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Factors Affecting Sediment Transport, Deposition and Erosion in Intertidal Wetlands in Louisiana.

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FACTORS AFFECTING SEDIMENT TRANSPORT, DEPOSITION AND EROSION IN INTERTIDAL WETLANDS IN LOUISIANA

A Dissertation

Submitted to the Graduate Faculty of the
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and
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by

Roelof Maria Boumans,
University of Amsterdam, 1986
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ABSTRACT

Processes leading to land loss in Louisiana are defined measured and analyzed with special emphasis on the deposited and suspended sediment budgets. The findings were used to assess coastal management in Louisiana. In a spatial model of long term habitat succession, the degradation of a Louisiana wetland was based upon simulated exchanges of sediments across irregularly shaped polygons. Habitat distribution within a cell was estimated by elevation The model findings were confirmed when the impact of two Louisiana marsh management plans caused reduced short-term sedimentation due to impaired water and materials flux. The sedimentation-erosion table (SET) is described and applied to assess the effect of experimental sediment fences on the elevation developments in intertidal and shallow sub tidal areas. The SET is precise to within a 1.5 millimeter range SET measurements made over a three year period showed a significant elevation increase of 1.38 cm/yr at sites close to sediment fences, and a non significant decreases of -0.73 cm/yr at sites away from the fences. The wave transmissivity coefficient for sediment fences is presented The potential impact of Louisiana marsh management and sediment fences on elevation changes and subsequently vegetation establishment were evaluated with the use of the hydrodynamic sector of the General Ecosystem Model (GEM) which was calibrated for inter tidal water bodies.

The results of this research are important because they provide more inside on the sediment dynamics of coastal wetland systems and the potential impact of different management options. The simulation models are designed to predict spatial changes and potentially can be used to develop dynamic geographic information systems (GIS)

CHAPTER 1

INTRODUCTION

Coastal land loss is one of the most serious problems faced by the state of Louisiana today. Loss rates as high as 120 km^2 of coastal wetlands per year have been reported as a result of natural and human induced processes (Gagliano et al. 1981; Baumann and DeLaune 1982; Turner and Cahoon 1987; Penland et al. 1990). Although coastal wetland loss is a global problem documented on many coasts (Milliman et al. 1989; Jeftic et al. 1992; Tooley and Jelgersma 1992; Stanley and Warne 1993), Louisiana represents one of the worst cases. The state contains more than 41% of the US coastal wetlands but 80% of the total coastal wetland loss (Turner and Cahoon 1987). Wetland loss may become worse as 0.5 to 1 meters of relative sea level rise is expected over the next century (Turner 1991). Studies indicate that global warming will cause an acceleration in the rate of eustatic sea level rise (Warrick and Oerlemans 1990; Bird 1993), while in addition, deltaic areas, like the Mississippi delta are subsiding, thus exacerbating the problem of eustatic sea level rise (Jeftic et al. 1992). Other causes documented include the dredging of canals, building of shore protection structures, and reduced sediment loads in the fresh water sources caused by river control structures and flood protection (Baumann and Adams 1981; Turner and Cahoon 1987) .

Eighty percent of the world population resides in the coastal zone, therefore coastal land loss has the potential for large economic but also ecological disasters (Gornitz et al. 1982). Such disasters are already documented for Bangladesh (Milliman et al. 1989) and Egypt (Stanley and Warne 1993). Salt marshes, often credited for their importance in fisheries and natural shoreline protection (Nixon 1980; Turner 1982), are not likely to migrate land ward during the prognosed sea level rises because they are bordered with an increasing amount of coastal protection structures necessary to guarantee the safety of the residing coastal population (Hackney and Cleary 1987; Pethick and Reed 1987; Rijkswaterstaat 1990; Williams et al. 1991). Marshes are expected

to survive only in cases where there is sufficient supply of new sediments to maintain an inter tidal elevation (Hatton et al. 1983; Templet and Meyer-Arendt 1988; Day and Templet 1989).

Deterioration of coastal wetlands is due to a collection of complex interactions between geological and biological phenomena (Craig et al. 1979; Gagliano and Wicker 1989; Penland et al. 1989; Penland et al. 1990; Evers et al. 1992). Inland wetland conversion to open water accounts for 70-93% of the total wetland loss (Leibowitz and Hill 1987), which appears to be related to the physiological stress associated with increased submergence and elevated salinity (Mendelssohn and McKee 1988). Following plant die-back, small ponds develop in the interior of the marsh. The formation of small ponds in the marsh allows for wind-generated waves to develop in the interior of the marsh.

