, 1988b), and both uptake (Wolaver et al., 1980, 1983) and release (Whiting et al., 1989) of DON by saltmarshes (Table 1). Our data showed DOC and DON release at the BB-BM site, a pattern of winter and spring DOC and DON release but summer uptake by the BB-SM, and consistent DOC marsh uptake in FLB (Figure 3). Brackish marshes of the BB estuary receive little inorganic sediment input and are highly organic (Conner and Day, 1987). Release of dissolved organics at the BB-BM site may reflect the breakdown and decomposition of the organic [peaty] soils underlying this marsh. The large, highly significant DOC import to the BB-SM observed in August (Figure 3a) was observed during the period of highest air and water temperatures of the year. In response to high environmental temperatures, microheterotrophic activity on the surfaces of the marsh sediment and grasses would be high. The uptake of DOC and DON from the inundating water column may be related to biological uptake of dissolved organic substrates by this active surficial heterotrophic community. DOC import by the FLB-BM is independent of tidal flow direction (2-factor ANOVA), indicating that the process(es) responsible for this flux are temporally constant. Microheterotrophic uptake would tend to be more temporally constant, in comparison to pulsed fluxes which are dominated by release early in the tidal cycle. The seasonal pattern of higher uptake in the summer at the FLB-BM site may also be related to this type of temperature-dependent activity of the heterotrophic community on the surfaces of the marsh (Figure 3a).

Relating Fluxes to Constituent Concentrations

One test of the efficacy of the throughflow flume technique is how fluxes during flood and ebb tides (Y_1) are associated with concentrations. NN flux at the BB-SM site

and NH_4^+ and SRP flux at the FLB-BM site, all exports, correlated with downstream concentrations (X_2), reflecting the marsh source (Table 4). These negative relationships demonstrate the effect that marsh NN and NH_4^+ release may have on overlying water column characteristics (Figures 5a and 5b, respectively). Significant relationships between the magnitude of fluxes measured within marsh flumes and the concentration of the overlying tidal waters have been reported before (Wolaver and Zieman, 1983; Wolaver et al., 1983; Whiting et al., 1989).

Flux of dissolved inorganics from the marsh [export] may be concentration gradient-driven, mainly by diffusion from the sediments (Gardner, 1975; Wolaver and Spurrier, 1988b), or caused by biological processes (Whiting et al., 1989). Diffusive processes depend on a concentration gradient from the interstitial waters to the water column, and release is greater when concentrations in flooding waters are lower (thus increasing the concentration gradient). We saw no significant correlations between upstream, "source" nutrient concentration and fluxes of nutrients characterized by exports (NH_4^+ and NN especially; Table 4), indicating that diffusion is not the only mechanism involved. There was a significant interaction ($p \leq 0.05$) between tidal direction and exports of NN at the BB-SM site and SRP at the FLB-BM site (2-factor ANOVA interaction term $p=0.029$ and 0.04 , respectively); most nutrient release occurred early in the tidal cycles, on the flooding tides. This may be related to a diffusive pulse of inorganic nutrients from the sediments shortly after exposure to the water column. On the other hand, uptake of a constituent by the marsh surface should be related to supply of that constituent. If so, flux will correlate with upstream concentrations. The only 2 cases of net uptake shown in Table 4 are with the BB-SM SRP and FLB-FM DOC fluxes, and in both cases these imports were significantly correlated to upstream, or source, concentrations. Apparently, when the marsh surface imports a dissolved constituent, the magnitude of that uptake is a function of the amount in the overlying water column [available to be taken up] (Figure 5c).

NN net tide flux at the BB-SM site was related to mean ebb tide NN concentration (X_5), mean downstream NN concentration (X_7), and overall mean NN concentration (X_8). In 3 of 4 samplings, the NN flux was higher on the flood tide, probably as a pulsed response early in the tidal cycle (see above). As a result, the NN concentrations in water overlying the marsh on the ebb tide were largely a product of this flood tide flux and Y_2 correlates with X_5 ($r=0.998$, $p=0.002$; Table 4). Both sites exported NN to the flooding water column, and apparently the source (upstream) NN concentration does not control this NN flux. The correlation between downstream NN concentration and export was demonstrated with the flood and ebb tide flux data (Figure 5a; Table 4). Combined NH_4^+ flux data failed to correlate with any concentration variable (Table 4) or show any interaction with the direction of tidal flow (ANOVA, 2-factor), possibly indicating an exchange mechanism in which active biological processes dominated over physical diffusion. The significant correlations between combined flood and ebb tide SRP flux and associated concentrations (X_1 and X_2) were related largely to the large export observed in June at the FLB-BM site.

There were clear differences between DOC flux dynamics at the BB-SM and FLB-BM sites. First, DOC concentrations are much greater in FLB, leading to marsh uptake. In fact, the nearly identical positive slopes of DOC flux vs. upstream (Figure 6a) and downstream (Figure 6b) concentrations indicate that, even with enhanced DOC uptake at higher concentrations, the marsh has little effect on the large DOC pool in the overlying water. ANOVA testing confirmed this, showing no significant interaction between DOC uptake and tidal phase. The negative slope of BB-SM DOC flux vs. downstream DOC concentration (Figure 6b) is because most half-tide fluxes here are exports which increase the DOC content of water passing over the marsh. Figure 6 thus shows an interesting difference between the two estuaries: Fourleague Bay receives large seasonal inputs of freshwater, apparently containing enough DOC to mask uptake by the associated marshes while Barataria Basin saltmarshes, which receive negligible freshwater inputs, are capable

of modifying estuarine DOC concentrations when inundated. Uptake of DOC was greater at higher concentrations, perhaps implicating a biological uptake process with a DOC substrate. DOC release (BB-SM) was higher when overlying water concentrations were lower and most of this release occurred shortly after flooding, on the flood tide (2-factor ANOVA interaction term $p=0.04$). This concentration-dependent, pulsed response implies a more passive process—perhaps one related to the rapid washout of dissolved organics leached from marsh macrophytes (Wetzel and Manny, 1972; Gallagher et al., 1976; Pomeroy et al., 1977)—compared to active biological uptake which would be more temporally constant.

Marsh Fluxes and the Deltaic Cycle

In the last 7000 years, the Mississippi River has built a series of delta lobes in conjunction with regular switching of the main river channel (Kolb and Van Lopik, 1966). The age of these depositional lobes is related to the "deltaic cycle" (Coleman and Gagliano, 1964). At the mouth of a recently-switched river channel, the first stage in the cycle is an embryonic delta becoming subaerial. The new Atchafalaya delta is an example of this first stage. The next stage of the deltaic cycle, exemplified by the Fourleague Bay area, involves horizontal growth of subaerial wetlands. Finally, when these wetlands are isolated from riverine sediments by a channel switch, subsidence begins to exceed deposition. The Barataria Basin is in this deterioration stage. Paradoxically, estuaries in the deteriorating stage of deltaic succession, such as the Barataria Basin, generally show the highest productivity (Gagliano and VanBeek, 1975), although Madden et al. (1988) showed that "younger" estuaries are actually more productive than once believed. The general premise is that nutrients are sequestered by marshes in early stages of the deltaic cycle, particularly in the subaerial [vertical] growth phase, and released by deteriorating marshes in later deltaic succession (Gagliano and VanBeek, 1975; Madden et al., 1988).

Many of the observed differences in flux patterns between Fourleague Bay and the Barataria Basin estuary may be due to the deltaic cycle of the MRDP. The strongest DIN

flux pattern was observed at the FLB-BM site, which is in the developmental stage of the delta succession. Marshes around Fourleague Bay take up DIN in the spring, when the Atchafalaya River freshet delivers large amounts of DIN to the bay. As summer progresses, riverine N inputs drop off and bay DIN concentrations decrease. During this time, the marsh releases DIN to the water column. Thus, the brackish marshes of Fourleague Bay appear to be acting as buffers for the water column by taking up DIN when supply is high and releasing DIN when supply is low (Childers, 1989; Figure 2a,b). In contrast, brackish and saline marshes of the BB estuary released DIN during all samplings. This N export may represent the release of N accumulated during the past depositional stages of the delta cycle (Figure 2a,b).

Phosphorus is adsorbed, and geochemically tied, to particulate matter in the water column (Froelich, 1988; many others), and high sediment inputs to Fourleague Bay from the Atchafalaya River carry bound P onto the adjacent marshes. Deposition of this bound P and incorporation into reducing marsh sediments thus may fuel the consistent release of SRP, observed at the FLB-BM site. Barataria Basin marshes receive lower inputs of suspended sediments and bound P, and thus appear to transform and exchange comparatively little P (Figure 2c). It is probable that the high release of SRP seen at the FLB-BM site (Figure 2c) is also related to the early delta cycle successional stage of this estuary compared to the late-stage Barataria Basin estuary, where SRP fluxes were low.

As we note above, the observed release of dissolved organics by the deteriorating marsh systems of the Barataria Basin--the brackish marsh in particular--was not unexpected. These marshes receive much lower inputs of inorganic sediment, compared to Fourleague Bay, and the organic-rich soils are an excellent source of dissolved organics. But also, Atchafalaya River discharge strongly influences Fourleague Bay (Madden et al., 1988). Odum (1984) proposed a dual-gradient concept for estuarine detrital [organic carbon] processing and transport. His gradient concept has implications to analysis of marsh:water column exchange data: concentrations of DOC generally decline with

increasing salinity due to: 1) supply of DOC by terrestrial freshwater and wetland sources, and 2) dilution by coastal oceanic water low in DOC content (Odum, 1984). Thus, DOC concentrations will be inversely related to estuarine salinity levels. In Fourleague Bay, the salinity gradient is temporal rather than spatial as high spring river discharge decreases baywide salinities to nearly fresh while the fall and winter are characterized by oceanic influence and much higher salinities. The FLB-BM marsh imported DOC (Figure 3a) during all samplings, but the largest uptake rates occurred during the spring and early summer. If Odum's hypothesis holds true in Fourleague Bay, then the marsh may be a sink for DOC, taking up greater quantities when open water concentrations are higher (during lower salinities; Childers, 1989).

SUMMARY AND CONCLUSIONS

Marsh:water column exchanges of dissolved C, N, and P were quantified on three marshes in Louisiana's MRDP using throughflow marsh flumes. The two estuaries in which we sampled marsh fluxes showed some interesting differences: Apparently, the [deltaic cycle] successional stage of an estuary is an important determiner of magnitude and direction of marsh:open water exchanges of dissolved nutrients. The Barataria Basin estuary is in the deteriorating stage of deltaic marsh succession, and both brackish and saltmarshes here supplied NH_4^+ , NN, DOC, DON, and TKN to the inundating water column while both took up SRP. The relatively constant supply of DIN by the Barataria Basin marshes was probably the release of accumulated N as these marshes deteriorate. Coastal brackish marshes of the Fourleague Bay estuary, part of an accreting marsh system in the an early, developmental stage of the deltaic cycle, also exported NH_4^+ , NN, and TKN to the open water estuary. The seasonal flux pattern observed in Fourleague Bay, though, indicated that these marshes acted as a short-term DIN buffer by removing excess DIN in the spring and supplying DIN in the summer and fall, when open water concentrations are lower.

Differences in P and dissolved organics fluxes in these two estuaries were observed. The FLB-BM site exported PO_4 and TP, while this marsh always imported DOC from the inundating water column. Exchanges of POC and DOC in the BB marshes were in the opposite direction. The FLB-BM P export is probably related to the large riverine supply of sediment-bound P to this estuary—freshwater inputs to the BB estuary are minor—which may be remobilized in the reducing soils of the marshes. Future research may show that this release of available P (as SRP) by surrounding marshes to be an important P source to aquatic productivity in Fourleague Bay, which is potentially P-limited for much of the year.

A correlation analysis of fluxes with concentrations, duration of tide, flooding water heights, and macrophyte biomass showed a significant relationship between dissolved nutrient exports and downstream mean concentrations, and between imports and upstream (source) concentrations. A lack of relationships between nutrient releases and upstream (source) concentrations may indicate that passive diffusion was not the only process controlling these marsh:open water exchanges. If it was, higher water column concentrations would decrease the sediment:water concentration gradient and slow diffusive export from the sediments. Failure of 1) NH_4^+ fluxes to correlate with any concentration variables and 2) NN exports to correlate with upstream (source) concentrations may indicate the domination of biologically-mediated processes over physical diffusion driven only by sediment:water concentration gradients.

The microtidal marshes of the MRDP appear to play an important role in estuarine nutrient dynamics. Estuarine water levels, though, are the ultimate controller of these marsh:water column dynamic interactions. Our sampling efforts (between September, 1986 and April, 1989) coincided with an extended period of anomalously low water levels in the Louisiana coastal zone (between July, 1987 and early 1989), causing a decreased marsh flooding regime. If the marsh rarely floods, its importance to the ecosystem is intrinsically reduced. The effect of this reduced marsh flooding regime on the estuary, a

phenomenon we refer to as "ephemeral wetland loss", has been analyzed in Chapter 4 (Childers et al., submitted). This event made the direct sampling of marsh:open water fluxes difficult and resulted in relatively few successful sampling events. Nonetheless, our results do indicate inter-estuary differences [within Louisiana] in marsh:water column interactions as well as differences between dissolved nutrient dynamics in Louisiana estuaries compared to those observed in mesotidal saltmarshes of the southeast Atlantic coast.

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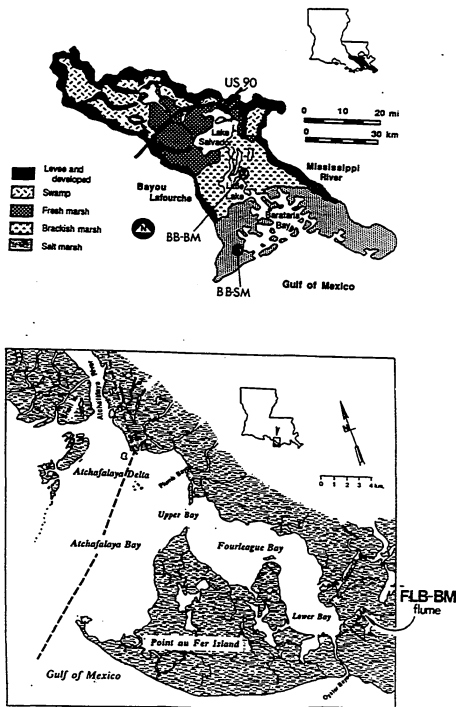


Figure 2-1: Site maps of the two estuaries studied. Each marsh flume site is marked with an "X". Note that both estuaries are oriented on an southeast-northwest axis from the Gulf of Mexico. (a) Barataria Basin estuary (BB)-bounded on the east by the Mississippi River levees, on the west by the Bayou Lafourche levees, and on the north by U.S. Highway 90 (modified from Conner and Day, 1987). (b) Fourleague Bay estuary (FLB)-open to Atchafalaya Bay and River to the north and connected to the Gulf of Mexico by Oyster Bayou to the south.

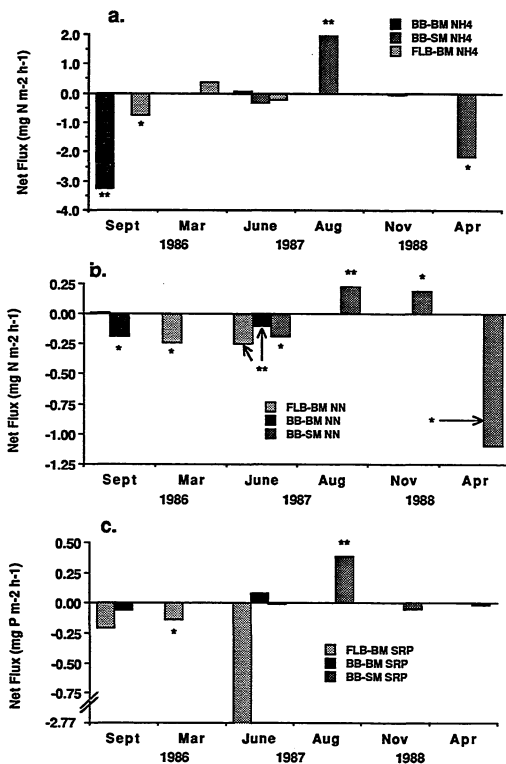


Figure 2-2: Fluxes of (a) ammonia (NH_4), (b) nitrate+nitrite (NN), and (c) soluble reactive phosphorus (SRP) at the Barataria Basin brackish (BB-BM) and saline (BB-SM) sites, and the Fourleague Bay (FLB-BM) site. Positive flux is import to the marsh surface; negative flux is export to the inundating water column. **=flux significantly different from 0 ($p \leq 0.05$); ***=flux highly significantly different from 0 ($p \leq 0.01$).

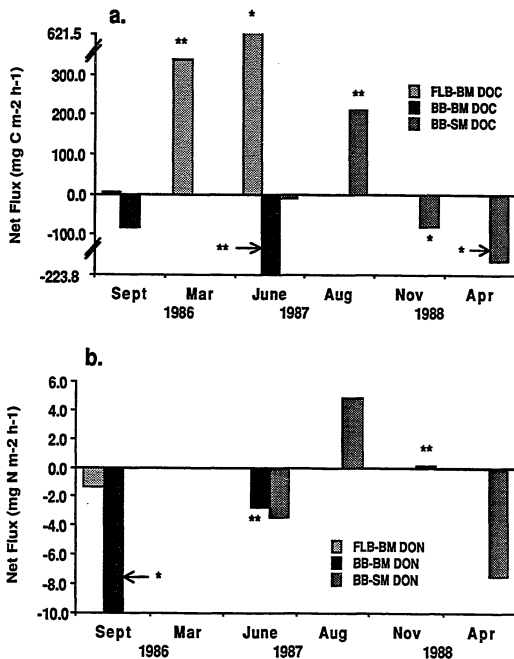


Figure 2-3: Fluxes of (a) dissolved organic carbon (=DOC) and (b) dissolved organic nitrogen (DON) at the Barataria Basin brackish (BB-BM) and saline (BB-SM) sites, and the Fourleague Bay (FLB-BM) site. Positive flux is import to the marsh surface; negative flux is export to the inundating water column. **=flux significantly different from 0 ($p \leq 0.05$); ***=flux highly significantly different from 0 ($p \leq 0.01$).

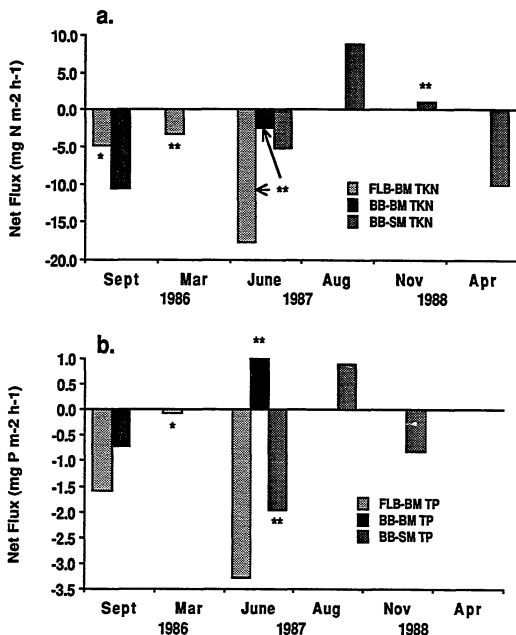


Figure 2-4: Fluxes of (a) total nitrogen (=TKN) and (b) total phosphorus (TP) at the Barataria Basin brackish (BB-BM) and saline (BB-SM) sites, and the Fourleague Bay (FLB-BM) site. Positive flux is import to the marsh surface; negative flux is export to the inundating water column. **=flux significantly different from 0 ($p \leq 0.05$); ***=flux highly significantly different from 0 ($p \leq 0.01$).