Vertical sediment accretion on the marsh surface, which determines whether or not the wetland can keep pace with rising water levels, depends on the balance between erosional and depositional processes. Sediment deposition may occur simultaneously both in space and in time with erosion and resuspension (Anderson et al. 1981). Although deposition of suspended sediments is high in many inter tidal areas in Louisiana, resuspension of these sediments often prevents marsh accretion from occurring (Baumann et al. 1984). The forces associated with sediment suspension in inter tidal waters have not been intensively studied. Casual relationships suggested erosional events during storms (Baumann et al. 1984; Meeder 1987; Rejmanek et al. 1988), although in inter tidal water areas, gentle waves or "wind ripples" can also cause appreciable resuspension and erosion (Anderson 1972; Carper and Bachmann 1984). In general, waves appear to be more important in resuspending bottom sediments than water currents (Gons et al. 1986).

Besides erosion, elevation can be lost through subsidence due to subsurface processes. The processes range from geological down warping of the oceanic crust to compaction of the upper sediment layer, and include de watering, organic matter decomposition and the release of gases produced in decomposition. Organic matter decomposition is potentially compensated at a vegetated site by newly produced organic matter as plants contribute to elevation by litter fall and root production (Nyman and DeLaune 1990).

Vegetation also influences the physical processes of sediment accretion in marshes by slowing water currents. Vegetation also appears to sequester sediments from the water column during tidal flooding. Fifty percent of the sediments lost from the water column during a spring tide were adhered to the stems of salt marsh grass in a Delaware marsh (Stumpf 1983). Benthic macrophytes may decrease erosion in shallow ponds by dampening wave energy and reducing resuspension (Ward 1984). Some researchers have suggested that erodebility of mud flats is highest during winter months when populations of benthic algae and bacteria are lowest (Young 1977; Coles 1979; Frostick and McCave 1979). Thus for each environment in the cycle of wetland deterioration (marsh pond and mud flat) vegetation can play an important role in reducing erosion and increasing deposition.

STUDY OBJECTIVES

The studies focused on the processes of erosion, transportation and deposition of sediments in the Louisiana coastal wetlands in relation to marsh stability. The primary objectives were to:

1) Quantify the relationship between sediment erosion and the following factors: wave stress, soil strength, and flood duration.

2) Determine the relative importance of storm and fair weather conditions in affecting sediment deposition and erosion in shallow coastal ponds and mud flats.

3) Develop a mathematical model to describe sedimentation and erosion patterns in shallow coastal ponds and mud flats.

4) Use data to test the validity of mathematically generated hypotheses about the effects of wind speed, fetch, water depth and wave parameters on sedimentation patterns.

The understanding gained through the primary objectives were weighted against two management practices as they are practiced in Louisiana today. The so called 'marsh management' option involves the regulation of hydrology with the use of levees and water flow regulating structures (Clark and Hartman 1990), while the 'sediment fence' method tends to regulate hydraulics with the use of structures open to water flow (Dijkema et al. 1988). Additional objectives involving these management options in coastal marshes are:

1) Determine the role of the following factors in encouraging vegetation re-establishment in an area of deteriorated coastal marsh: elevation, soil formation, water level and wave energy.

2) Monitor the effect of marsh management on sediment erosion, transportation and deposition patterns in inter tidal coastal areas.

3) Monitor the effects of sediment fences on sediment erosion, transportation and deposition patterns in inter tidal coastal areas.

4) Synthesize the information for management design criteria for coastal managers.

The following hypothesis then were formulated:

EROSION:
Bottom shear stress, caused by wave orbital velocities in coastal wetlands, occasionally will exceed the strength threshold values associated with the bottom sediments in coastal wetlands to cause erosion.

TRANSPORT:
Newly introduced mineral suspended sediments are crucial to maintain marsh habitat.

DEPOSITION:
Sedimentation will occur when water turbulence levels are sufficiently decreased so that sediments are not kept in suspension.

ELEVATION:
1) Elevation is primarily a balance between erosion and deposition.
2) Elevation increase will be higher in areas near the sedimentation fences than in the control areas.
3) Elevations will be higher in small fetch areas versus large fetch areas
4) Elevation increase will be greater during storm events than during fair weather conditions

VEGETATION:
Vegetation will become established more rapidly in higher elevations as opposed to lower elevation.

The specific objectives set for the modeling effort involve process sensitivity testing and spatial representation. In particular the model was used to evaluate the following:

1) The sensitivity of erosion versus subsidence with respect to elevation loss.
2) The effect of higher suspended sediment concentrations versus the effect of lower wave energy levels on elevation increase of mudflats.
3) The optimum design criteria for placing brush fences.