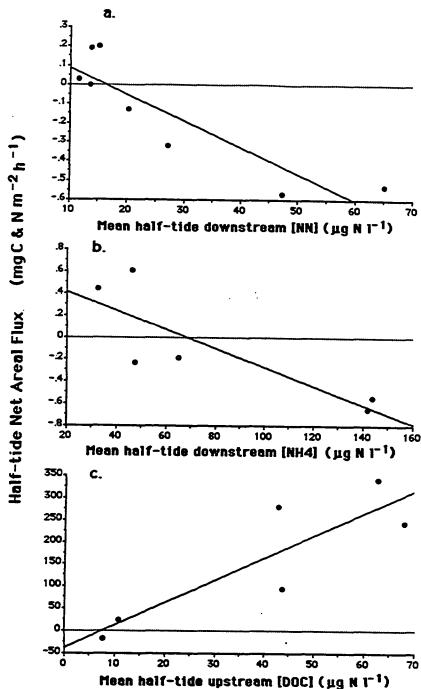


Figure 2-5: Relationships between half-tide flux data (where each point represents either a flooding or ebbing tide flux measured in that flume) and the associated mean concentrations. (a) BB-SM half-tide nitrate-nitrite flux vs. mean downstream NN concentration during that half-tide, (b) FLB-BM half-tide ammonium flux vs. mean downstream NH_4^+ concentration during that half-tide, (c) FLB-BM half-tide DOC flux vs. mean upstream [source] DOC concentration during that half-tide.

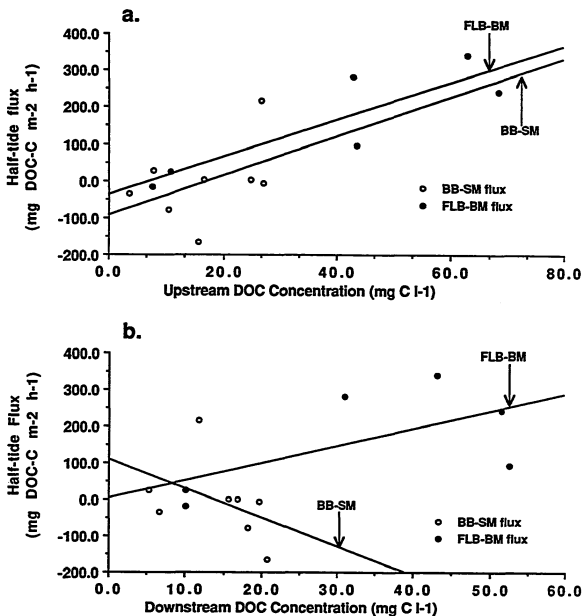


Figure 2-6: Relationships between half-tide DOC flux data from both the BB-SM and FLB-BM sites and: (a) mean upstream DOC concentration during that half-tide, and (b) mean downstream DOC concentration during that half-tide. In both panels, individual site data are plotted together but regressed separately, as indicated.

Table 2-1: A review of net annual dissolved nutrient flux data from recent flume studies. Positive values are uptake by the marsh while negative values are marsh export, and all values are in $\text{g m}^{-2} \text{yr}^{-1}$ for carbon (C), nitrogen (N), or phosphorus (P). **Bold type**=significant ($p < 0.05$) flux; ***Bold italics type***=highly significant ($p < 0.01$) flux. Marsh type abbreviations: SM=saltmarsh; FM=fresh marsh. See text for nutrient abbreviations.

<u>Citation</u>	<u>Type</u>	<u>NH₄⁺</u>	<u>NN</u>	<u>IN</u>	<u>PO₄</u>	<u>TP</u>	<u>DON</u>	<u>DOC</u>
Whiting et al., 1989	SM	2.9	0.62	0.92			-5.9	
¹ Wolaver et al., 1980, 1983	SM	1.9	0.82	31.62	0.20	10.92	4.8	
Wolaver & Spurrier, 1988a	SM				0.46	2.47		
Wolaver & Spurrier, 1988b	SM							2.9
Chalmers et al., 1985	SM							175.2
² Bowden, 1986	FM	+	+					

¹levels of statistical significance for annual fluxes not presented

²quantitative data on annual fluxes not presented

Table 2-2: Contribution of dissolved inorganic nitrogen (DIN) and phosphate (SRP) to total nitrogen (TKN) and phosphorus (TP) flux, in percent of totals, and N:P ratio of totals fluxes. *=cases where the fluxes were in different directions and index had no ecological meaning. BB-BM=Barataria Basin brackish marsh flume; BB-SM=Barataria Basin saltmarsh flume; FLB-BM=Fourleague Bay brackish marsh flume.

<u>Site</u>	<u>Month</u>	<u>Year</u>	<u>DIN/TN</u>	<u>SRP/TP</u>	<u>TKN:TP</u>
BB-BM	June	87	0.8	8.0	2.5
BB-BM	September	86	32.2	8.5	15.0
BB-SM	April	88	32.2	500.0	2535.0
BB-SM	June	87	9.7	0.5	2.66
BB-SM	August	87	25.2	42.2	9.71
BB-SM	November	87	*	6.1	*
FLB-BM	March	87	*	175.0	41.8
FLB-BM	June	87	2.6	84.7	5.4
FLB-BM	September	86	14.8	13.3	3.1

Table 2-3: Standing crop biomass data for each sampling event, in gdw m^{-2} . In each case, the full site data are the mean and standard deviation of samples taken in the creekside marsh ($\approx 1-3$ m from bayou, CkS), the inland marsh (≈ 30 m from bayou, InL), and between the two (Cen).

<u>Site & Date</u>	<u>Biomass</u>	<u>Replicates Data</u>			<u>Full Site</u>	<u>Data</u>
	<u>Type</u>	<u>CkS</u>	<u>Cen</u>	<u>InL</u>	<u>Mean</u>	<u>S.D</u>
A. Barataria Basin Brackish Marsh (BB-BM):						
9/18/86	total	1873.6	2870.0	1882.4	2208.7	572.6
6/23/87	live	1389.2	674.0	1263.2	1108.8	381.8
	dead	798.8	411.6	832.8	681.1	234.0
	total	2188.0	1085.6	2096.0	1789.9	611.7
B. Barataria Basin Saltmarsh (BB-SM):						
6/2/87	live	1334.0	1564.0	1152.8	1350.3	206.1
	dead	906.8	780.8	1066.8	918.1	143.3
	total	2240.8	2344.8	2219.6	2268.4	67.0
8/10/87	live	732.4	597.2	817.6	715.7	111.1
	dead	470.8	616.8	717.2	601.6	123.9
	total	1203.2	1214.0	1534.8	1317.3	188.4
11/16/87	live	429.2	529.6	548.0	502.3	63.9
	dead	454.8	769.6	1117.2	780.5	331.3
	total	884.0	1299.2	1665.2	1282.8	390.9
4/18/88	live	903.6	390.8	366.4	553.6	303.4
	dead	959.2	656.4	650.8	755.5	176.5
	total	1862.8	1047.2	1017.2	1309.1	479.8
C. Fourleague Bay Brackish Marsh (FLB-BM):						
3/8/87	total	2096.4	2314.0	1639.2	2016.5	344.4
6/10/87	live	1470.0	826.0	1122.8	1139.6	322.3
	dead	288.8	675.2	414.0	459.3	197.1
	total	1758.8	1501.2	1536.8	1598.9	139.6

Table 2-4: Correlation matrix of full and half-tide NH₄, NN, PO₄, and DOC fluxes with concentration means, hydrologic data, and biomass data at the BB-SM and FLB-BM sites (see text for abbreviations). Each X(i,j) position shows the r and p-values for that relationship. Where p<0.05, bold type designates significance; where p<0.01, bold italics type designates high significance.

Half Tide Flux		X1		X2		X3	
Site, Constituent	n	Half tide upstr.()	Half tide dnstr.()	Half tide upstr.()	Half tide dnstr.()	Half tide duration	
BB-SM:							
Y1=Half tide NH ₄ flux	8	.60, .118	.10, .814	.06, .881			
Y1=Half tide NN flux	8	.36, .385	.89, .003	.40, .327			
Y1=Half tide PO ₄ flux	8	.82, .013	.89, .017	.04, .932			
Y1=Half tide DOC flux	8	.44, .276	.43, .286	.28, .499			
FLB-BM:							
Y1=Half tide NH ₄ flux	6	.03, .949	.83, .042	.79, .061			
Y1=Half tide NN flux	6	.20, .707	.68, .138	.21, .69			
Y1=Half tide PO ₄ flux	6	.57, .241	.97, .001	.31, .555			
Y1=Half tide DOC flux	6	.87, .025	.63, .161	.02, .972			
Combined data:							
Y1=Half tide NH ₄ flux	14	.46, .101	.15, .605	.01, .984			
Y1=Half tide NN flux	14	.33, .253	.78, .001	.35, .227			
Y1=Half tide PO ₄ flux	14	.62, .018	.91, .0001	.05, .876			
Y1=Half tide DOC flux	14	.81, .001	.58, .032	.16, .585			

Full Tide Flux		X4		X5		X6		X7		X8		X9		X10		X11		X12	
Site, Constituent	n	Full tide flood()	Full tide ebb()	Full tide upstr.()	Full tide dnstr.()	Full tide upstr.()	Full tide dnstr.()	Full tide mean()	Max tidal height	Biomass	Live biomass	Dead biomass							
BB-SM:																			
Y2=full tide NH ₄ flux	4	.71, .289	.29, .712	.69, .308	.41, .588	.59, .413	.44, .561	.05, .952	.12, .881	.54, .461									
Y2=full tide NN flux	4	.59, .111	.99, .002	.81, .189	1.00, .003	.97, .030	.10, .905	.02, .976	.10, .897	.22, .781									
Y2=full tide PO ₄ flux	4	.11, .893	.23, .770	.07, .934	.22, .776	.15, .847	.05, .448	.26, .744	.05, .950	.81, .195									
Y2=full tide DOC flux	4	.50, .553	.03, .972	.59, .414	.16, .838	.31, .688	.63, .367	.02, .980	.21, .788	.57, .432									
FLB-BM:																			
Y2=full tide NH ₄ flux	3	.96, .184	.97, .147	1.00, .011	.98, .128	.98, .142	.99, .091												
Y2=full tide NN flux	3	.46, .695	.37, .760	.16, .895	.56, .626	.41, .729	.74, .466												
Y2=full tide PO ₄ flux	3	1.00, .018	1.00, .001	1.00, .011	1.00, .005	1.00, .008	.20, .874												
Y2=full tide DOC flux	3	.72, .493	.82, .387	.89, .303	.86, .338	.88, .317	.38, .751												
Combined data:																			
Y2=full tide NH ₄ flux	7	.57, .183	.17, .717	.64, .123	.20, .67	.47, .293	.33, .469	.04, .948	.10, .868	.34, .582									
Y2=full tide NN flux	7	.76, .048	.87, .012	.65, .117	.87, .011	.83, .022	.13, .779	.02, .978	.08, .896	.12, .847									
Y2=full tide PO ₄ flux	7	.77, .042	.67, .100	.68, .094	.77, .043	.73, .061	.02, .971	.03, .961	.43, .469	.69, .194									
Y2=full tide DOC flux	7	.78, .039	.80, .030	.88, .008	.77, .043	.85, .016	.19, .686	.19, .720	.47, .424	.85, .067									

Chapter 3

Marsh:Water Column Interactions in two Louisiana Estuaries.

II. Particulate Materials

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ABSTRACT

Throughflow marsh flumes were used to measure organic and total sediment exchanges (POC, PON, and TSS, respectively) between the marshes and water column of two Mississippi River Deltaic Plain (MRDP) estuaries. One, the Barataria Basin estuary, Louisiana, is isolated from significant riverine sediment input. There were significant ($p < 0.05$) imports of 33.9 to 442.9 mg TSS $m^{-2} h^{-1}$ at the Barataria Basin brackish marsh site (BB-BM). The Barataria Basin saltmarsh site (BB-SM) exported TSS in 2 summer samplings, but significant uptake was measured in April (166 mg $m^{-2} h^{-1}$) and November (45 mg $m^{-2} h^{-1}$) during a winter frontal passage event. The other estuary, Fourleague Bay, receives large sediment inputs from the Atchafalaya River, and TSS imports of 22.5 to 118.5 mg $m^{-2} h^{-1}$ were measured at the brackish marsh site here (FLB-BM). Marshes of the Barataria Basin estuary imported 8.52 and 23.1 mg POC-C $m^{-2} h^{-1}$ and 1.58 and 4.54 mg PON-N $m^{-2} h^{-1}$ at the BB-BM and BB-SM sites, respectively. The FLB-BM, however, exported POC and PON during all samplings, with average releases of -10.5 mg $m^{-2} h^{-1}$ and -2.5 mg $m^{-2} h^{-1}$.

We calculated sediment accumulation from fluxes quantified in marsh flumes using site-specific sedimentological data and flooding regimes at each site. Water level records from May, 1987 to April, 1989 showed an extended period of unusually low flooding frequencies. As a result, calculated accretion rates are low, with monthly rates of 0.02 to 0.11 mm at the BB-BM, -0.06 to 0.06 mm at the BB-SM, and -0.18 to 0.08 mm at the FLB-BM marsh flume sites. Actual net sediment deposition, determined by feldspar marker horizon analysis, was 0.7-1.6 mm mo^{-1} at the BB-SM and 0.2-1.3 mm mo^{-1} at the FLB-BM. Even the highest calculated accretion rates, based on flume measurements, were half to one order of magnitude lower than actual measured sediment deposition. This discrepancy was probably because: 1) most sedimentation occurs during episodic events, such as Hurricane Gilbert in September, 1988, which deposited 3.5-15.5 mm of sediment on the BB-SM, or 2) most vertical accretion in MRDP marshes is by deposition of *in situ*

organic matter rather than by influx of allochthonous sediment. Our results affirm the variability of short-term sediment transport and depositional processes, the close coupling of meteorologic forcing and flooding regime to sediment dynamics, and the importance of understanding these interrelated mechanisms in the context of longer-term measurements.

INTRODUCTION

Vertical accretion is the primary mechanism by which intertidal marshes maintain themselves against apparent sealevel rise (ASLR), which is due to the combined effects of eustatic sealevel change as well as local subsidence. In 1972, Redfield suggested that tidal marshes are able to maintain their relative elevation during rapid sealevel rise by sediment capture and increased accumulation of *in situ* belowground production (primarily macrophytic). Letzsch and Frey (1980) stated further that marsh aggradation rates are actually enhanced by coastal submergence. A number of reviews of sediment transport and deposition in estuaries can be found in the literature (Postma, 1967; Meade, 1972; Gardner and Kitchens, 1977; Boto and Patrick, 1978; Stevenson et al., 1988). Stevenson et al. (1985, 1988) recently challenged Redfield's idea (1972) and concluded that many marshes may not be able to maintain their elevations against ASLR, particularly in areas with reduced sediment supply and low tidal amplitudes.

Coastal wetland loss has been reported in many regions with substantial marsh acreage, and a number of mechanisms have been cited. Most studies quantifying coastal marsh loss concluded that a reduced sediment supply was at least partially responsible for the inability of wetlands to accrete sediments at rates equivalent to coastal submergence. Most marsh loss in Delaware Bay, for example, [appears to be] related to shoreline erosion and subsequent marsh drowning processes which are exacerbated by reduced sediment supply to the estuary (Phillips, 1986). Hackney and Cleary (1987) reported that wetland deterioration in some North Carolina estuaries was most often manifested as shoreline erosion triggered by the removal of subtidal sediments when inlets were dredged. On the Chesapeake Bay eastern shore, Stevenson et al. (1985, 1988) reported that much of the marsh erosion begins with the formation of inland ponds that enlarge by marginal slumping. Decreased sediment supply is one hypothesized reason for the initial formation of these inland ponds.

Wetland loss is perhaps most visible in Louisiana's Mississippi River Deltaic Plain (MRDP), which contains about 40% of all coastal wetlands in the contiguous U.S. but where over 80% of all coastal marsh loss is occurring (Gagliano et al, 1981; Turner, 1982; others). Here, wetland loss has been related to a number of impacts, including canal dredging, subsidence, and fringe erosion (Cleveland et al., 1981; Baumann et al., 1984; Swensen and Turner, 1987; others). All of these studies noted that sediment deficits either triggered or exacerbated secondary mechanisms of wetland loss (Templett and Meyer-Arendt, 1988).

Sediments needed for vertical accretion must ultimately be produced by intertidal marshes [as *in situ* production] or supplied by the inundating water column, although significant reworking of those sediments may occur on the marsh surface (Reed, 1988). A few studies have attempted to quantify sediment transport and depositional processes which occur on the marsh surface over individual tidal cycles. This flux of particulates between the estuarine water column and the vegetated marsh during [tidal] inundation has been measured in South Carolina and Georgia, but never before in Louisiana. In both studies, marsh flumes were used to quantify short-term [tidal] exchanges; results are summarized in Table 1. At North Inlet, SC, Wolaver et al. (1988) reported net annual import of both inorganic and organic suspended sediments ($827 \text{ g m}^{-2} \text{ yr}^{-1}$ and $185 \text{ g m}^{-2} \text{ yr}^{-1}$, respectively). Chalmers et al. (1985) reported particulate organic carbon (POC) import to Sapelo Island, GA, marshes (Table 1). This uptake of POC was greatest in the summer, when POC supply to the estuary, by the Duplin River, was greatest. At North Inlet, SC, marshes appear to take up both POC (Wolaver and Spurrier, 1988) and PON (as particulate $\text{N}=\text{PN}$, Whiting et al., 1989; Table 1). Inter-estuary differences in particulates fluxes, especially particulate organics (POM), may vary depending on several geophysical factors, including geomorphology, tidal range, and freshwater influences (Odum et al., 1979). Odum (1984) proposed a dual-gradient concept for estuarine detrital processing and transport which has implications for analysis of marsh:water column exchange data: POC

concentrations generally decline with increasing salinity due to 1) supply by terrestrial freshwater and wetland sources, and 2) dilution down-estuary by coastal oceanic water low in POC content (Odum, 1984). Direct measurements of marsh:water column particulates fluxes have never been made in microtidal MRDP estuaries. In this paper, we report preliminary results from short-term particulates flux measurements conducted with throughflow marsh flumes in the Barataria Basin (BB) and Fourleague Bay (FLB) estuaries, Louisiana.

When estuarine water floods the marsh surface, sediment may be [passively] deposited and/or [actively] removed, by processes such as biological filtration, from the water column (Chalmers et al., 1985; Wolaver et al., 1988; others). In marshes where *in situ* production cannot compensate for ASLR, this depositional process is another critical mechanism by which marshes build vertically. Baumann et al. (1984) used marker horizons to quantify sedimentation rates in BB saltmarshes and in the coastal marshes surrounding FLB, located near the Atchafalaya River. They reported a difference in both the seasonality of deposition and the sources of sediment at the two sites. In BB, 36-40% of all sedimentation in the years studied (1975-1979) was related to hurricane and tropical storm activity. In non-hurricane years, though, 70-80% of total annual sediment deposition occurred during winter frontal passages. For both tropical storms and frontal passages, resuspension of bottom sediments in nearby water bodies appeared to supply most of the deposited sediment. In FLB, however, 70-95% of the annual sediment accretion occurred during the spring, with the Atchafalaya River as the key sediment source (Baumann et al., 1984). Stumpf (1983) documented a similar storm-dominated accretionary process for microtidal Delaware marshes. Rejmanek et al. (1988) described an even stronger dependence on storm-supplied sediments in freshwater marshes of upper Fourleague Bay. In this paper, we use short-term particulates flux data to calculate deposition and vertical accretion of sediments in these two Louisiana estuaries and compare

these calculated rates to actual net accretion measured at the same sites using feldspar marker horizons.

MATERIALS AND METHODS

The two Louisiana estuaries in which we quantified sediment flux and deposition have markedly different sediment transport characteristics. The Barataria Basin estuary, located just west of the Mississippi River (Figure 1a), is characterized by high wetland loss rates. The levees which enclose it have virtually eliminated riverine inorganic sediment inputs and, as a combination of this and regional subsidence, marshes here are deteriorating (Boesch et al., 1983). This type of land loss scenario is representative of the "deterioration" stage of the deltaic cycle (Coleman and Gagliano, 1964; Kolb and Van Lopik, 1966; see Conner and Day, 1987, for a detailed site description). The second study area, Fourleague Bay (Figure 1b), is one of the few areas in the Louisiana coastal zone where marshes are relatively stable. The nearby Atchafalaya River provides a sediment supply adequate to maintain marshes against ASLR and prevent major local wetland loss. Fourleague Bay is thus in the young, "depositional" stage of deltaic succession (Coleman and Gagliano, 1964; Kolb and Van Lopik, 1966; see Madden et al., 1988 for a detailed site description).

Flux Measurements

Marsh flumes were built in brackish and saline marshes of the Barataria Basin estuary and in coastal brackish marshes of Fourleague Bay. We will refer to these flumes as BB-BM, BB-SM, and FLB-BM, respectively. The flumes consist of clear, corrugated fiberglass panels (each 0.66 m wide and 2.7 m long) which form vertical walls 2 m apart. Boardwalks on either side prevent disturbance of the marsh surface. The flumes, approximately 30 m long each, serve to prevent lateral water movement, without altering normal flow, during marsh flooding. The panels are removed after each sampling to prevent long-term panel effects, such as shading, edge scouring, and wrack accumulation.

Between May, 1987 and April, 1989, we continuously monitored water levels at the BB-SM and BB-BM sites with Richards-type water level recorders (Weathertronics Inc. Model 6510). Water levels at the BB-SM site were correlated with discrete water level measurements made at the FLB-BM coastal marsh site during 9 sampling trips, so BB-SM data was used to determine the frequency of marsh inundation at both sites.

A throughflow design is used in these flumes. This modification was necessary for use in Louisiana's expansive, microtidal, low energy estuarine marshes. Replicate water samples were drawn simultaneously from both ends of the flume every 30-45 minutes during a tide. We determined fluxes of constituents by comparing instantaneous flux into the flume with that out of the flume. Our method follows the convention that negative fluxes are exports from the marsh surface to the water column and positive fluxes are imports. Replicate water samples were filtered through pre-ashed, pre-weighed Whatman GF-F glass fiber filters and filters were immediately frozen in the field. At each sampling, aboveground macrophyte biomass samples (0.25 m^2) were taken at randomly chosen sites near the creekside, central, and inland areas of the flume.

Filters were dried and weighed for total suspended sediment (TSS) data, then acidified and combusted with a Perkin-Elmer Model 240-B elemental analyzer to determine particulate organic carbon (POC) and nitrogen (PON) concentrations. Biomass samples were separated into live and dead components, dried at 60°C , and weighed. Fluxes of TSS, POC, and PON (calculated as mg m^{-2} marsh inundated h⁻¹ inundation) were based on these concentrations multiplied by the associated water flux. We determined the statistical significance of measured fluxes by comparing upstream and downstream concentrations using paired t-test analysis (Starview 512+® Statistical Package). Further details of the flume design, sample treatments, flux computations, and data interpretation are presented in Chapters 1 (=Childers and Day; 1988) and 2 (=Childers and Day, submitted).

In this throughflow flume technique, full tide fluxes may be decomposed into flood tide flux, from bankfull to slack high water, and an ebb tide flux, from slack high water to bankfull again. There is thus always an "upstream" end, where water is sampled before exposure to the marsh within the flume, and a "downstream" end--after passing through the flume. With flumes oriented roughly normal to the adjacent bayou, "upstream" on flood tides is at the bayou end of the flume while "upstream" on ebb tides is at the inland end of the flume. To ascertain potential mechanisms responsible for particulate flux patterns, we correlated full and half-tide fluxes with various mean concentrations, tidal duration, maximum water height, and macrophyte biomass. This correlation analysis relates half-tide fluxes (flooding or ebbing) and full-tide fluxes (both) to their associated mean upstream and downstream concentrations.

Accretion Rate Calculations

Soil bulk density and organic matter content were determined for creekside and inland locations at each flume site. At each location, triplicate cores (10 cm diameter X 30 cm length) were taken without compacting the sediment column. The cores were sliced into 5 cm portions and the 0-5 cm segments were immediately frozen for analysis. In the lab, each segment was subsampled to a known volume and weighed. The subsamples were dried at 90°C to a constant weight and reweighed for [dry] bulk density. Samples were then ground, ashed at 435°C, and weighed. Organic content was calculated as [(dry wt.-ashed wt.)/dry wt.]*100 (Blake, 1965; Allen, 1974).

In April, 1988, clay feldspar marker horizons were established adjacent to the BB-SM and FLB-BM flumes. We placed a thin layer of white feldspar over a 0.25 m² area of marsh adjacent to the creek (CkS), near the inland end of each flume (30 m from the creek, InL), and between the two (Cen). Triplicate 6 cm diameter cores were taken from each location in September and October, 1988 at the BB-SM site and in December, 1988, and April, 1989 at the FLB-BM site. Cores were immediately sealed and frozen. In the lab,

we laterally sectioned the frozen cores and measured the amount of sediment accumulated over each horizon.

Areo-temporal TSS fluxes were converted to an accumulation rate (cm mo^{-1}) by: 1) dividing flux by the bulk density of the sediment (g cm^{-3}); 2) multiplying by 7.2×10^{-5} (to convert cm^2 to m^2 , g to mg , and hours to months; see below), then; 3) dividing through by the inorganic fraction (as %) of the sediment. We assumed that the uptake of TSS is an input only to the inorganic fraction of the soils for two reasons. First, particulate organics (POM) are only a small fraction, by weight, of both the TSS flux and of the sediment organic matter (SOM) content (apparently, most SOM is produced *in situ*). Also, much of the POM imported by the marsh may actually have been filtered and assimilated by infauna (particularly mussels; Jordan and Valiela, 1982, 1983; Wright et al., 1982), while filtered inorganic sediments are subsequently released (Bertness, 1984). When the accumulation value was corrected for the total time in which the marsh was inundated that month, the result was an absolute accretion value, in cm :

$$\frac{\left[\frac{\text{TSS flux}}{\text{Bulk density}} * 7.2 \times 10^{-5} \right]}{\left[\frac{100 - \% \text{OM}}{100} \right]} = \text{cm mo}^{-1} * \% \text{ month inundated} = \text{cm}$$

The conversion factor was determined by dimensional analysis:

$$\frac{\text{mg} * \text{m}^{-2} * \text{h}^{-1}}{\text{g} * \text{cm}^{-3}} * \left[\frac{\text{m}^2}{10^4 \text{ cm}} * \frac{\text{g}}{10^3 \text{ mg}} * \frac{720 \text{ h}}{\text{mo}} \right] = \text{cm mo}^{-1}$$

The accretion rate calculation model shown above was based on several assumptions. Positive TSS flux measured in the marsh flumes is uptake by the marsh; we assume this represents direct addition to the mineral fraction of the marsh sediments and that the uptake is manifested as sedimentation and vertical accretion. Most sedimentary organic matter is produced *in situ* and we assume that, over our short-term studies, the average rate of production is sufficient to maintain this % organic matter at a constant level.

We extracted the % time of inundation value at BB-BM and BB-SM sites directly from our water level data. The relationship between water levels at our two coastal marsh sites allows the use of BB-SM flooding frequencies at the FLB-BM site.

RESULTS

During the 2.5 years in which we sampled these three marsh flumes, there was an extended period of anomalously low sealevels on the Louisiana coast (Chapter 4=Childers et al., submitted). The result was dramatically reduced estuarine marsh inundation regimes between Summer, 1987 and early 1989. During this time, the BB-SM was inundated 32-92% less frequently, on a monthly basis, compared to a 5-year average flooding regime (Figure 2; Baumann, 1987). Inundation frequency was even more reduced further inland, where the BB-BM was flooded less than 10% of the time between July, 1987 and August, 1988 (Figure 2) and did not flood for 92 consecutive days between mid-December, 1987 and mid-March, 1988. Water levels at the FLB-BM site were closely monitored during each sampling trip, and regressions of these discrete water level observations with concurrent measurements from the BB-SM site made using a continuous water level gauge were significant ($p \leq 0.05$), although the two sites are about a half tidal cycle out of phase. The two are approximately the same distance from the Gulf. Given this relationship, we assumed that the inundation regime measured at the BB-SM site was representative of the FLB-BM site as well.

Flux Results

The marshes of the Barataria Basin estuary exported small amounts of particulate organics (POC and PON) in the summer and imported POC and PON in the spring and fall (Figures 3a,b). In most cases, the marsh:water column exchanges of POC and PON were in the same direction and where they were not, the POC flux was not statistically significant (ex: in June at both the BB-BM and the BB-SM sites, Figure 3a). The coastal brackish

marsh site in the Fourleague Bay estuary exported particulate organics on all tides sampled (Figure 3). The largest release of POC and PON occurred in the summer (June), the same time of year when POM exports were measured at the BB-BM and BB-SM site. Net POC and PON fluxes from the FLB-BM site averaged $-10.5 \text{ mg m}^{-2} \text{ h}^{-1}$ and $-2.5 \text{ mg m}^{-2} \text{ h}^{-1}$, respectively (analytical problems precluded use of March PON data).

Suspended sediment (TSS) fluxes measured over individual tidal cycles in Louisiana marshes were quite variable (Figure 4). The BB-BM site imported TSS in both June and September, including the greatest sediment import measured at any site ($=443 \text{ mg m}^{-2} \text{ h}^{-1}$ in September). The BB-SM actually exported TSS during both summer samplings, but neither flux was significant and the significant measured imports in April and November were large enough to generate an average sediment uptake of $49.6 \text{ mg m}^{-2} \text{ h}^{-1}$ (Figure 4). TSS fluxes at the FLB-BM site ranged from a large export in March to imports in June and September (Figure 4), with average release of $-23.2 \text{ mg m}^{-2} \text{ h}^{-1}$. In most samplings, the fraction of TSS flux related to POC+PON exchange was inversely related to the magnitude of that flux: when TSS flux was low, the % of that flux attributable to particulate organics flux was relatively high, and visa versa. Particulate organics fluxes appear to be less variable than TSS flux, and seem somewhat independent of inorganic sediment exchange as well.

In comparisons of particulate fluxes and concentrations in water overlying the marsh, the flux may be a result of the supply of material and is thus dependent on the concentration gradient along the flume. On the other hand, observed concentration gradients may be a result of measured flux, with the flux generating this gradient. In throughflow flumes, upstream concentrations may be viewed as a "source" to the marsh within the flume; downstream concentrations have been "influenced" by this marsh. Flood and ebb (Y_1), and net (Y_2) tide TSS fluxes at the BB-SM site were significantly related to upstream (source, X_1 and X_6) and downstream (X_2 and X_7) TSS concentrations in the tidal water (Table 2). In both cases, low concentrations are observed with TSS exports

while higher sediment loads in flooding waters appear to cause sediment uptake (Figure 5). We found no correlation between POC and PON concentration (as upstream, downstream, flood tide, ebb tide, or whole tide means) and whole or half-tide POC and PON fluxes at either site (see Table 2 for correlation matrix), however. In fact, POC and PON fluxes at the BB-SM and FLB-BM sites were not correlated with the time or depth of inundation or any measure of standing crop macrophyte biomass (Table 2; for biomass data, see Table 3).

Calculated and Actual Sediment Accretion

Accretion data calculated from marsh flume TSS flux data, in conjunction with site-specific sedimentological characteristics (Table 4), show low rates of sediment deposition in both estuaries (Figure 6). In three samplings, actual losses [erosion] of sediment were observed (shown in Figure 6 as negative accumulation). The BB-BM accreted sediments in both months sampled (Figure 6). At the BB-SM site, sediment was deposited in the spring and fall—the summer losses of sediment were associated with low-magnitude (non significant) TSS exports (Figure 6).

Depths of actual [measured] sediment accretion at the BB-SM and FLB-BM sites, determined from the feldspar marker horizon analysis (Figure 7), were compared to the sediment deposition rates calculated from flux data (Figure 6). In all samplings, there is a clear spatial gradient of maximal sediment deposition in the creekside locations (CkS) to minimal deposition at the site furthest from the bayou (InL). From April through September, 1988, net deposition at the BB-SM site was 0.7-1.6 mm mo⁻¹. By October, 1988, only a month later but after Hurricane Gilbert, 16, 7, and 4 mm of new sediment had been deposited at the CkS, Cen, and InL locations, respectively, at this site (Figure 7). The April through December accretion rate at the FLB-BM site was 0.2-1.3 mm mo⁻¹ (range=InL to CkS locations) and measured accumulation at the FLB-BM site after one year totalled 2-14 mm for a net accumulation rate of 0.2-1.1 mm mo⁻¹ (Figure 7). In all

cases, actual sediment was deposited to total depths greater than the accretion rates calculated from marsh flume flux data.

DISCUSSION

Marsh Flux Results

Particulate organic matter (POC and PON) was imported at the BB-BM and BB-SM sites, but FLB-BM fluxes were all exports. POC fluxes in the BB and FLB marshes varied from 0.5 to 23.5 mg m⁻² h⁻¹ (Figure 3a). Duplin River, GA, marshes imported POC from the inundating water column, with little seasonal variability in patterns of exchange (Chalmers et al., 1985). The mean uptake of POC there, assuming 6 hours of inundation on the average tide, was 60 mg m⁻² h⁻¹. Their average is markedly greater than our measured POC fluxes, and may be related to the higher tidal amplitudes at Sapelo Island--meso- and macrotidal marshes tend to import sediments while those in microtidal settings often export sediments (Stevenson et al.; 1988). In general, POM fluxes were also greater in Fourleague Bay than in the Barataria Basin estuary (Figure 3). This difference is probably related to the higher POC and PON concentrations at the FLB-BM compared to Barataria Basin values, and these higher POM concentrations are probably due to the strong riverine influence in Fourleague Bay. Atchafalaya River discharge imposes a temporal rather than a spatial salinity gradient on Fourleague Bay, and the estuary is mostly fresh for much of the spring and early summer. The organic content of sediments suspended in the low salinity "ends" of estuaries tends to be higher because of freshwater POM inputs (Folger, 1972; Nichols, 1972; Odum, 1984). If this holds true in Louisiana estuaries, then 1) flux should be greatest in the spring, when freshwater inflow is highest, or 2) there should be a relationship between POM supply, as upstream concentration, and POM flux. In fact, the greatest POM flux at the FLB-BM site was in June, when river discharge is

intermediate (Figure 3), and there was no significant relationship between POC or PON flux and upstream POC concentration at this site (Table 2).

Wolaver et al. (1988) reported imports of both inorganic and organic sediments to a saltmarsh at North Inlet, SC. They estimated total marsh uptake at about 1000 g TSS $\text{m}^{-2} \text{yr}^{-1}$. Assuming that their marsh was flooded 50% of the time, this represents a mean flux of 228 mg TSS $\text{m}^{-2} \text{h}^{-1}$. Their greatest fluxes were measured in the fall and winter, with imports as high as 1000 mg TSS $\text{m}^{-2} \text{h}^{-1}$. Flux data from the BB-BM and FLB-BM sites were within the range of their average while fluxes at the BB-SM site were well below it (Figure 4).

Climatological forcing is often a more important regulator of water levels in microtidal, expansive MRDP estuaries [and hence on marsh:water column exchanges] than astronomical tidal variability (Baumann, 1987; Chapter 4=Childers et al., submitted). As a result, sedimentation in microtidal marshes may be strongly event-related. The November sampling at the BB-SM site, which took place during a strong early winter frontal passage, demonstrates this well. The day before the sampling, southeasterly winds increased from 15 to 25-30 kts. and water levels increased markedly as winds pushed coastal water into the estuary. The astronomical component of water level variation during this time was semidiurnal and low, so virtually all of the observed water level increase (>50 cm) and associated long-duration (>28 hours) marsh flooding was related to the southerly winds. During this wind-induced flooding event, a large influx of TSS to the saltmarsh occurred (=44.6 mg $\text{m}^{-2} \text{h}^{-1}$; Figure 4), supporting the conclusion of Baumann et al. (1984) that much of the sediment deposition on BB coastal marshes occurs during winter storms. As expected, overall TSS flux was dominated by the exchange during rising water levels (42.1 mg $\text{m}^{-2} \text{h}^{-1}$ during flood tide compared to 2.5 mg $\text{m}^{-2} \text{h}^{-1}$ during ebb tide).

The spring TSS export at FLB-BM (Figure 4) is paradoxical. In the spring, the Atchafalaya River discharges its greatest volume of water and sediment into Fourleague Bay. Baumann et al. (1984) showed that the greatest sediment deposition in marshes of the

lower bay occurs at this time, but Rejmanek et al. (1988) reported very little riverine impact on sedimentation in fresh marshes of the upper bay. Notably, in the March sampling we measured higher TSS concentrations on the ebbing tide (representing water from the marsh) than on the flooding tide (as source water from the bayou). Earlier that day, a frontal system generated a rainstorm during low tide marsh exposure which probably disturbed the surface sediment enough to enrich flooding waters with resuspended sediments as the tide came up, perhaps altering the ebbing tide concentration patterns seen in the flume. This is consistent with observed TSS exports from South Carolina marshes during low tide rainstorms (Settlemyre and Gardner, 1975; Ward, 1981; Chalmers et al., 1985). If so, this sediment export was real, but it is a paradox in the expected spring sediment influx scenario for lower Fourleague Bay because it was caused by an isolated low tide rainstorm.

Flood and ebb tide TSS flux and mean upstream ("source") TSS content are closely related (Table 2), indicating that flux from the marsh [export] switches to uptake by the marsh as concentrations in the flooding tide increase (Figure 5a). This gradient-driven concentration control implies that passive mechanisms (driven primarily by the amount of sediment supply), such as gravitational settling, are dominating the TSS flux patterns here. Interestingly, TSS flux vs. downstream TSS content show a similar positive relationship (Figure 5b). The slopes of the upstream and downstream TSS concentration vs. flood-ebb tide flux plots relationships are of similar magnitude (compare Figures 5a and 5b), implying that the marsh within the flume has little effect on water column concentrations. Apparently, uptake by the first 30 m of the marsh, as measured with our marsh flumes, removes little of the available sediment; much of this sediment load must be transported further into these flat, expansive marshes.