CHAPTER 2

A POLYGON-BASED SPATIAL (PBS) MODEL FOR SIMULATING LANDSCAPE CHANGE

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INTRODUCTION

Explicit incorporation of spatial parameters into ecological simulation models is important. The strong spatial aspects of many ecological processes requires environmental regulatory agencies to consider information on spatial and temporal responses (Weinstein and Shugart 1983; Risser et al. 1984; Sklar et al. 1985; Forman and Godron 1986; Turner 1987). In aquatic systems for example; how does the location of dikes and weirs alter fish migration, or how does the alteration of water flow patterns impact regional marsh productivity, or what types of coastal landscape changes lead to indirect habitat modifications (Chabreck 1972; Mendelssohn et al. 1983; Steiner 1983; Costanza et al. 1986)? These decisions are difficult to make without at least a conceptual model of spatial interactions. How then does one incorporate spatial interactions into ecological simulation models? To address this question we present a spatial model of a case study which uses wetland loss in Louisiana as the focus for analysis, irregular shaped grid cells (i.e., polygons) as the spatial classification, and population interaction equations (May and MacArthur 1972; Maynard Smith 1974) as the basis for habitat succession.

Numerous researchers have documented that wetlands in Louisiana are diminishing at a rate estimated to be as high as 100 km^2 a year (Gagliano and Beek 1970; Adams et al. 1976; Craig et al. 1979; Hopkinson and Day 1980; Gagliano et al. 1981; Boesch 1982; Scaife et al. 1983; Turner and Cahoon 1987; Leibowitz et al. 1988). Factors influencing land loss include: (1) rising sea levels, (2) subsidence of deltaic sediments, (3) nutrient and inorganic sediment depletion, (4) canal construction (5) salt water intrusion, and (6) decreasing bio-productivity. Attempts to model land loss as the cumulative impact of all these parameters have focused on multi variate regression analyses (Scaife et al. 1983; Deegan et al. 1984) and process-based simulation models with fixed grid cells

(Sklar et al. 1985; Costanza et al. 1988). We continue this analysis by designing a spatial model of land loss which uses irregularly shaped grid cells that conform to the natural hydrological "mini-basins" in a landscape.

The use of polygons in a spatial ecosystem model is new. Previous models with spatial detail were mostly of the fixed grid type (Costanza and Sklar 1985), the node-network type (Hopkinson and Day 1980), and the finite element models so often used for hydrodynamic simulations, such as EPA's Storm Water Management Model (1975). The polygon approach has the advantage of being less constrained by artificial boundary conditions and hence the "more natural" way to subdivide a landscape. The LaBranche wetland in Louisiana was chosen to test this approach because it had been dissected into five separate sub-basins by attempts at flood control and agriculture early this century.

We approached simulation of habitat succession within a polygon-based model differently than earlier fixed grid models. In a fixed grid design the simulation of habitat succession is a simple process because each cell is composed of only one habitat and the rules governing change are implemented on a cell by cell basis (Sklar et al. 1985; Costanza et al. 1986; Sklar et al. 1989). In an irregular grid design, such as the one we present, the cells can be composed of many habitats and succession is implemented on a habitat by habitat basis within each cell. We simulated the "competition" for space within a polygon by modifying the population models of (May and MacArthur 1972). These equations were used in the PBS model because the temporal interactions of the three major habitat types within a polygon (swamp, marsh, and open water) appear to mimic competitive exclusion processes that is, open water habitats give way to marsh habitats and marsh habitats give way to swamp habitats as deltas and coastal marshes prograde.

and vice versa as deltas and marshes degrade (Gagliano and Beek 1970; Baumann and Adams 1981). These equations are widely accepted by population ecologists and we do not question their validity in this paper.

The LaBranche wetland, located on the southern shore of Lake Pontchartrain between the Bonnet Carre Spillway and New Orleans (Figure 2.1), is among the largest and most productive low salinity habitats within the lake Pontchartrain basin (Cramer et al. 1978). Since 1900, when 1000 hectares of marsh were reclaimed (i.e. impounded and drained) for an unsuccessful agricultural project, the wetland has undergone a progressive deterioration resulting in considerable land loss. Total land loss has increased over the last 50 years in the LaBranche wetland (Figure 2.1) from 106 ha in 1952 to 165 ha in 1956 to 1645 ha in 1972, decreasing slightly to 1607 ha in 1982 (Pierce et al. 1985).