Active biological processes may play a significant role in particulates exchanges, especially POM fluxes. In this case, flux may be a function of the amount of time the marsh is exposed to tidal water or, perhaps, of macrophyte biomass (Wolaver and Zieman,

1983; Wolaver and Spurrier, 1988). We compared correlation coefficients and p-values from organic (POC and PON) and [largely] inorganic (TSS) fluxes (Table 2) to ascertain whether active processes were involved in our exchange data. In all cases, the flood and ebb tide time duration (X_3 ; Table 2) explained more of the variability observed in flood and ebb tide POC flux than in TSS flux (manifested as a greater r and lower a p-value, Table 2). POC flux was thus more closely related to the amount of time the marsh was exposed to overlying water, suggesting that temporally constant processes, such as biological filtration, may have exerted slightly more control over particulate organics fluxes than over TSS fluxes. Also, correlations showed that live and dead standing crop biomass (X_{11} and X_{12} , respectively; Table 2) explained more of the observed variability in combined (as BB-SM+FLB-BM) TSS flux data than in POC flux results (compare r and p-values, Table 2). The positive TSS flux vs. live and dead biomass relationship, while not significant, was probably because greater densities of standing macrophytes better sheltered the inundating water column, enhancing settlement and inhibiting resuspension. This closer relationship between biomass parameters and TSS flux, compared to POC flux, is further indication that TSS flux may have been dominated by physical processes while particulate organics were exchanged both actively [biological uptake] and passively [physical effects].

Calculated and Actual Sediment Accretion

Sediment accretion rates calculated from flux measurements (Figure 6) were lower than actual deposition determined with marker horizons (Figure 7). The negative accumulation rates--perhaps representing erosion--were associated with measured TSS export. At the BB-SM site, the summer "losses" of sediment (in June and August) were both calculated from non-significant TSS fluxes [exports]. As discussed above, the "loss" of sediment in March at the FLB-BM site may have been a result of sediment resuspended by rain falling on the exposed marsh. Deposition occurred on the other two sampling events (Figure 6). The accretion rates calculated from tidal flux data were low compared to measured sediment accumulation, suggesting that Barataria Basin saltmarshes and

Fourleague Bay brackish marshes either accrete sediments at low rates or export TSS during routine tidal inundation. A number of past studies have demonstrated variability in sediment fluxes with depth and duration of flooding events (Stevenson et al., 1988). In work on the macrotidal southeast coast of England, Reed (1988) found that most sediment delivery to the marsh surface occurred during spring tides, while neap tides--when flooding events were short and depths of inundation much lower--were characterized by variable fluxes and even sediment exports. The low sediment accretion rates, calculated from TSS fluxes, may also have been related to the anomalously low water levels observed in 1987-88. All flux measurements at the BB-SM site were made after the onset of this low-flooding event, and comparison of sediment accretion rates from this site with those at the BB-BM and FLB-BM sites (where all flux measurements were made before the low water level event) indicated no obvious differences. Sediment flux data apparently showed that sediment uptake in MRDP marshes during this extended period of low water levels, per hour flooded, were no greater than during years with more average water levels.

There was a clear gradient of sediment accretion into the marsh, with greatest deposition at the locations adjacent to the tidal creeks (CkS)--nearest to the sediment source (Figure 7). Baumann et al. (1984) reported the same gradient in lower Fourleague Bay, for only creekside and inland locations. Sediment accumulation data from the freshmarshes of upper Fourleague Bay did not show this type of trend (Rejmanek et al., 1988) while Barataria Basin freshmarshes did show a gradient (Hatton et al., 1983). The BB-SM data encompassed spring and summer (April, 1988 to September, 1988=149 day duration; April, 1988 to October, 1988=187 day duration). Baumann et al. (1984) measured a spring and summer streamside sediment accumulation of 7.7 mm in BB marshes (with 51.5% of total annual accretion during these two seasons). Our September accretion rate of 8 mm is in close agreement with Baumann et al. (1984), while the October accumulation total of 23.5 mm greatly exceeds it. In the month between the two BB-SM samplings, the eye of Hurricane Gilbert passed across the southern Gulf of Mexico between September 14

and 18, 1988, immediately after the first cores were taken at the BB-SM site. We sampled a second time only a month later, and found that 16, 7, and 4 mm of new sediment had been deposited at the creekside, central, and inland locations, respectively, during this time (Figure 7). Apparently, the weather generated in the Louisiana coastal zone by Hurricane Gilbert caused a major pulse of sediments to the saltmarshes of the Barataria Basin estuary. This follows the storm-dominated depositional scenario suggested for MRDP estuaries (Baumann et al., 1984; Rejmanek et al., 1988; Conner et al., 1989).

April through December streamside accretion at the BB-SM site totaled 11.0 mm (250 day duration), and after about one year (369 days) 13.5 mm of sediment had accumulated (Figure 7). Both values are nearly equal to spring and summer (there was no differentiation between fall and winter) and annual sediment deposition measurements of 12 and 13 mm, respectively, for this estuary (Baumann et al., 1984). The major accretion event seen in October at the BB-SM site was not observed in the FLB-BM samples (Figure 7). A probable explanation lies in the relative locations of the BB-SM and FLB-BM sites. Hurricane Gilbert generated strong easterly and northeasterly winds in the Louisiana coast as it passed across the southern Gulf. In this situation, the BB-SM was downwind of several large bays (Caminada and Barataria) and thus was "downstream" from wind-generated and transported sediments resuspended in these open water areas. The FLB-BM site, however, was east and windward of Fourleague Bay and thus appears to have received little bay sediment resuspended by these easterly and northeasterly winds (see Figure 1). Others have noted the importance of storm orientation and wind direction to sediment transport and deposition in estuaries (Dyer, 1986) and MRDP marshes (Rejmanek et al., 1988; Conner et al., 1989).

Assuming uniform monthly deposition, the BB-SM accreted sediment at rates of 0.7 to 1.6 mm mo⁻¹ (InL to CkS), without the hurricane event. During these months (April through September), the highest sediment accretion rate calculated from TSS flux data (in April) was 0.07 mm mo⁻¹. Actual accretion rates at the FLB-BM site were 0.16 to

1.1 mm mo⁻¹, based on the 369 day duration data (Figure 7), and the maximum accretion rate, calculated from June tidal flux data (Figure 6), was 0.08 mm mo⁻¹. At the FLB-BM site, the value calculated for June was half the low range values observed in actual marker horizon data, while the deposition rate calculated from BB-SM TSS April flux data was an order of magnitude lower than the actual rate. During a "normal" tidal flooding event, little sediment is exchanged, and export may even be seen. Most sediment transport onto, and deposition on, the marsh probably occurs during fairly irregular, weather-induced events, as previously discussed (Baumann et al., 1984; Rejmanek et al., 1988).

One problem with sampling sediment exchanges over individual tidal cycles is the difficulty in sampling during these important episodic events, such as Hurricane Gilbert. The differences between calculated and measured accretion rates reflect this problem of extrapolating short-term measurements to longer time scales when the sedimentation process is largely episodic, or event-driven. It is also difficult to differentiate between *in situ* production of organic matter, which accumulates both surficially and at depth in marsh soils, and to quantify the importance of autochthonous particulate matter inputs to feldspar marker horizon studies. Sediment accumulation rates calculated from our flume data represent only contributions from waterborne particulates. Recent studies have shown that, in fact, a large fraction of sediment accumulating over marker horizons in MRDP marshes is autochthonous organic material, from dead and decaying plants, to the extent that these marshes accrete vertically primarily by organic matter accumulation rather than terrigenous sediment deposition (DeLaune et al., 1987; Rejmanek et al., 1988). The large discrepancy between sediment accretion rates calculated from TSS flux data and actual sediment deposition may be this contribution of *in situ* organic matter.

SUMMARY AND CONCLUSIONS

These data suggest some interesting patterns in sediment transport and deposition in the two Louisiana estuaries being studied. Past work quantifying sediment accumulation in

Louisiana's estuarine marshes has generally overlooked the dynamics of short-term sediment exchange with the marsh surface. Sediment flux data from throughflow marsh flumes demonstrates how variable this transport may be. An understanding of the processes occurring over individual flooding events and on time scales of hours is critical to any study of sediment dynamics in estuaries. It is important to link the variability observed in short-term sediment transport to important meteorological events and subsequent hydrologic forcing. Sediment accretion rates calculated from these short-term flux measurements must also be compared to actual measurements of net sediment deposition in the same marshes.

Past research efforts quantifying marsh sedimentation in coastal Louisiana have identified the magnitude of the wetland loss problem, and many future efforts will be directed towards solutions to this problem. Environmental engineering, which directs minimal human intervention toward enhancement of existing natural processes, will be the focus of some of these efforts. As an example of this type of solution, freshwater diversions into the Barataria Basin estuary have been proposed (Templett and Meyer-Arendt, 1988). We have little *a priori* knowledge of how natural hydrologic processes will distribute this new sediment input or if the sediments will, in fact, be deposited on marsh surfaces. Using the flume method and accretion model we present here, short-term flux measurements coupled with water level data and measurements of actual deposition may be used to determine deposition rates in these applied situations. Thus, these simple and easily applied techniques may represent an important part of future applied research efforts to solve Louisiana's coastal land loss problems.

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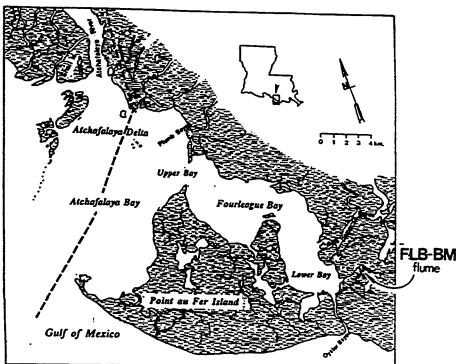
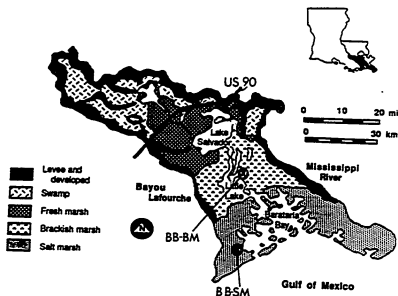


Figure 3-1: Site maps of the two estuaries studied. Each marsh flume site is marked with an "X". Note that both estuaries are oriented on an southeast-northwest axis from the Gulf of Mexico. (a) Barataria Basin estuary (BB)-bounded on the east by the Mississippi River levees, on the west by the Bayou Lafourche levees, and on the north by U.S. Highway 90 (modified from Conner and Day, 1987). (b) Fourleague Bay estuary (FLB)-open to Atchafalaya Bay and River to the north and connected to the Gulf of Mexico by Oyster Bayou to the south.

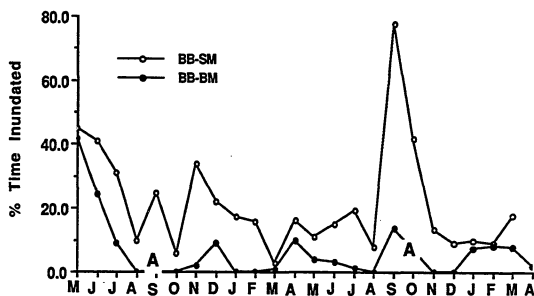


Figure 3-2: Flooding regimes (hours of marsh inundation as percent of total hours each month) at the BB-BM (hollow) and BB-SM (filled) sites for May, 1987 through April, 1989, as determined from continuous water level records from each location. A=missing data.

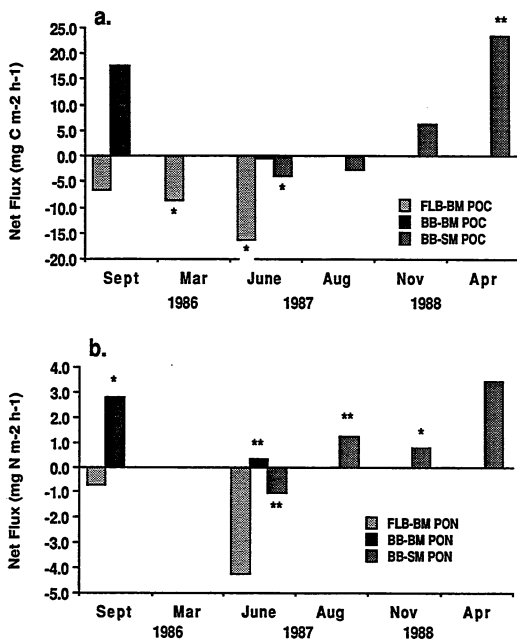


Figure 3-3: Fluxes of (a) particulate organic carbon (=POC) and (b) particulate organic nitrogen (PON) at the Barataria Basin brackish (BB-BM) and saline (BB-SM) sites, and the Fourleague Bay (FLB-BM) site. Positive flux is import to the marsh surface; negative flux is export to the inundating water column. **=flux significantly different from 0 ($p \leq 0.05$); ***=flux highly significantly different from 0 ($p \leq 0.01$).

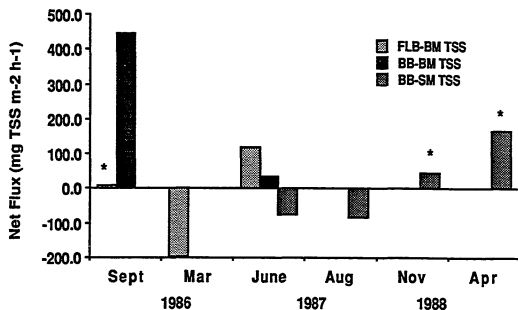


Figure 3-4: Fluxes of total suspended sediments (=TSS) at the Barataria Basin brackish (BB-BM) and saline (BB-SM) sites, and the Fourleague Bay (FLB-BM) site. Positive flux is import to the marsh surface; negative flux is export to the inundating water column. **=flux significantly different from 0 ($p \leq 0.05$); ***=flux highly significantly different from 0 ($p \leq 0.01$).

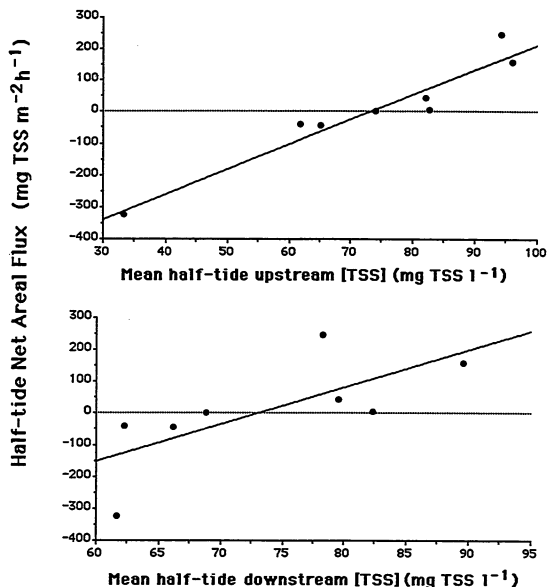


Figure 3-5: Relationships between flood and ebb tide TSS flux data and the associated mean (a) upstream [source] TSS concentrations ($r^2=0.96$, $p<0.001$) and (b) downstream TSS concentrations ($r^2=0.72$, $p=0.04$) at the BB-SM site.

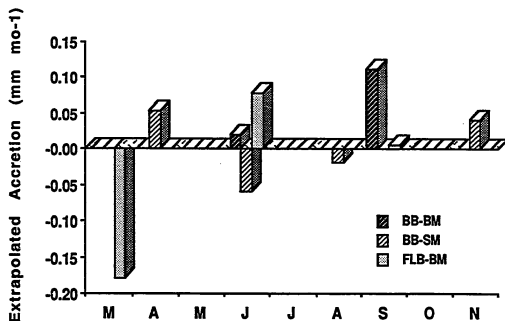


Figure 3-6: Monthly accretion of sediment (in mm) calculated from TSS flux data at the three BB and FLB sites. Accretion depths have been extrapolated to the entire month indicated by that bar, based on the actual % of time that month in which the marsh was flooded (Figure 2).

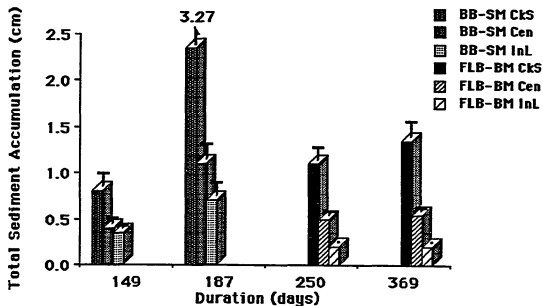


Figure 3-7: Actual measured sediment accretion at the BB-SM and FLB-BM marsh flume sites, as total sediment accumulation over feldspar marker horizons over the time span indicated (mean \pm 1 SD). For each, results from three locations along the marsh flume transect normal to the creek edge are presented: CkS=creekside, in the marsh adjacent to the creek; InL=30 m into the marsh from the creek, at the back of the flume, and; Cen=15 m into the marsh, between the CkS and InL locations.

Table 3-1: A review of net annual particulate flux data from recent flume studies. Positive values are uptake by the marsh while negative values are marsh export, and all values are in $\text{g m}^{-2} \text{yr}^{-1}$ for carbon (C) or nitrogen (N). **Bold type**=significant ($p<0.05$) flux; **Bold italics type**=highly significant ($p<0.01$) flux. Marsh type abbreviations: SM=saltmarsh. See text for nutrient abbreviations.

<u>Citation</u>	<u>Type</u>	<u>POC</u>	<u>PON</u>	<u>TSS</u>
Whiting et al.,1989	SM		3.3	
Wolaver & Spurrier,1988	SM	83.3		
Wolaver et al.,1988	SM			1022.0
Chalmers et al., 1988	SM	262.8		

Table 3-3: Standing crop biomass data for each sampling event, in gdw m^{-2} . In each case the full site data are the means and standard deviations of samples taken in the creekside marsh ($\approx 1\text{-}3\text{ m}$ from bayou, CkS), the inland marsh ($\approx 30\text{ m}$ from bayou, InL), and between the two (Cen).

<u>Site & Date</u>	<u>Biomass Type</u>	<u>Replicates Data</u>			<u>Full Site Mean</u>	<u>Data SD</u>
		<u>CkS</u>	<u>Cen</u>	<u>InL</u>		
A. Barataria Basin Brackish Marsh (BB-BM):						
9/18/86	total	1873.6	2870.0	1882.4	2208.7	572.6
6/23/87	live	1389.2	674.0	1263.2	1108.8	381.8
	dead	798.8	411.6	832.8	681.1	234.0
	total	2188.0	1085.6	2096.0	1789.9	611.7
B. Barataria Basin Saltmarsh (BB-SM):						
6/2/87	live	1334.0	1564.0	1152.8	1350.3	206.1
	dead	906.8	780.8	1066.8	918.1	143.3
	total	2240.8	2344.8	2219.6	2268.4	67.0
8/10/87	live	732.4	597.2	817.6	715.7	111.1
	dead	470.8	616.8	717.2	601.6	123.9
	total	1203.2	1214.0	1534.8	1317.3	188.4
11/16/87	live	429.2	529.6	548.0	502.3	63.9
	dead	454.8	769.6	1117.2	780.5	331.3
	total	884.0	1299.2	1665.2	1282.8	390.9
4/18/88	live	903.6	390.8	366.4	553.6	303.4
	dead	959.2	656.4	650.8	755.5	176.5
	total	1862.8	1047.2	1017.2	1309.1	479.8
C. Fourleague Bay Brackish Marsh (FLB-BM):						
3/8/87	total	2096.4	2314.0	1639.2	2016.5	344.4
6/10/87	live	1470.0	826.0	1122.8	1139.6	322.3
	dead	288.8	675.2	414.0	459.3	197.1
	total	1758.8	1501.2	1536.8	1598.9	139.6

Table 3-4: Sedimentological characteristics from creekside (CkS, $\approx 2\text{--}3$ m from creekbank) and inland (InL, ≈ 30 m from creekbank) locations at each flume site. Bulk densities are in g cm^{-3} and organic matter is in percent of total. All values are means ± 1 standard deviation of triplicate cores, from the upper 5 cm of each core. Whole site values are means of data from both CkS and InL locations. See text for definitions of site abbreviations.