METHODS

The LaBranche PBS model (Figure 2.2) divides the wetland study area into five different hydrological polygons based on the description of the area by Pierce et al. (1985) and the geometry of the sub-basins. Each polygon and each boundary condition was given a name and number to facilitate discussion and simulation. Borders along interconnected cells were registered in the model as numbers 1-12 and for each border a flow parameter, indicative of the degree (0-100%) of impoundment in each cell, was assigned. Polygons completely impounded, with no exchange with surrounding cells, were assigned flow values, RES_X (x=border number), of zero for each border. RES_X values for borders 1 to 12 are shown in Table 2.1.

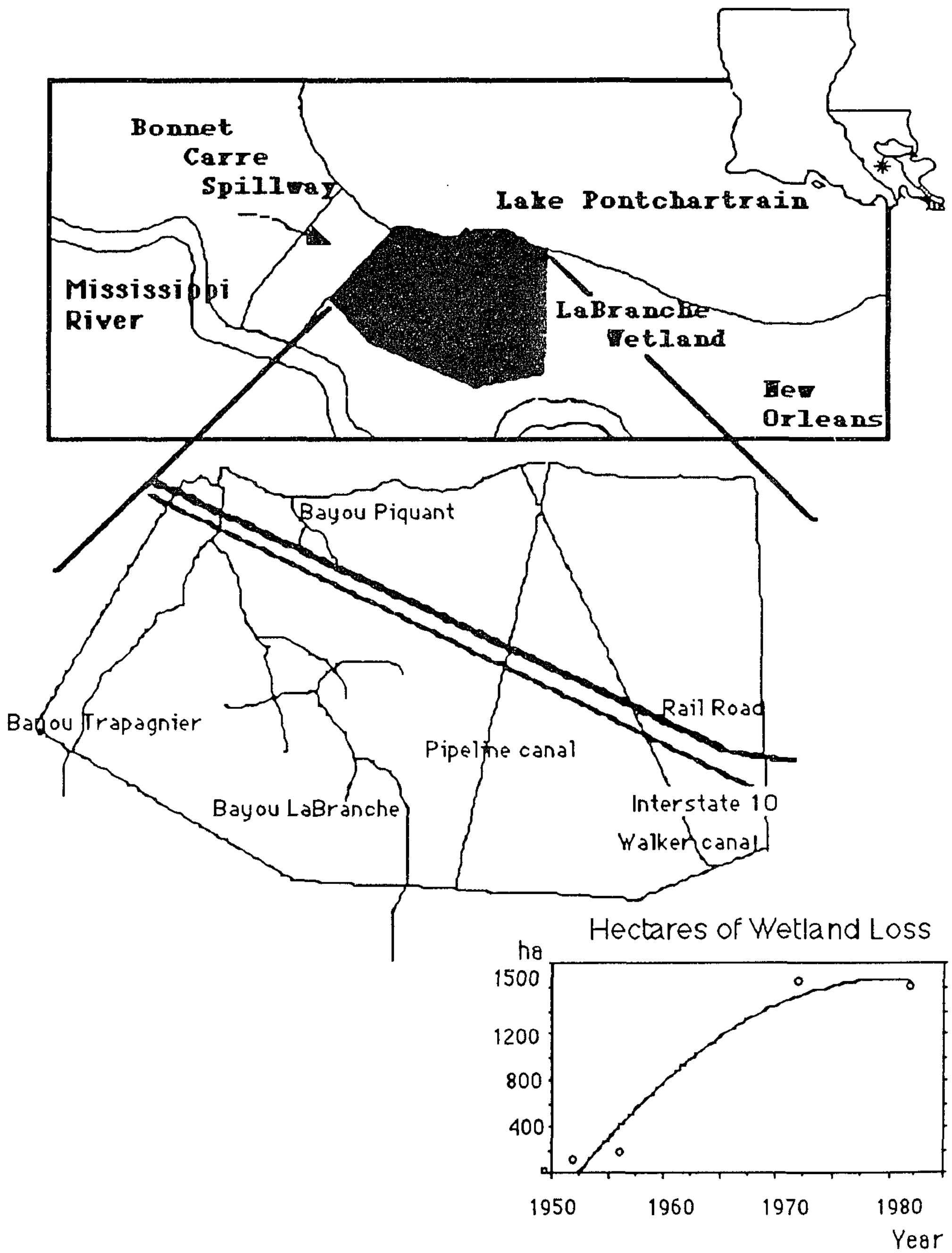

Figure 2.1. Map of the LaBranche wetland study area along the southern coastline of Lake Pontchartrain, Louisiana. Wetland loss has been getting progressively worse in this area as subsidence produces increasing hectares of open water as illustrated in the small plot on the right from (Pierce 1985).