<u>Site</u>	<u>Bulk Density Data</u>			<u>% Organic Matter Content</u>		
	<u>InL</u>	<u>CkS</u>	<u>Site</u>	<u>InL</u>	<u>CkS</u>	<u>Site</u>
BB-BM	0.29 \pm 0.04	0.42 \pm 0.09	0.36 \pm 0.09	26.0 \pm 4.9	17.4 \pm 2.6	21.7 \pm 5.9
BB-SM	0.34 \pm 0.00	0.38 \pm 0.07	0.36 \pm 0.05	21.0 \pm 2.3	19.6 \pm 2.1	20.1 \pm 2.0
FLB-BM	0.42 \pm 0.13	0.41 \pm 0.10	0.42 \pm 0.10	17.2 \pm 3.4	17.2 \pm 6.7	17.2 \pm 4.8

Chapter 4

Ecological Consequences of El Niño-Southern Oscillation Events in Mississippi Deltaic Plain Estuaries

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ABSTRACT

Between summer, 1987 and winter, 1988, coastal water levels and marsh inundation regimes in the Mississippi River Deltaic Plain (MRDP) were anomalously low, resulting in a dramatic decrease in the marsh:open estuary interactions mediated by intertidal flooding. This temporary phenomenon, which we refer to as "ephemeral wetland loss", resulted in a loss of the functional value of wetlands to the estuarine ecosystem when marshes were effectively isolated from the associated water column. This period of "ephemeral wetland loss" corresponded with the 1987-88 El Niño-Southern Oscillation (ENSO) event, and we hypothesized that the low estuarine water levels were a local manifestation of large-scale climatological forcing related to the ENSO event. To test this, and to determine if past ENSO events had effects on MRDP estuaries, we analyzed 28 years (1961-1988) of hydrologic, climatological, and biological data and related commercially-valuable estuarine productivity to ENSO-induced variability in marsh flooding regimes.

Records of continuous water levels at brackish and saline marsh sites in the Barataria Basin estuary, LA from 1987-89 showed this extended period of low marsh inundation. We used a 25-year water level record (1963-88) from a nearby gauge to ascertain the historical frequency of this type of phenomenon. Monthly means from 1987-88 were correlated with our site-specific marsh flooding data, indicating that the historical water level data adequately represented historical marsh inundation. Multivariate analysis indicated that wind direction, local precipitation and temperature, and Mississippi River discharge together significantly explained 37% of the intermonthly variability and 46% of the interannual variability in this 26-year coastal water level record, hence in historical marsh inundation. Using inshore shrimp harvest (1961-1988) to represent commercially-valuable estuarine productivity, we found that coastal water levels explained a significant portion ($r^2=0.26$) of the observed interannual variability in shrimp catch (per unit effort). Interestingly, the relationship between water levels [marsh inundation] and shrimp harvest

was not linear. Shrimp catches were low at both low and high coastal water levels while highest shrimp productivity was seen in years with intermediate water levels. In most cases, low shrimp catches coincided with ENSO events. The 1987-88 ENSO event, known as a "La Niña", coincided with anomalously low marsh inundation regimes and the ensuing phenomenon of "ephemeral wetland loss" probably negatively affected shrimp harvests by denying juveniles vertical access to intertidal habitat. When ENSO events coincided with anomalously high marsh inundation regimes, high water levels were associated with large local and remote inputs of fresh water to estuaries. These fresh water inputs probably decreased juvenile shrimp habitat horizontally, as zones of higher salinity waters were spatially reduced, and shrimp harvests declined. Climatological forcing thus had direct effects on estuarine flooding regimes and inshore shrimp harvests in coastal Louisiana, and the nonlinear relationship between shrimp catch and estuarine marsh inundation regimes is largely a result of low-catch endpoints during strong ENSO events.

INTRODUCTION

Apparent randomness in biological patterns is often related to physical forcing by one or more environmental parameters (Legendre and Demers, 1984). In estuarine systems, coastal water levels and associated wetland inundation regimes together form an important hydrologic control on ecosystem processes (Odum, 1980, many others). This is particularly true in flat, expansive, microtidal estuaries, such as those of the Mississippi River Deltaic Plain (MRDP). Water level patterns in these estuaries are strongly influenced by various climatological parameters on both short (days to a week) and intermediate (months to a year) time scales (Conner and Day, 1987; Turner, 1987).

Between summer 1987 and winter 1988, coastal water levels and marsh inundation regimes in Louisiana estuaries were anomalously low. The result was a dramatic decrease in the marsh:open estuary interactions mediated by intertidal flooding. This "ephemeral wetland loss" phenomenon actually represented a period of functional loss as marshes were effectively isolated from the associated water column, although they were not permanently lost. To document this unusual event, we examined which environmental factors were responsible for the low water levels and what effect this "ephemeral" wetland loss had on the estuarine ecosystem. We hypothesized that this situation, which coincided with the 1988 North American drought, was a local manifestation of large-scale climatological forcing related to the 1987-88 El Niño-Southern Oscillation (ENSO), or "La Niña", event. If such global climatological events have direct, forced effects on water levels and flooding regimes in Louisiana estuaries, then the effects should be seen in measures of estuarine productivity, such as landings from a marsh-dependent fishery. To test this, we analyzed 28 years (1961-1988) of hydrologic and climatological data from the MRDP and related historical inshore shrimp harvest, as a measure of commercially-important estuarine productivity, to weather-induced variability in marsh flooding regimes.

Ecological Importance of Marsh Inundation

There are important interactions between intertidal marshes and associated estuarine waters that are driven by natural variability in water levels. A number of recent studies have directly quantified exchanges of nutrients and materials between the water column and saltmarshes (Lee, 1979; Wolaver et al., 1980; Wolaver and Zieman, 1983a,b; Chalmers et al., 1985; Chrzanowski and Spurrier, 1987; Wolaver and Spurrier, 1988a,b; Whiting et al., 1989; Childers, 1989) as well as tidal freshwater marshes (Bowden, 1986; McIvor and Odum, 1986). While these studies generated a wide range of results regarding the magnitude and direction of net exchanges of carbon, nitrogen, phosphorus, sediments, and biomass, all concluded that the marsh:water column exchange processes are important components of estuarine nutrient dynamics and energetics. When low water levels reduce marsh inundation, these interactions are curtailed. If anomalously low water levels persist for long periods of time, the isolation of marsh from open water—"ephemeral wetland loss"—may have a pronounced effect on the estuarine ecosystem.

The intertidal marsh is also an important habitat for estuarine fish and invertebrates, many of which form the basis for commercially exploitable fisheries. Boesch and Turner (1984) found a high correlation between inshore shrimp yields and area of vegetated marsh in estuaries of the northern Gulf of Mexico. In a Texas saltmarsh, densities of juvenile brown shrimp (*Penaeus aztecus*) in non-winter months were significantly greater in *Spartina alterniflora* marshes relative to adjacent tidal creeks (Zimmerman and Minello, 1984). The vegetated wetland serves as both a refuge against predators and a source of nutrition to these animals (Boesch and Turner, 1984) many others. Tidal freshwater marsh fish, for example, preferentially utilize the shallow habitats which fringe marsh edges at low tide and quickly seek shelter on the marsh surface as water levels increase on the flooding tide (McIvor and Odum, 1988).

Zimmerman and Minello (1984) noted a close relationship between brown shrimp densities in Texas saltmarshes and seasons with high water levels. They hypothesized that

exploitation of vegetated habitat may improve crucial estuarine nursery functions and that the timing of higher water levels may determine the ability of brown shrimp to use a marsh. Clearly, an anomalously low flooding regime will prevent juvenile and estuarine fish and invertebrates from using the marsh habitat for refuge and feeding. If the condition persists through months crucial to a species' juvenile development, the result of this "ephemeral" loss of marsh habitat may well be seen in a reduced year class of organisms. Postlarval brown shrimp begin migrating into Louisiana estuaries in late winter, and remain until early summer. White shrimp (*Penaeus setiferus*) follow a similar life cycle but reside in estuaries from early summer through mid-fall (Thompson and Forman, 1987). Thus, non-winter marsh flooding regimes are most critical to the commercially important shrimp species of MRDP estuaries.

Coastal Water Levels and MRDP Marsh Inundation Regimes

Estuarine water levels vary hierarchically over a number of time scales, with periods of hours to decades. The importance of longer term sealevel variability to inundation-moderated interactions between marshes and estuaries has been recognized for some time (Kjerfve et al., 1978). In this century, coastal water levels along the northern Gulf of Mexico have shown a 20-year long-term cyclic pattern around a long term apparent sealevel rise (ASLR) of approximately 1 cm yr⁻¹. ASLR from 1942 to 1962 was greater than from 1962 to 1982, for example (Turner, 1987). Average monthly water levels on the Louisiana coast vary seasonally by approximately 25 cm and mean marsh flooding regimes closely follow this seasonality (Baumann, 1987). Lowest water levels and minimal inundation occur in the winter, when coastal saltmarshes are flooded an average of 25-40% of the time, and maximal water levels and inundation occur in early summer and early fall, when saltmarshes are flooded up to 80% of the time. A secondary low seen in midsummer and a secondary high in the spring complete the characteristic bimodal annual pattern observed along the Louisiana coast (Baumann, 1987). Astronomical tides control hourly and daily water level fluctuations near the Louisiana coast, with wind forcing in the flat,

microtidal MRDP estuaries increasing in importance with distance from the coast (Wax, 1977).

Climatological Forcing and MRDP Marsh Inundation Regimes

Climatological forcing and impacts on estuarine and marsh biological systems can often be related to large-scale atmospheric phenomena occurring either locally or far away from the coastal environments. In coastal Louisiana, local precipitation and runoff significantly impact estuarine water levels (hence marsh inundation regimes) and salinities because many estuaries are at least partially enclosed by flood control levees and direct runoff to the Gulf is delayed (Baumann, 1987). In addition, water levels and salinities are also impacted by runoff from river basins draining into the coastal wetlands and especially by the discharge of the Mississippi River system--representing runoff from 41% of the contiguous United States (Muller, 1975; Baumann, 1987). Hence, excessive runoff from a massive mid-latitude cyclone over the upper Midwest can affect water levels, salinities, and water temperatures in MRDP wetlands (Meade and Emery, 1971). In fact, much of the bimodal seasonality in water levels on the Louisiana coast is related to remote climatological forcing. The spring peak in water levels is partly related to Mississippi River discharge to the coastal zone (Meade and Emery, 1971), while the fall peak is more related to steric effects (Whitaker, 1971; Sturges and Blaha, 1976; Baumann, 1987) as well as the return of water, stored during colder months as continental soil moisture, to rivers and coastal oceans (Mather, 1974).

Astronomical tidal ranges on the Gulf coast average only 32 cm (Baumann, 1980) and wind systems have a strong effect on estuarine water levels in Louisiana (Baumann, 1987). Louisiana estuaries are expansive and shallow, and long fetches coupled with minimal topographic resistance further increase the importance of wind forcing. These estuaries are generally oriented along roughly north-south axes; northerly winds tend to lower estuarine water levels and marsh flooding regimes while southerly winds will "push" Gulf waters into estuaries and raise water levels (Baumann, 1980). Weather patterns

associated with regional wind systems over southeastern Louisiana have been organized into synoptic weather types (Muller, 1977) and summarized in calendars beginning in 1961 (Muller and Willis, 1983). Some of these weather types have been grouped into Continental Index (CI) and Tropical Index (TI) weather, representing regional generalizations of the weather dominated by northerly and southerly winds, respectively, that have been related to rising and falling estuarine water levels (Wax, 1977).

El Niño-Southern Oscillation Events

One of the largest interannual fluctuations in the global ocean-atmosphere climatological system is the warming of the eastern south Pacific ocean, known as the El Niño, and closely related reversal of South Pacific atmospheric pressure centers--the Southern Oscillation (U.S. Climate Research Committee, 1983; many others). These closely coupled events, referred to as ENSO events, are natural oscillations in the ocean-atmosphere system that influence weather patterns over much of the globe (Graham and White, 1988; Hamilton, 1988). ENSO events occur with a long-term irregular frequency of 2-7 years over the last 450 years (Quinn et al, 1987), but have occurred at a more regular 5 year interval in the last 40-50 years (Rasmussen and Carpenter, 1982; Graham and White, 1988). Since 1961, there have been two moderate ENSO events, in 1965 and 1976, a strong event in 1972-73, a very strong event in 1982-83, and a recent strong event in 1987-88 (Quinn et al., 1978; Mysak, 1986; Barnett et al., 1988; Cheney and Miller, 1988). Mysak (1986) and Enfield (1987) both present good reviews of the onset and progression of an ENSO event.

In the northern hemisphere, most extratropical responses to ENSO events are driven by increased teleconnections between atmospheric pressure centers (Yarnal, 1985; Hamilton, 1988). The Pacific-North American teleconnection pattern (PNA), which links the central Pacific High, the Aleutian Low, the Canadian High, and a persistent low over the southeastern U.S., is the important link between tropical Pacific pressure center oscillations and U.S. Gulf Coast weather patterns (Yarnal, 1985). Most ENSO-related

variability in northern hemispheric climatological conditions occurs in the winter half of the year, when pressure differentials and teleconnections are strongest (Horel and Wallace, 1981; Hamilton, 1988), and there is generally a 1-2 season lag between the fall onset of a typical ENSO event and the wintertime climatological response in the continental U.S. (Horel and Wallace, 1981).

Strong ENSO events generate several important climatological anomalies in southeastern U.S. and Gulf Coast weather patterns, including: 1) southerly displacement of the winter jet stream, triggering unusually stormy weather (Chen, 1983); 2) strengthening of low pressure over the S.E. U.S. and of the geostrophic pressure gradient [teleconnection] between this low and the Canadian High (Horel and Wallace, 1981), and; 3) increased cloud cover (Angell and Korshover, 1987). In the MRDP, these anomalies usually cause increased local and remote precipitation (and higher runoff and river discharge, respectively), enhancement in the northerly-southerly components of local winds, and lower than average temperatures, all of which may affect estuarine water level and flooding regimes. In fact, the very strong ENSO event of 1982-83 triggered massive rains in south Louisiana, extensive local flooding, greater than average discharge from the Mississippi River, and record discharges from several local rivers (Muller and Faiers, 1984).

The 1987-88 ENSO event began in mid-1986, reached strength by November-December, 1986, and remained strong for about 17 months (Barnett et al., 1988; Cheney and Miller, 1988). This event was peculiar, however, because of what happened next: In June, 1988, the warming of the eastern tropical Pacific (the El Niño) quickly reversed and waters became anomalously cold (Kerr, 1988). This pattern reversal, recently called a "La Niña" event, also reversed the climatological response in the continental U.S., and very dry conditions developed throughout the S.E. U.S. (Kerr, 1988). The 1988 drought appears to be a Pacific-North American teleconnection response to this negative sea surface temperature anomaly in the Tropical Pacific (Trenberth et al., 1988). It was during this

peculiar ENSO event that we identified persistently low water levels in Louisiana estuaries which resulted in anomalously low marsh inundation regimes. We hypothesized that most ENSO events may thus increase MRDP marsh flooding through high precipitation while unusual ENSO events (La Niña) which generate dry conditions may decrease marsh flooding. Using historical data from southeastern Louisiana for coastal water levels, Mississippi River discharge, and local winds, precipitation, and temperatures, we investigated the coincidence of unusual patterns in estuarine flooding and ENSO events, and the effect these concomitant occurrences have on inshore shrimp harvests.

MATERIALS AND METHODS

Water level patterns and the occurrence of marsh inundation were measured at a coastal saltmarsh site (SM) and a brackish marsh site (BM) in the Barataria Basin estuary, LA, using Richards-type water level recorders (Weathertronics, Inc. Model 6510) installed at each site in April-May, 1987 (Figure 1 and Conner and Day, 1987). These sites are located approximately 10 and 40 km from the Gulf of Mexico, respectively. Surveys of the marsh microtopography at each site, relative to the gauge datum, allowed us to determine water height at bankfull stage. From the continuous water level records, we calculated hours of marsh inundation as a percent of total hours each month.

Coastal water level records were obtained from the U.S. Army Corps of Engineers (U.S.A.C.E.) Bayou Rigaud Gauging Station, on Grand Isle, LA. This coastal gauge is approximately 15 km from the saltmarsh site described above. Hourly data from January, 1963 through May, 1988, were averaged by month to smooth astronomical tidal variability. Apparent sealevel rise was removed from this water level record by regressing the 25-year monthly mean time series against time ($r^2=0.35$, $p<0.01$) and filtering the resultant slope from the data set. Annual mean water levels are averages of monthly means. Both monthly and annual means were compared to the overall 25-year mean to determine positive and negative anomalies. We correlated these water level data to our site-specific

records of marsh flooding in the Barataria Basin estuary, LA, and looked for past recurrences of the anomalously low flooding regime observed in 1987-8.

A number of climatological parameters from the same time period were also analyzed. Annual mean temperatures and annual total precipitation were calculated for 1961-88 from monthly means of data taken at the New Orleans Moisant Airport (Louisiana Office of State Climatology, Baton Rouge, LA), which is near the northern boundary of the Barataria Basin estuary. Both monthly and annual data were used to lessen error associated with unknown lags in climatologically-forced environmental responses. We analyzed wind data using synoptic climatological indices that categorize weather parameters by similarity of conditions (Muller, 1977). The weather indices important to the analysis are those dominated by winds blowing parallel to the estuarine axis, as this wind stress is most important to water level variability. The Continental Index (CI) represents weather types where northerly winds dominate and the Tropical Index (TI) where southerly winds predominate (Wax and Muller, 1977). CI weather forces water from the estuary, lowering water levels, while TI weather forces water into the estuary (Wax, 1977; Baumann, 1987). These data are presented as the percent of total recorded hours per month when each synoptic index type was observed each month (Louisiana Office of State Climatology). We calculated annual means from these monthly percentages. Annual mean discharge of the Mississippi River was calculated from monthly averaged discharge data for 1961-1988 at the gauging station at Tarbert Landing, MS (U.S.A.C.E.). This station was chosen because it is upstream of the Louisiana coastal zone and therefore represents river discharge independent of the effects of local precipitation and runoff.

We investigated the relative importance of these various environmental parameters to observed coastal water levels with multivariate analysis, including stepwise and multiple regression techniques (Statview 512+® Statistical Package). This multivariate analysis emphasized determining which environmental parameters exhibited the most influence on coastal water level patterns on both monthly and annual time frames. Data for March

through October were used for the monthly analyses, as this is when juvenile shrimp utilize marsh habitats in MRDP estuaries (Thompson and Forman, 1987; others). Bayou Rigaud water level (WL) was the dependent variable, and annual precipitation (P), mean temperature (T), mean Tropical Index (TI), mean Continental Index (CI), and Mississippi River discharge (Miss. R. Q) were the independent variables.

Catch data from the Louisiana shrimp fishery, including pounds landed inshore and effort expended, were obtained from the National Marine Fisheries Service (New Orleans, LA) for 1962-1988. The inshore shrimp fishery (primarily brown, *Penaeus aztecus*, and white shrimp, *Penaeus setiferus*) was chosen as the indicator of commercial estuarine productivity for a number of reasons: 1) Historical data were both reliable and available; 2) shrimp utilize the marsh surface and thus depend on access via marsh inundation; 3) it was possible to look only at estuarine catch, reducing variability related to offshore population dynamics, and 4) shrimp grow quickly, reducing complications related to year classes and long lags in population responses to environmental forcing. We converted inshore shrimp catch values to catch per unit effort (CPUE) by dividing landings by effort (as kg trip⁻¹). Louisiana shrimp catch showed a highly significant increase ($r^2=0.59$, $p<0.001$) over this time period. To be consistent in our treatment of time series data, we filtered this temporal increase in shrimp CPUE from the data by computing the residual CPUE values from this regression and identifying them as normalized CPUE.

As with the Bayou Rigaud water level data set, we subjected shrimp CPUE and normalized CPUE (dependent variables) to multivariate analysis using Bayou Rigaud WL, P, T, CI, TI, and Miss. R. Q as independent variables. In this step, shrimp harvest data were considered to be an indicator of commercially important estuarine productivity. Both normalized and uncorrected CPUE data were offset one year earlier, to simulate a potential 1-year lag in population response to flooding regime variability. We also analyzed this lagged data set with the multivariate techniques discussed above.

RESULTS AND DISCUSSION

The inundation regimes between May, 1987 and April, 1989 at the brackish and saline marsh sites are shown in Figure 2 (BM=a, SM=b). For comparison, a 5-year mean flooding regime for Barataria Basin saltmarshes, from 1975-9 (Baumann, 1980), is superimposed over our data in Figure 2b--there is no published average of brackish marsh inundation for comparison with Figure 2a (C.Sasser, Louisiana State Univ., pers. comm.). Several important features are evident in Figure 2. First, flooding regimes at both sites declined dramatically in July, 1987 and remained low through late 1988. At the brackish marsh site, the marsh was inundated less than 10% of the time from July, 1987 through August, 1988 (Figure 2a). In fact, between mid-December, 1987 and mid-March, 1988, this site failed to flood for 92 consecutive days.

Barataria Basin saltmarshes typically experience 160 tidal inundations per year (Madden et al., 1988) compared to over 700 in a mesotidal and semidiurnal southeastern U.S. saltmarsh. The marsh at the SM site flooded 123 times between May and December of 1987, 117 times in 1988 (12 months), and 30 times between January and March of 1987. Reductions in total inundation events, from 1987 to 1988 and 1989, were 26% and 15% respectively. In ecological terms, however, the duration of flooding may be more critical. Comparison of 1987-9 saltmarsh flooding frequencies with the 5-year mean (Figure 2b; Baumann, 1980) indicated that monthly flooding durations were reduced 32-92% at the SM site. The bimodal seasonal pattern typical of the monthly mean data was nearly completely attenuated, and did not return until late 1988-early 1989. At both sites, the 1987-88 marsh inundation regime was anomalously low and the temporal pattern in flooding was markedly attenuated for an extended period of time.

Annual means of hourly water level readings at the Bayou Rigaud gauging station on Grand Isle, LA, provide a 25-year picture (January, 1963- May, 1988) of Gulf coast water levels in the Barataria Basin estuary (U.S.A.C.E.). These data showed an expected apparent sealevel rise over the time period analyzed. The regression of Bayou Rigaud

mean annual water height against time ($r^2=0.35$, $p<0.01$) generated an ASLR of 1.40 cm yr⁻¹. The temporal pattern with this increase filtered out is shown in Figure 3a. The 25-year mean water level, 128.6 cm, is also shown. The low water event in 1987-88 is clearly observable in Figure 3a. A multiple-year period of high water levels, between 1972 and 1975, is also seen.

The seasonal variability in water levels at Bayou Rigaud is seen in the monthly means of the hourly data (1980-88; Figure 3b). The monthly % flooding values at the SM site (Figure 2b) were significantly correlated ($p=0.04$) with mean water levels at Bayou Rigaud for the same months, confirming that the historical Bayou Rigaud water level record adequately represents marsh inundation. The upper panel of Figure 3b is mean monthly water levels through the 25-year record. The average curve shows a bimodal seasonal pattern. Notably, this pattern was markedly attenuated in the actual data from mid-1982 to mid-1983 and for all of 1987-88, as seen in Figure 3b when the two curves are superimposed. In 1982-83, actual water levels were above the average and corresponded to a period of high precipitation in coastal Louisiana (Muller and Faiers, 1984). In 1987-88, though, water levels were well below predicted water levels as most of North America experienced drought conditions (Kerr, 1988; Trenberth et al., 1988; see shaded areas, Figure 3b). These two time periods also corresponded with the very strong 1982-83 ENSO event and the recent 1987-88 La Niña event. In the next section, we explore these potential effects of climatological forcing on estuarine marsh flooding regimes by relating weather and river discharge parameters to water levels.

MRDP Marsh Inundation Regimes and Climatological Effects

The climatological parameters described above, each contribute to the intermonthly and interannual variability observed in water levels on the Louisiana coast. Using Bayou Rigaud WL as a surrogate for the marsh flooding regime, we used multivariate analytical techniques to relate P (X_1), T (X_2), CI (X_3) and TI (X_4) weather conditions, and Miss. R. Q (X_5) to coastal water levels (Y). These five climatological indicators explained 36% of

the variability in March through October monthly mean coastal water levels at Bayou Rigaud ($p < 0.001$; Table 1). Temperature had the strongest explanatory capability, as expected with the seasonal pattern of high summer and low winter water levels (Figure 3b). Southerly winds (TI) and precipitation also contributed significantly ($p \leq 0.05$) to the nonforced stepwise model. All three had positive relationships with Bayou Rigaud WL--with increased temperature, precipitation, and prevalence of southerly winds, coastal water levels increased (Table 1).

A significant ($p = 0.032$) multiple regression revealed that 46% of the variability in mean annual water levels at Bayou Rigaud is explained by P (X_1), T (X_2), CI (X_3) and TI (X_4) weather conditions, and Miss. R. Q (X_5 ; Table 2). In this analysis of annual data, precipitation (P) had the greatest effect on coastal water levels ($r^2 = 0.274$) and both wind indices explained a considerable percentage of the interannual variability in water levels ($r^2 = 0.16$), but Miss. R. Q and T had very little effect ($r^2 = 0.028$). Interestingly, the regression coefficient in the annual model ($r^2 = 0.462$; Table 2) was greater than in the monthly model ($r^2 = 0.38$; Table 1). This is probably due to the wide-ranging lags in water level response times to the different climatological parameters. Coastal water levels, for example, respond quickly to wind forcing (hours to a few days) and local precipitation (less than a month; Wax, 1977; Baumann, 1987) but there will be a longer lag between Mississippi River discharge measured above the coastal zone and water levels. In fact, while river discharge has a positive effect on Louisiana Gulf coast water levels on an annual basis (Meade and Emery, 1971), the relationship seen in the monthly data model is negative (Table 1).

Very little interannual variability was seen in mean annual temperature (T), with a 25-year range of only about 2°C (Figure 4a). Annual precipitation (P) varies considerably, though, with high rainfall observed in 1961, 1966, 1973-75, and 1983 (Figure 4a). This high local precipitation had a positive effect on water levels in the Barataria Basin estuary (Figure 3a), because it is enclosed by levees on three sides (Conner and Day, 1987).

These high precipitation years also correspond to a moderate and 2 strong ENSO events, in 1965, 1972-73, and 1982-83 respectively, indicating that marsh flooding was enhanced during these 3 ENSO events. Low precipitation in 1987 and 1988 coincided with the low water level ["ephemeral wetland loss"] phenomenon we have documented here, and with the 1987-88 La Niña event.

The weather indices shown in Figure 4b aggregate daily weather by dominant wind direction, with the Continental Index (CI) corresponding to northerly winds and the Tropical Index (TI) to southerly winds (Muller, 1977). Higher than average occurrence of CI conditions and northerly winds leads to decreased water levels in the Barataria Basin estuary as water is forced south to the coast. The opposite happens when TI conditions prevail (Baumann, 1987). Prolonged periods of lower CI and higher TI conditions were recorded between 1970 and 1975 and, to a lesser degree, in 1984-86 (Figure 4b). The 1970-75 predominance by southerly winds corresponds to the 1972-73 ENSO event and to higher water levels (Figure 3a). The predominance of north winds (CI) observed in 1982-83 are generally associated with lower estuarine water levels (Wax, 1977; Baumann, 1987), yet Bayou Rigaud WL in 1983 was above the 25-year average (Figure 3a). Wind patterns during the 1987-88 La Niña event were not particularly anomalous (Figure 4b).

Mississippi River discharge peaked in 1961, 1973, 1979, and 1983 (Figure 4c). High river discharges in 1973 and 1983 coincided with high precipitation (P) and with the 1972-73 and 1982-83 ENSO events (Figure 4a). Apparently, these two events were associated with high precipitation rates both in south Louisiana and in the Mississippi River basin in general. The discharge of unusually large volumes of Mississippi River water to the Louisiana coastal zone has been shown to elevate coastal water levels (Meade and Emery, 1971; Baumann, 1987). In fact, the large peaks in Mississippi River output in 1973 and 1983 corresponded with high MRDP water levels (Figure 3a) and the strong 1972-73 and 1982-83 ENSO events. On the other hand, both coastal water levels and

Miss. R. Q were low in 1987 and 1988 during the La Niña event which has been related to the North American drought (Kerr, 1988; Trenberth et al., 1988).

MRDP Estuarine Inundation Regimes and the Louisiana Shrimp Fishery

The corrected inshore shrimp catch per unit effort (CPUE, in kg trip⁻¹) data showed a significant temporal increase between 1962 and 1988 ($r^2=0.594$, $p<0.001$, Figure 5a). The normalized shrimp CPUE, after filtering out this temporal increase by computing residual CPUE values from the CPUE vs. time regression described above, are plotted in Figure 5b. In both cases, several distinctive patterns can be seen. Inshore shrimp CPUE declined markedly in 1972 and remained low for 4 years. CPUE was also low in 1981, 1983-84, and has declined since 1985 (1986-88). In three of these cases, low harvests of shrimp coincided with ENSO (1972-75 and 1983-84) or La Niña (1987-88) events.

The results of multiple and unforced stepwise regression analysis of catch data and environmental parameters are presented in Table 2. The important dependent variables were CPUE, normalized CPUE, 1-year lag CPUE, and 1-year lag normalized CPUE. The 1-year lag CPUE accounted for the possibility that environmental conditions detrimental to estuarine shrimp populations may not have been manifested until the following yearclass of organisms. In this analysis, we related climatological (P, CI, TI, and T) and hydrologic (Bayou Rigaud WL, Miss. R. Q) parameters to interannual variability in these four measures of estuarine productivity. Although none of the multiple regressions were significant at the $\alpha=0.05$ level, environmental parameters were able to explain 34% of the variability in uncorrected shrimp harvest ($p=0.259$, Table 2). In this model, CPUE was dependent mainly on precipitation and the occurrence of northerly winds (which tend to lower estuarine water levels); there was a negative relationship between shrimp harvest and both variables. This is not surprising because: 1) estuarine shrimp populations have specific salinity requirements and show poor fisheries yields in years with heavy rainfall when estuarine habitat with higher salinity water is restricted (Barrett, 1975; Turner, 1979;

Boesch and Turner, 1984), and 2) juvenile shrimp actively seek food and refuge on estuarine marshes and thus require water levels high enough to flood these marshes (Boesch and Turner, 1984; Zimmerman and Minello, 1984).

Earlier, we hypothesized that climatological forcing affects estuarine productivity (as measured by commercial shrimp fishery output) primarily by its effect on estuarine flooding regimes. In fact, a second-order polynomial model of Bayou Rigaud mean annual water level (X) and normalized CPUE (Norm CPUE= Y) was significant ($r^2=0.259$, $p=0.043$; Table 3). Thus, 46% of the interannual variability in coastal water levels [and marsh inundation] was explained with five environmental variables (P , CI , TI , $Miss. R.$, Q , and T , Table 2) and 26% of the interannual variability in normalized inshore shrimp harvest data was explained with only coastal water levels, hence marsh flooding (Table 3).

Perhaps the most interesting result was the nonlinear relationship between water level and shrimp catch, shown in Figure 6. Shrimp catch per unit effort was highest when water levels were intermediate, and low when flooding regimes were both low--inhibiting access to vegetated marsh habitat--and high--when high inputs of fresh water decreased estuarine salinities (Ho and Barrett, 1977). As examples of the coincidence of ENSO events and lower shrimp harvests, we noted the position of years associated with a high precipitation ENSO event and the low precipitation La Niña event on Figure 6. The four low harvest points associated with the highest water levels are from 1972-75 and correspond to the 1972-73 ENSO event (Figure 6). While estuarine flooding regimes were high during this 4-year period, and access to the marsh was [apparently] not hydrologically hindered, local precipitation inputs to the Louisiana coast were also high (Figure 4a), probably lowering estuarine salinities (Wax, 1977). Mississippi River discharge was also very high (Figure 4c), and a predominance of southerly winds (TI) over northerly winds (CI , Figure 4b) probably facilitated movement of this fresh Mississippi River input from the nearshore coastal zone into the estuaries, further enhancing the freshening process (Ho and Barrett, 1977; Madden et al., 1988). The lowest combined shrimp harvest and water

level, seen in 1987, indicates that the "ephemeral wetland loss" documented here, caused by an extended period of anomalously low water levels during the peculiar 1987-88 La Niña event, had a detrimental effect on inshore shrimp production (Figure 6).

SUMMARY AND CONCLUSIONS

Relationships between environmental causes and ecological effects are complex. Often, the apparent randomness observed in nature is related to some interaction between biological systems and their environment. In estuarine ecosystems, hydrology is one of the strongest environmental forces acting on ecological processes. Water levels and currents vary dynamically on hourly time scales, driving exchanges between the open estuary and both intertidal wetlands and the nearshore coastal ocean. In the expansive, flat, and microtidal estuaries of coastal Louisiana, climatological parameters exert important controls on water levels and flooding regimes. The ecological implications of this forcing include effects on marsh:open water exchanges of nutrients and materials and intertidal habitat accessibility for organisms.

In this paper, we explored the relationship between climatological forcing, water levels on the Louisiana coast, estuarine inundation regimes, and productivity of a commercially important estuarine fishery. In 1987-88, we observed an extended period of reduced marsh inundation. This "ephemeral wetland loss" phenomenon was caused by (and significantly related to) anomalously low coastal water levels, and occurred during the 1987-88 La Niña event. Multivariate analysis of 25 years of historical data indicated that wind direction, local precipitation and temperature, and Mississippi River discharge together significantly explained 36% and 46% of the intermonthly and interannual variability in coastal water levels and marsh inundation, respectively. Further, coastal water levels alone explained a significant 26% of the observed interannual variability in normalized catch per unit effort data from the Louisiana inshore shrimp fishery. Interestingly, the relationship between water levels [marsh inundation] and shrimp harvest

was not linear: Shrimp catches were low at both low and high coastal water levels while highest shrimp productivity were seen in years with intermediate water levels.

Climatological forcing has direct effects on estuarine flooding regimes in coastal Louisiana, and subsequent effects on estuarine commercial production as measured by inshore shrimp fishery statistics. Thus, there is a direct link between environmental forcing on hydrologic variability and ecological processes dependent on marsh:open water interactions. In this paper, we report for the first time that large-scale ENSO events may have direct ecological effects on Gulf coast estuaries in the S.E. U.S. Most ENSO events cause higher than average precipitation and coincide with anomalously high water levels. High marsh inundation regimes in these years are associated with large local and remote inputs of fresh water to estuaries. These fresh water inputs probably decrease juvenile shrimp habitat horizontally, as zones of higher salinity waters are spatially reduced, and shrimp harvests decline. The 1987-88 La Niña event, however, caused drought conditions and anomalously low marsh inundation regimes. Shrimp harvests declined as juvenile shrimp were denied vertical access to intertidal habitat during the ensuing "ephemeral wetland loss" phenomenon. The nonlinear relationship between shrimp harvest and estuarine marsh inundation regimes is thus largely a result of low-catch endpoints caused by local effects of global ENSO events.

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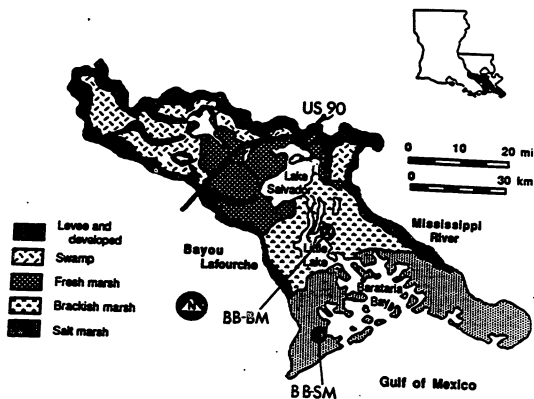


Figure 5-1: Site maps of the Barataria Basin estuary (BB), bounded on the east by the Mississippi River levees, on the west by the Bayou Lafourche levees, and on the north by U.S. Highway 90. Water level gauge sites are marked with an "X". Note that the estuary is oriented on a southeast-northwest axis from the Gulf of Mexico (modified from Conner and Day, 1987).

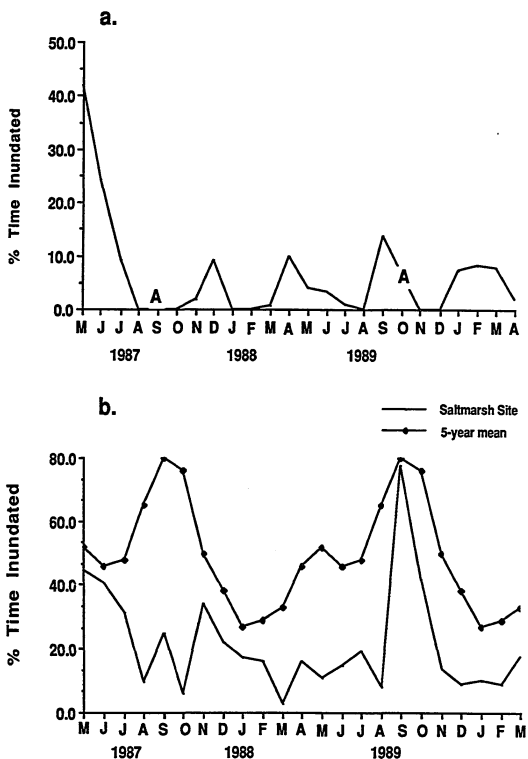


Figure 4-2: Flooding regimes measured at the brackish (a) and saltmarsh (b) sites of the Barataria Basin estuary. In (a), A=no data (gauge down). In (b) the 5-year average, 1975-9, is from Baumann (1980).

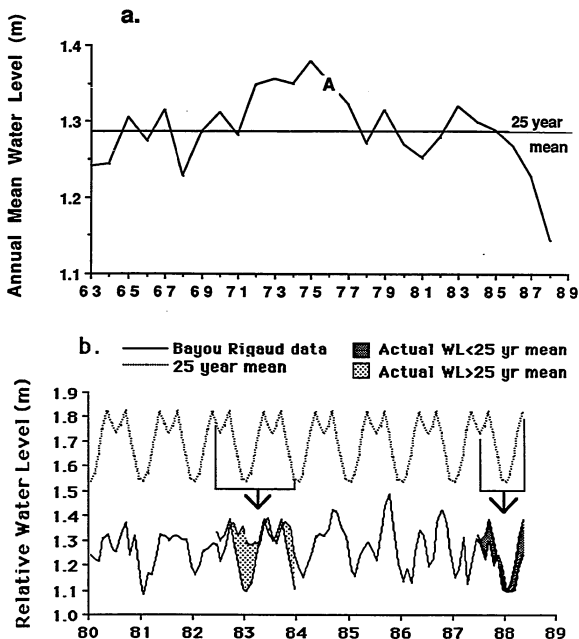


Figure 4-3: (a) Filtered annual mean water levels from the Bayou Rigaud coastal gauge on Grand Isle, LA, computed from hourly water level readings (U.S.A.C.E.). A=incomplete data set (only 3 months of data were available from 1976). (b) Bottom panel=filtered monthly mean water levels from the same gauge, computed from hourly water level readings (U.S.A.C.E.); top panel=25-year average monthly water levels. The 1982-83 and 1987-88 segments of the top panel are overlain onto the bottom panel, and light shaded areas indicate where actual coastal water levels were anomalously high while dark shaded areas indicate where actual coastal water levels were anomalously low (see text).

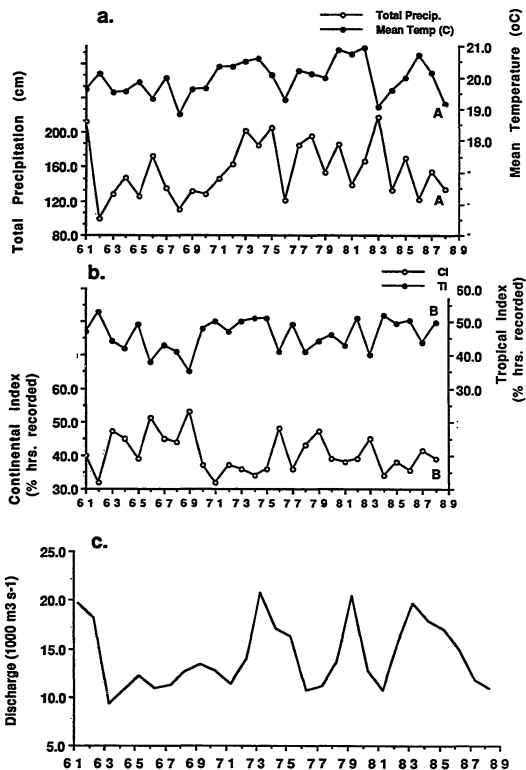


Figure 4-4: (a) Mean annual air temperatures on top curve and annual precipitation on bottom curve, from daily observations at the New Orleans Moisant Airport (L.O.S.C.). A=data through July, 1988 only. (b) Tropical Index weather conditions on top curve and Continental Index weather conditions on bottom curve, from synoptic indices summarizing weather observations at the New Orleans Moisant Airport characterized by southerly and northerly winds, respectively (Muller, 1977). B=data through October, 1988 only. (c) Mean annual Mississippi River discharge at Tarbert Landing, MS, based on daily readings (U.S.A.C.E.).

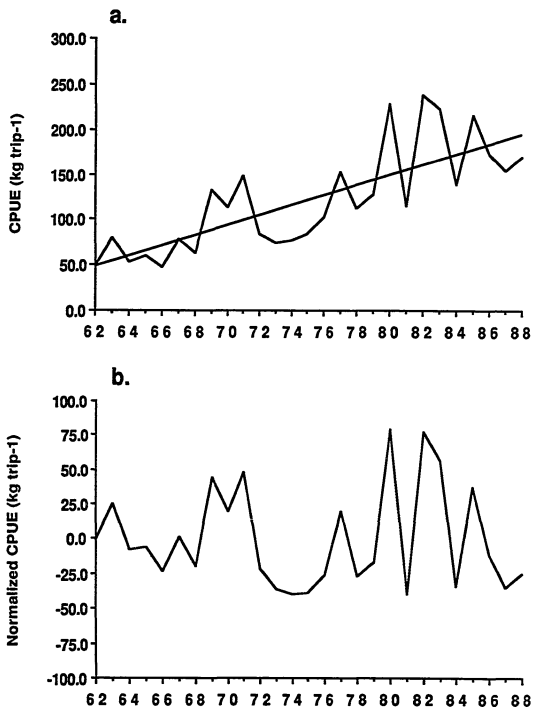


Figure 4-5: (a) Inshore shrimp landings for Louisiana, corrected for effort (N.M.F.S.). The significant temporal increase noted by the regression line is described in the text ($Y=5.56X-295.51$, $r^2=0.594$, $p<0.001$). (b) Normalized inshore shrimp catch per unit effort for Louisiana, as residual values from the regression shown in (a).

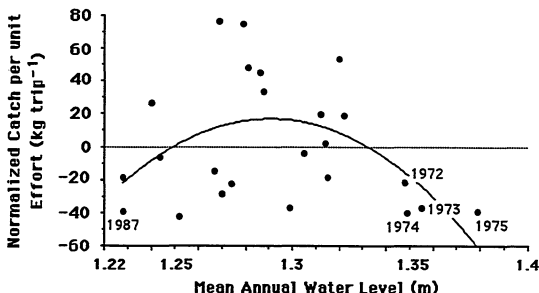


Figure 4-6: Relationship between normalized total inshore shrimp catch per unit effort (dependent variable) and mean annual coastal water level at the Bayou Rigaud gauging station (independent variable). Quadratic curve is the line predicted by the significant regression ($Y = -16549.6 + 25673.1X - 9946.4X^2$, $r^2 = 0.259$, $p = 0.043$). The years indicated are related to ENSO events and show unusual climatological conditions, described in the text. Summarized:
 1973=4th highest total precipitation in 28 year data set, highest river discharge in 24 year data set, % time under TI (southerly wind) conditions >28 year average in 7 of 12 months.
 1975=3rd highest total precipitation in 28 year data set, % time under TI (southerly wind) conditions >28 year average in 8 of 12 months.
 1987-8=low precipitation, low river discharge, lowest water levels.

Table 4-1: Contributions of independent variables (columns, X_j) to intermonthly variability observed in Bayou Rigaud water levels (X_i) using monthly means for March through October. Each X_{ij} position indicates the cumulative regression coefficient generated from multiple regression analysis of the model. "+"=positive slope, "-"=negative slope. Numbers in parentheses indicate significantly contributing variables as determined by nonforced stepwise regression techniques. Overall X_i model r^2 is shown in the right column; the model p-value was <0.001 ($n=180$). T=mean annual temperature; TI=Tropical Index weather; P=monthly precipitation; CI=Continental Index weather.

<u>Statistic</u>	Independent Variables				<u>Total</u>
	I	TI	P	CI	
r-square	0.241	0.318	0.357	0.365	0.365
slope	+	+	+	-	-
stepwise order	1	2	3		

Table 4-2: Contributions of independent variables (columns, X_j) to interannual variability observed in dependent variables (rows, X_i), as annual means. Each X_{ij} position indicates the cumulative regression coefficient generated from multiple regression analysis of the model shown in that row (X_i). Overall r^2 and p values for each X_i model are shown in the right column. BRWL=mean annual water level at Bayou Rigaud; P=annual precipitation; CI=Continental Index weather; TI=Tropical Index weather; MRQ=Mississippi River mean annual discharge; T=mean annual temperature.

Dependent Variable	Independent Variables						Total r^2, p
	BRWL	P	CI	TI	MRQ	T	
BRWL	NA	0.274	0.364	0.434	0.45	0.462	0.462 0.034
CPUE	0.012	0.148	0.226	0.262	0.291	0.336	0.336 0.259
Normalized CPUE	0.023	0.052	0.055	0.096	0.101	0.107	0.107 0.907
1-Year Lagged CPUE	0.039	0.04	0.084	0.09	0.207	0.256	0.256 0.471
1-Year Lagged Normalized CPUE	0.084	0.211	0.236	0.237	0.27	0.278	0.278 0.407

Table 4-3: Simple and complex (polynomial) regressions of Bayou Rigaud mean annual water level (BRWL; X) and annual shrimp harvest (Y). In each case, the both simple and second-order polynomial regressions were run and the more significant case is presented below, as r^2 and p values. Superscript "a"=simple linear model, superscript "b"=polynomial model. CPUE=catch per unit effort; Norm CPUE=normalized CPUE; Lag CPUE=1-year lagged CPUE; Lag N CPUE=1-year lagged normalized CPUE.

Statistical parameter	Dependent Variables			
	CPUE	Norm CPUE	Lag CPUE	Lag N CPUE
r-square	0.17 ^b	0.259 ^b	0.087 ^b	0.133 ^a
p-value	0.141	0.043	0.400	0.087

Chapter 5

Wetland Loss and the Estuarine Ecosystem: *An Added Dimension*

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ABSTRACT

Some degree of wetland loss characterizes most coastal systems of the United States. This loss is generally reported as a decrease in wetland area, but most of this coastal land loss entails wetland submergence and conversion to open water. This concurrent increase in the area of aquatic habitat decreases the marsh:water ratio, effectively diluting the area of remaining marsh relative to the aquatic system. The functional loss of intertidal wetlands to the ecosystem, caused by this dilution, represents an *added dimension* to wetland loss which may significantly alter ecological functions dependent on the interactive coupling of marsh and aquatic habitats. The magnitude of this functional loss is strongly dependent on the marsh:water ratio of a particular estuary. In estuaries with open bay-type morphologies, the open water area is already large and functional loss of marsh by additional dilution may only be slightly greater than the areal marsh loss. Where estuaries are wetland-dominated, however, conversion of even a small percentage of marsh to water drastically alters the marsh:water ratio. In these cases, functional losses by dilution are much greater than the rate of areal wetland loss.

Many important estuarine ecosystem processes involve marsh:water column exchanges. We are currently quantifying nutrient and materials exchanges between the marsh and associated water column in the Barataria Basin estuary, LA. Given a wetland loss rate for this estuary of 0.8% yr⁻¹, 15.4% of the remaining saltmarsh was lost between 1967 and 1987. By converting 15.4% of the marsh to aquatic habitat, the supply of inorganic nutrients and organic matter to the estuarine water column by the marsh (per unit area of open water) may have been reduced by over 27%—a 27% functional loss due to a 15.4 areal loss. Extrapolating this scenario another 20 years, there may be a 45% functional loss of saltmarsh, relative to 1967, by the year 2007 with a total areal loss of 28%. This dramatic difference between the areal loss of marsh and the functional loss due to habitat change is the what we refer to as the *added dimension* of wetland loss. This functional loss may have great significance where intertidal marshes supply energy to the

estuarine ecosystem--by exports of organic matter, nutrients, or reduced chemical compounds. It is conceivable that an estuary characterized by wetland loss may reach a point where, although some marsh remains, its functional value to the ecosystem is essentially gone.

Introduction

Wetland loss in the United States and the environmental, economic, and social consequences of this loss have received considerable attention. Most inland wetlands (marshes and swamps) have been lost by conversion to agricultural and urban uses. For example, over 40% of North Carolina's pocosin wetlands have been cleared, mostly for row crops (Richardson et al., 1981). Fewer than 30% of the original forested wetlands in the coterminous U.S. remain (Harris and Gosselink, 1986), and 80% of the Mississippi River alluvial floodplain forests have been cleared (MacDonald et al., 1979).

In coastal zones, fewer than 50% of the tidal wetlands intact in the late 1800's remain today (Gosselink and Baumann, 1980). Coastal wetland losses have been reported in nearly all regions of the U.S. containing substantial marsh acreage, and a number of mechanisms have been cited. Most net marsh loss in Delaware Bay appears to be related to shoreline erosion and marsh drowning (Phillips, 1986). Wetland deterioration occurring in North Carolina's estuaries is a function of both shoreline erosion and inlet dredging (Hackney and Cleary, 1987). On the eastern shore of the Chesapeake Bay, however, Stevenson et al. (1985) reported that much of the marsh erosion begins with the formation of inland ponds that enlarge by marginal slumping.

The wetland loss phenomenon is most pronounced in coastal Louisiana, which contains about 40% of all coastal wetlands in the contiguous U.S. but where over 80% of all marsh loss is occurring (Gagliano et al., 1981; Turner, 1982; Templet and Meyer-Arendt, 1989). Here, wetland loss has been related to a number of impacts, including canal dredging, subsidence, and fringe erosion (Cleveland et al., 1981; Baumann et al., 1984; Swensen and Turner, 1987; many others). Loss of both inland and coastal wetlands is [nearly always] reported as a simple areal loss of marsh habitat.

Many studies quantifying coastal marsh loss concluded that: 1) a sediment deficit was primarily responsible for the inability of wetlands to maintain themselves against

coastal submergence, and; 2) that the often-irreversible marsh loss nearly always represents a conversion of intertidal vegetated habitat to open water habitat. If relative sealevel rise continues at predicted rates of up to 17 cm by the year 2000 (Hoffman, 1984), marshes deprived of sufficient supplies of allochthonous sediment will continue to become open estuarine water bodies. Some regions are still receiving sediment loads considered adequate for maintenance against sealevel rise--the South Carolina coast, for example (Gardner and Kitchens, 1978; Wolaver et al., 1988). These systems will be maintained by vertical accretion coupled with lateral [inland] migration. Upland geomorphological features may limit inland migration of these transgressive marsh systems, though. Glacially formed estuaries of the west coast and New England not threatened by erosion or sediment starvation may well lose marshes to this limitation of shoreline recession as well. In Louisiana, most loss of coastal marsh is by conversion to open water, although considerable areas of wetland are functionally lost by impoundment or filling alterations (Craig et al., 1979). The estuarine ecosystem is affected by the areal loss of wetlands, but also by the concurrent increase in open water. This additional open water habitat effectively dilutes the influence of the remaining marsh on the estuary.

Dilution and Loss of Wetland Function

Loss of a wetland represents a removal of its functional value to the associated aquatic ecosystem. Turner (1982), for example, showed a correlation between decreased marsh area and decreased fisheries harvest in Louisiana; he related these correlative declines to a loss of the functional value of the marshes to the estuary. This type of temporal areal loss (from time= T_1 to T_2) is generally considered with the marsh as the frame of reference, and with the area at T_2 (MA_2 ; usually the current area) expressed as its fraction of the area at T_1 (MA_1):

$$1) \quad \left(1 - \frac{MA_2}{MA_1} \right) * 100 \quad (\text{in } \%)$$

Most estuarine wetlands are actually lost by conversion to open water, representing areal change of both marsh [loss] and open water [gain]. In this conversion, the loss of ecological function combines the areal marsh loss and the dilution of remaining marsh by the increased open water area. The ecological functions of intertidal marshes include providing refuge for aquatic organisms and habitat for migratory waterfowl (Boesch and Turner, 1984; Zimmerman and Minello, 1984; many others), and producing and transforming chemical energy (Nixon, 1980; many others). The tidal exchange of materials is one of the most important functional contributions of wetlands to the estuarine ecosystem. Inorganic nutrients are taken up by some marshes while others export them to the estuarine water column, and the same is true of organic matter (Nixon, 1980). An increase in aquatic habitat dilutes this functional importance of the remaining marsh, thus significantly increasing the dimensions of wetland loss. As decreased marsh area interacts with increased estuarine water volume, the loss of wetland function is greater than expected if only decreased area is considered. The functional loss relates marsh areas (MA_1 and MA_2) to open water areas (WA_1 and WA_2) at the respective times (T_1 and T_2):

$$2) \quad \left[1 - \left(\frac{MA_2}{WA_2} \right) * \left(\frac{MA_1}{WA_1} \right)^{-1} \right] * 100 \quad (\text{in } \%)$$

For example: A hypothetical estuary had 6000 m² of marsh and 2000 m² of open water originally (T_1). By T_2 , it has experienced a 25% loss of wetland area. Using Equation 1, MA_2 becomes 4500 m² and WA_2 becomes 3500 m². Equation 2 calculates an effective functional loss to the conceptual estuarine ecosystem of 57.1% while the areal loss of marsh was only 25%. In other words, at T_1 every m² of aquatic habitat could potentially interact with 3 m² of intertidal marsh. By T_2 , the increased open water area coupled with the decreased marsh area has altered that interaction to where every m² of aquatic habitat

could interact with only 1.29 m² of wetland. This 57.1% decrease is the functional loss of dilution, an *added dimension* in the wetland loss problem. The functional loss we present here is even greater if the aquatic habitat deepens as marsh is converted to open water.

Functional Losses, Dilution, and Estuarine Morphology

The most important controller of the magnitude of this functional loss appears to be estuarine morphology; specifically, the basic ratio of intertidal wetland to open water. In Figures 1-3, we conceptually represent three estuarine types with different marsh:water ratios. The first is an open bay, with limited marshes fringing the perimeter and where most of the total area is open water (Figure 1a). Drowned river valley estuaries, such as Chesapeake Bay and Narragansett Bay, RI, are examples of this morphological type. The initial marsh area (MA_1) is 1000 ha and open water area (WA_1) is 10000 ha. Figure 1b shows the areal loss of marsh calculated with Equation 1 and the functional loss of that marsh to the ecosystem, after accounting for the *added dimension* of wetland loss (Equation 2). The functional loss is only slightly greater than the areal loss, because the area of aquatic habitat is initially large, and increases little as marsh becomes open water--interactions with the marsh are already considerably diluted.

The second type of estuary, the saltmarsh coastal plain geomorphology, has more extensive marsh areas interspersed by a network of tidal channels (Figure 2a). This estuarine type is characteristic of the mesotidal southeastern U.S. coast and includes estuaries such as North Inlet, SC, and Sapelo Island, GA. In this scenario, the functional loss of marsh is considerably greater than the areal loss because more marsh originally interacted with less open water. A 40% areal loss in such an estuary, for example, represents a 74.3% loss of the ecological function of that marsh (Figure 2b).

Estuaries with expansive microtidal marsh systems and few tidal channels, such as those found in coastal Louisiana and south Florida, are most strongly affected by the conversion of wetlands to open water (see Figure 3a). The dilution effects of this *added*

dimension are very striking in this morphological type. Here, a small conversion of marsh to open water dramatically reduces the functional relationship between the two. Beginning with 10000 ha of marsh (MA_1) and 500 ha of open water (WA_1), each hectare of open water may potentially interact with 20 ha of marsh. A conversion of 10% of the marsh area to open water reduces that possible interaction to 6 ha of marsh per hectare of water column—a 70% reduction (Figure 3b).

Marsh:Estuarine Exchanges

In these conceptual models, we emphasize the coupling between the wetland and water column habitats—and the related exchanges—in our determinations of functional loss. Data from exchange studies are highly variable, and generalizations about marsh:water column interactions are difficult (Nixon, 1980). Work published in the 1960's and 1970's implicated coastal marshes as sources of organic carbon, but with a wide range of flux values. Coastal marshes appear to transform nitrogen—by importing oxidized inorganic species and exporting dissolved and particulate reduced forms—as well as phosphorus—by taking up total phosphorus and releasing remobilized phosphate (Nixon, 1980). In the 1980's, additional studies have quantified exchanges between marshes and adjacent water bodies. Using a marsh flume on Carter's Creek, VA, Wolaver et al. (1980, 1983) reported marsh uptake of ammonium (NH_4^+), nitrate (NO_3^-), phosphate ($PO_4^{3-}=SRP$), total N and P, and dissolved organic nitrogen (DON) and export of nitrite (NO_2^-). Sapelo Island, GA, marshes released particulate organic carbon (POC) but show no significant net dissolved organic carbon (DOC) flux (Chalmers et al., 1985). Freshwater marshes along the North River, MA, imported all forms of dissolved inorganic nitrogen (Bowden, 1986). Work at North Inlet, SC, showed net fluxes of NH_4^+ , NO_3^- , and NO_2^- onto the saltmarsh while DON was released (Whiting et al., in prep.). Results continue to be variable in both magnitude and direction of flux, depending in part on estuarine morphology.

We have quantified nutrient and materials exchanges in the brackish and saline marshes of the Barataria Basin estuary, Louisiana (see Conner and Day, 1987, for a

detailed site description). At each site, we used a throughflow marsh flume to measure fluxes between the vegetated marsh and the inundating water column. Details of the sampling, computational, and statistical methods are presented in Chapter 1 (Childers and Day; 1988). We will use data from the saltmarsh flume to examine the ecological implications of functional losses [the *added dimension*] to the wetland loss dilemma.

The Barataria Basin estuary is in a later, deteriorating stage of the deltaic cycle (Madden et al., 1988; others). Wetland loss rates of 0.8 to 1.5% of the remaining marsh yr⁻¹ are among the highest in the state's coastal zone (Craig et al., 1979; Gagliano et al., 1981; Scaife et al., 1983; Conner and Day, 1987; Swensen and Turner, 1987; Templet and Meyer-Arendt, 1989). Using the 0.8% yr⁻¹ loss rate, as a conservative value, and the 1987 area of saltmarsh and open water in the Barataria Basin estuary (Conner and Day, 1987), we hindcast 1967 marsh and open water areas to be 74900 ha and 70600 ha, respectively, and forecast areas for the year 2007 of 53603 ha (saltmarsh) and 91897 ha (open water). Areal and functional losses were calculated for this specific Barataria Basin saltmarsh example, with the 1967 hindcast values as initial areas and using equations 1 and 2 (Figure 4). In this scenario, 15.4% of the 1967 area of saltmarsh had been lost by 1987, and by the year 2007 28.4% will be lost (Equation 1). The functional loss related to habitat conversion and dilution, however, was calculated to be 27.3% in 1987 and 45% in 2007 (Figure 4). If the 1987 areas are used as initial values, the 0.8% yr⁻¹ loss rate again generates a 20-year areal loss of saltmarsh of 15.4% by 2007 but with a functional loss of over 24% in the same time period.

The Functional Loss of Marsh:Estuarine Exchanges

To demonstrate the importance of the *added dimension* of habitat conversion and functional loss to the estuarine ecosystem, we have applied our flux results from the Barataria Basin saltmarsh flume to the wetland loss scenario for this estuarine system (Figure 4). Several generalizing assumptions were necessary, though. We assumed that:

1) our area-specific marsh fluxes are representative of Barataria Basin saltmarshes, 2) in a complete flooding and draining tidal cycle, the full area of these flat saltmarshes come in contact with the water column, and 3) fluxes to or from the marsh are uniformly distributed through the full area (volume) of estuarine open water.

Ammonia and nitrate-nitrite ($\text{NN}=\text{NO}_3^- + \text{NO}_2^-$) flux data from the saltmarsh flume are presented in Table 1. In August, 1987, we measured a highly significant NH_4^+ and NN (as dissolved inorganic N=DIN) uptake by the saltmarsh that totaled 2.2 mg m⁻² h⁻¹. Ammonia was exported from the saltmarsh during all other samplings and NN was released in all but the November, 1987, samplings. The largest export was measured in April, 1988, when the saltmarsh supplied 3.3 mg DIN-N m⁻² h⁻¹ to the estuarine water column (Table 1). The inundation regime at the saltmarsh flume was determined from continuous water level records at each site (see Chapters 1 and 4—Childers and Day, 1988; Childers et al., submitted). Monthly inundation data, as percent of total hours, are shown in Table 1. Given 9.7 and 16.2% inundation for these two months, respectively, total monthly fluxes are 159 mg DIN-N m⁻² mo⁻¹ (August, uptake) and 385 mg DIN-N m⁻² mo⁻¹ (April, release). The total mass of N exchanged by the saltmarsh in 1987 is calculated as:

$$3) \quad \text{MA}_1 * \text{flux}_m * (0.01 \text{ m}^2 \text{ kg ha}^{-1} \text{ mg}^{-1}) \quad (\text{in kg h}^{-1})$$

where flux_m is the empirical value from our flume studies and $\text{MA}_1=63360$ ha, the initial [1987] area of saltmarsh (Conner and Day, 1987). The saltmarsh thus removed 1394 kg DIN-N h⁻¹ from the water column in August, 1987, and in April, 1988, it supplied 2091 kg N h⁻¹ to the aquatic habitat (=total marsh flux). The effect of these exchanges on the estuarine water column, per unit area of open water (flux_{ow}), is determined as:

$$4) \quad \frac{\text{Total Marsh Flux}}{\text{WA}_1} * (100 \text{ mg ha kg}^{-1} \text{ m}^{-2}) \quad (\text{in mg m}^{-2} \text{ h}^{-1})$$

given a 1987 open water area (WA_1) of 82140 ha. The 1394 kg N h⁻¹ uptake by the saltmarsh represents a 1.7 mg N m⁻² open water h⁻¹ (=123 mg m⁻² mo⁻¹) removal from

the estuarine aquatic habitat while the 2091 kg N h⁻¹ export from the saltmarsh supplies that open water system with 2.6 mg N m⁻² open water h⁻¹ (=297 mg DIN-N m⁻² mo⁻¹).

As we note above, this 15.4% loss of saltmarsh area in the Barataria Basin estuary (1987-2007) may actually represent a 24% functional loss when the *added dimension* of intertidal to aquatic habitat conversion is considered. The effect this has on the marsh:water column exchanges is exemplified by recalculating exchanges in the 15.4% wetland loss scenario. Equations 3 and 4 are simplified into 5a and 5b, and fluxes determined with areas forecast for the year 2007 [MA₂=53603 ha and WA₂=91897 ha]:

$$5a) \quad MA_1 * flux_m = WA_1 * flux_{ow}$$

$$5b) \quad MA_2 * flux_m = WA_2 * flux_{ow}$$

where flux_{ow} is the marsh nutrient flux per unit area of open water. After a loss of 15.4% of saltmarsh in this conceptual scenario—as wetland losses are typically reported—Barataria Basin saltmarshes may only be capable of removing 1.3 mg N m⁻² open water h⁻¹ inundated from the aquatic habitat (Equation 5b), where before they removed 1.7 mg N m⁻² open water h⁻¹. In April, 1988, the Barataria Basin saltmarsh exported 2.6 mg N m⁻² open water h⁻¹ to the interacting water column, but the functional loss associated with a the 0.8 % yr⁻¹ areal loss rate may reduce this N supply to 1.9 mg m⁻² h⁻¹. Often, water column productivity patterns are enhanced near the marsh:open water interface, reflecting a wetland nutrient source such as this (Owens et al., 1986; Childers and Day, unpubl. data). In this scenario, a conversion of 15.4% of the intertidal marsh to open water represents a more than 24% decrease in marsh nitrogen supply to the estuary. Where aquatic productivity is at least partly dependent on a supply of limiting nutrients from the nearby marsh, wetland loss may have deleterious ecological effects exceeding the actual areas loss (Figure 4). Whether the marsh removes excess inorganic nutrients from the estuary or supplies critical limiting nutrients to the estuary, its loss should be viewed as a functional

loss to the ecosystem (45% in the saltmarshes of the Barataria Basin estuary, between 1967 and 2007), augmenting the loss of land area (28.4% in our scenario).

A basic tenet of estuarine ecology has been that organic matter originating from marsh macrophytic productivity is exported to estuaries where it provides nutrition to a variety of consumers either directly, or indirectly through microbial processing (Odum, 1984; Peterson and Howarth, 1987; many others). Organic matter flux data from the August, 1987, and April, 1988, samplings of our saltmarsh flume show some interesting differences in behavior of dissolved (DOC) and particulate (POM=POC+PON) organics (Table 1). In August, POM fluxes were comparatively low and only the measured PON uptake was significant, while the marsh took up a large amount of DOC ($=210.5 \text{ mg m}^{-2} \text{ h}^{-1}$). In April, 1988, we observed the largest POM flux at this flume site as both POC and PON were imported, yet the saltmarsh released DOC (all three April fluxes were significant; see Table 1). In all samplings, though, the saltmarsh supplied some form of organic matter to the estuarine water column.

We return to the hypothetical 1987-2007 Barataria Basin saltmarsh scenario of a 15.4% wetland loss, by area, to demonstrate the consequences of the associated functional loss on organic matter export by the saltmarsh. In April, 1988, the saltmarsh exported $165 \text{ mg DOC-C m}^{-2} \text{ h}^{-1}$, equivalent to $19.3 \text{ kg m}^{-2} \text{ mo}^{-1}$. The June, 1987, POC+PON release amounted to $4.9 \text{ mg POM m}^{-2} \text{ saltmarsh h}^{-1}$, or $1.44 \text{ kg m}^{-2} \text{ mo}^{-1}$ (Table 1). These fluxes translate into inputs to the estuarine aquatic habitat of $14.8 \text{ kg DOC-C m}^{-2} \text{ open water mo}^{-1}$ and $1.1 \text{ kg POM m}^{-2} \text{ open water mo}^{-1}$ (using 1987 areas and Equation 5a). At an areal loss rate of $0.8\% \text{ yr}^{-1}$, the year 2007 will bring a conversion of 15.4% of the 1987 saltmarsh habitat to open water. By this time, the same rate of organic matter export by these marshes may supply only $11.3 \text{ kg DOC-C m}^{-2} \text{ open water mo}^{-1}$ and $0.6 \text{ kg POM m}^{-2} \text{ open water mo}^{-1}$ to the saltmarsh water column. In both cases, the 15.4% loss of marsh represents a decrease of over 24% in organic matter supply to the estuarine water column. The impact of wetland loss on estuarine detrital dynamics, which to some degree depend on

a tidally-mediated supply of organic substrate, is a product of both areal and functional loss of marsh. Ironically, this impact may be greater in marsh-dominated estuaries, such as the Barataria Basin, than in those with an open bay-type morphology.

Conclusions

The concept of areal transitions when marsh becomes open water may be intuitively obvious. The importance of this dimensional difference, however, cannot be overemphasized. The loss of marsh to an estuary must be considered in the context of both the marsh and the estuarine water column interacting with that marsh. Apparently, as the wetland:open water ratio increases with varying estuarine morphology, the functional loss of that marsh, relative to the aquatic habitat, can increase dramatically. Thus, the type of estuary may play an important role in determining how great the functional loss of marsh conversion to open water will be. In the three examples used here (Figures 1-3), the conceptual estuary with the greatest proportional area of marsh suffered the greatest functional loss per unit area of wetland lost, with functional loss to the estuarine ecosystem potentially exceeding 90% after only a 25% loss of marsh area (Figure 3b). Even in the saltmarshes of the Barataria Basin estuary, where historical wetland conversion to open water has already lowered the marsh:open water ratio to less than one, our 20- and 40-year hypothetical wetland loss scenarios demonstrated that the functional loss of saltmarsh here may be nearly twice the areal loss of those wetlands.

Clearly, the conceptual models we have presented here are generalizations. The hypothetical examples are based on a number of assumptions, including complete inundation of the entire area of marsh during nutrient and organic matter exchange and complete mixing of the water column. While we used published areas and loss rates in the Barataria Basin saltmarsh example, the initial areas assigned to the conceptual estuaries in Figures 1-3 were arbitrary, and for explanatory purposes only. Testing our hypothesis of functional wetland loss as an *added dimension* in wetland loss would be difficult, but

nonetheless, the dynamic importance of marsh:open water areal ratios should be considered in estuaries characterized by wetland loss.

Intertidal marshes supply energy to their associated estuaries in a variety of forms. In some cases, organic matter fixed on the marsh surface is exported for oxidation in the water column. In others, nutrients critical to aquatic primary productivity are regenerated by the marsh. Marshes also export reduced chemical compounds (including N, Fe, and S). Some of the reduced energy stored in these compounds may create pelagic microbial biomass when they are oxidized in the water column (Peterson et al., 1986; Howarth, 1984). In estuaries characterized by wetland conversion to open water, such as those of coastal Louisiana, there may well be a point at which the areal loss of marsh is sufficient to cause nearly total functional loss of that marsh from the system. Even though marsh would remain, it is so effectively "diluted" by an estuarine water column once strongly dependent on exchanges with the marsh that, in an ecological sense, it is functionally lost to the system.

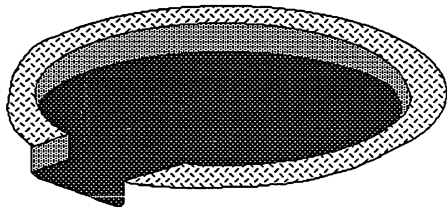
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a.



Initial Areas: Marsh=  =1000
Open Water=  =10000

b.

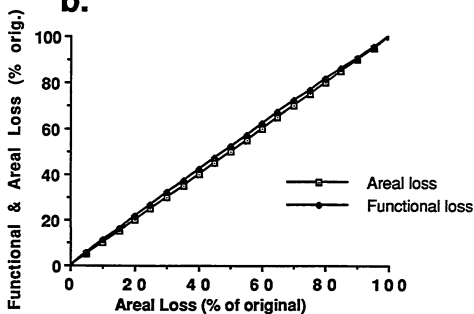


Figure 5-1: a) Conceptual diagram of a bay-type estuarine morphology dominated by the aquatic habitat, with marshes fringing the margin. Marsh:open water areal ratio is 1:10 in this conceptual estuary.; b) Results of the marsh loss and functional loss models from Equations 1 and 2, respectively (see text). The marsh loss curve plots actual areal marsh loss against itself, from Equation 1. This compares to the functional loss curve which accounts for the conversion of that marsh to open water.

a.

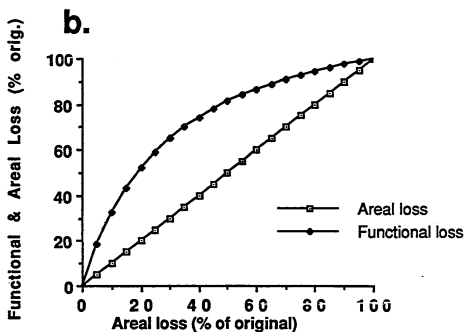
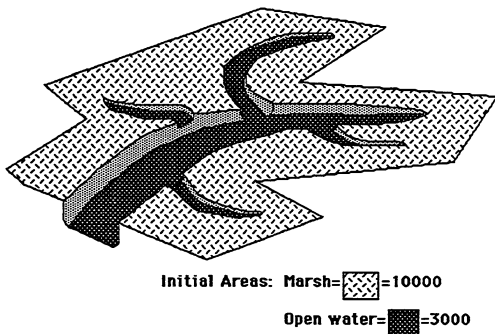
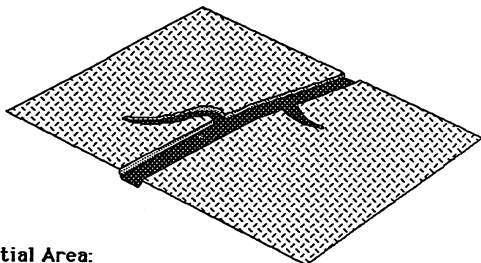


Figure 5-2: a) Conceptual diagram of a mesotidal, enclosed estuarine morphology with marshes and an extensive network of tidal creeks. Marsh:open water areal ratio is 10:3 in this conceptual estuary.; b) Results of the marsh loss and functional loss models from Equations 1 and 2, respectively (see text). The marsh loss curve plots actual areal marsh loss against itself, from Equation 1. This compares to the functional loss curve which accounts for the conversion of that marsh to open water.

a.



Initial Area:


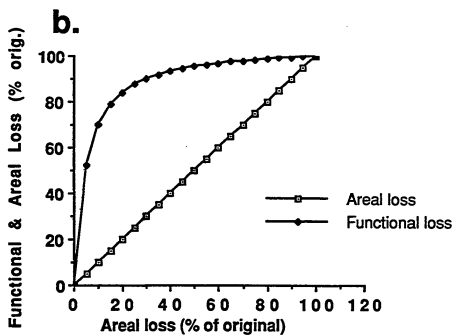
Marsh=  =10000Open Water=  =500

Figure 5-3: a) Conceptual diagram of a microtidal, expansive estuarine morphology dominated by the marsh habitat, infrequently intersected with tidal creeks. Marsh:open water areal ratio is 20:1 in this conceptual estuary.; b) Results of the marsh loss and functional loss models from Equations 1 and 2, respectively (see text). The marsh loss curve plots actual areal marsh loss against itself, from Equation 1. This compares to the functional loss curve which accounts for the conversion of that marsh to open water.

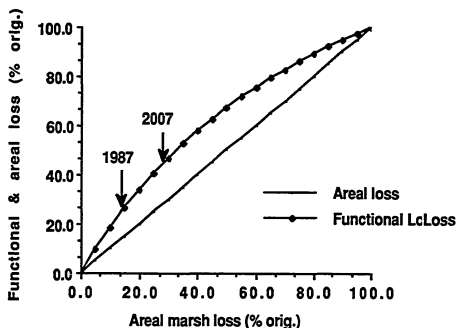


Figure 5-4: Results of the marsh loss and functional loss models from Equations 1 and 2, respectively, for the Barataria Basin saltmarsh example (see text). Curves are based on published marsh and open water areas for 1987 (Conner and Day, 1987) and a $0.8\% \text{ yr}^{-1}$ loss rate. The marsh loss curve plots actual areal marsh loss against itself, from Equation 1. This compares to the functional loss curve which accounts for the conversion of that marsh to open water.

Table 5-1: Barataria Basin saltmarsh flume data for inorganic nitrogen and organic matter flux (see text for abbreviations). Where flux shown is significantly different from zero ($\alpha \leq 0.05$), the value is in **bold type**; where flux shown is highly significantly different from zero ($\alpha \leq 0.01$), **bold italics** are used. "+" flux represents an import to the marsh, from the water column; "-" flux is an export from the marsh to the overlying water column.

<u>Date</u>	<u>NH₄</u>	<u>Flux Data (mg X m⁻² h⁻¹)</u>			<u>DOC</u>	<u>Flooding Data (%)</u>
		<u>NN</u>	<u>POC</u>	<u>PON</u>		
June, 1987	-0.3	-0.2	-3.9	-1.0	-9.2	40.8
August, 1987	2.0	0.2	-2.7	1.3	210.5	9.7
November, 1987	-0.2	0.2	6.2	0.8	-78.1	36.0
April, 1988	-2.2	-1.1	23.5	3.46	-165.0	16.2

VITA

Daniel L. Childers was born on October 1, 1961, in Arlington, Virginia, and was raised in Oakton, Virginia. He graduated from Oakton High School in June, 1979. His undergraduate education at The University of Virginia, in Charlottesville, included Bachelor of Arts degrees in both Biology and Environmental Science as well as a year and a half of field and lab research experience studying the ecology of a population of freshwater clams [under the guidance of Dr. Daniel Hornbach]. He graduated from The University of Virginia in May, 1983, and continued his education at the University of South Carolina, where he received a Master's of Science degree in Marine Science in August, 1985. Under the guidance of Dr. Henry McKellar, Jr., his Master's thesis was entitled "A Simulation of Saltmarsh Watercolumn Dynamics". Since August, 1985, he has been at the Coastal Ecology Institute and Department of Marine Sciences in Louisiana State University's Center for Wetland Resources. His major advisor at Louisiana State University, and for this dissertation, was Dr. John W. Day, Jr.

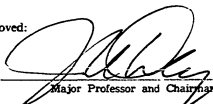
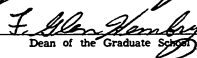
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Candidate: Daniel L. Childers

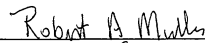
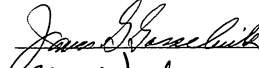
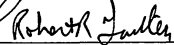
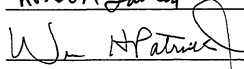
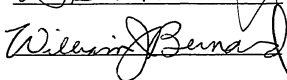
Major Field: Marine Sciences

Title of Dissertation: Marsh: Water Column Interactions in Two Louisiana Estuaries:
Flux Measurements and Conceptual Implications

Approved:


Major Professor and Chairman

Dean of the Graduate School

EXAMINING COMMITTEE:

Date of Examination:

June 9, 1989