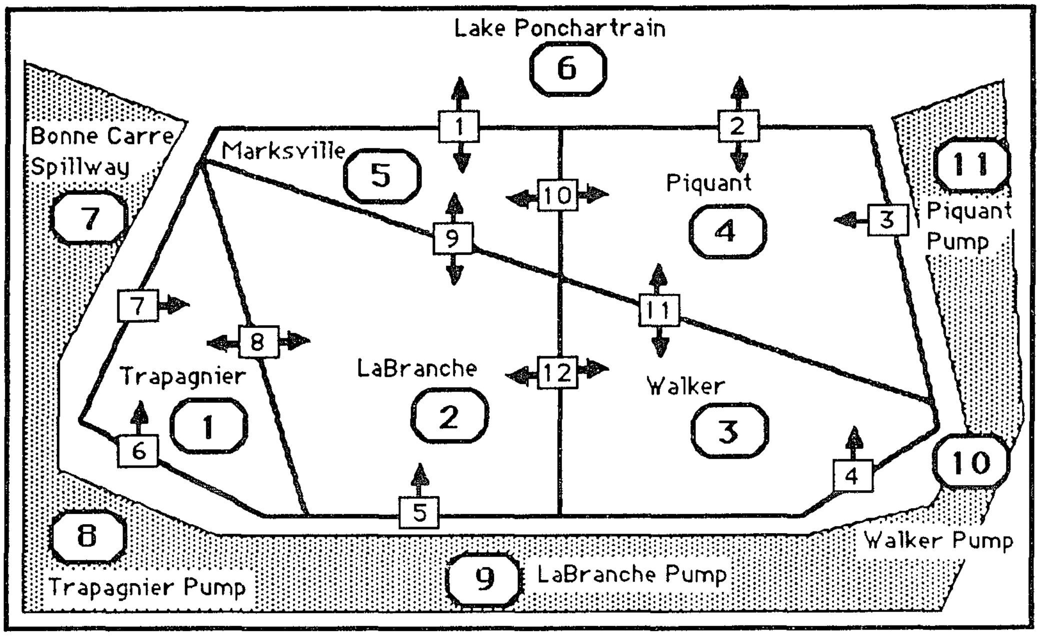

Figure 2.2. Internal polygons (1 to 5) and external drainage basins (6 to 11) were chosen to represent the different hydrological drainage units found in the study area. The arrows represent the direction of water flow between the polygons. The numbers in the boxes (1 to 12) represent the coding used to designate the proportional flow coefficients (Res_X).

The hydrologic forcing functions in the PBS model were based upon the head difference between Lake Pontchartrain water levels and the water levels inside the polygons. Water flowed into Lake Pontchartrain when the sum of runoff and water level in a cell exceeded that of lake water, and flowed into the cells when the sum was less than that of the lake. The average water height of Lake Pontchartrain, based upon water gauge data in the middle of the lake (Stone and Hinchee 1980), varied bimodaly over a year (Figure 2.3). This bimodal distribution was described as:

$$LAKE = 3.81 + SIN(Day \cdot (-0.035))$$

where, LAKE equals the water levels (cm) in Lake Pontchartrain and Day equals the time of year. Water inside the polygons accumulates as a function of lake water inputs, rainfall and each polygon's ability to drain (i.e., RES_X values, Table 2.1). Runoff values equal surpluses of water based upon actual rainfall events from 1910 to 1983, taken from the National Oceanic and Atmospheric Administrations monthly publication, Climatological Data, and from the U.S. Weather Bureau's Decennial Census of U.S. Climate and incorporated into a (Thornthwaite and Mather 1955) water budget program (Yoshioka 1971). The annual average distribution of this runoff (Figure 2.3) was used by the PBS model as fresh water inputs according to the sin function:

$$RUNOFF = 5+3.7 \cdot SIN(Day \cdot 0.0175)$$

where, RUNOFF equals the water level in the polygon due to surplus rain and Day equals the time of year. Most of the year RUNOFF is greater than LAKE and water flows from the wetlands into the lake. When the water elevations in the wetlands is lower than that of the lake, typically during late summer, there is a net flow of lake water into the area. Flows of water between polygons were calculated as difference equations:

Table 2.1. Flow parameters values are indicative of the conductivity across the interface of adjacent polygons. Subscribed values correspond with flow parameter designations in Figure 2.2.

Flow coefficient	Value	Flow coefficient	Value
RES1	0.3	RES7	0
RES2	0.1	RES8	0.4
RES3	0	RES9	0.4
RES4	0.2	RES10	0
RES5	0	RES11	0.3
RES6	0.2	RES12	0.2

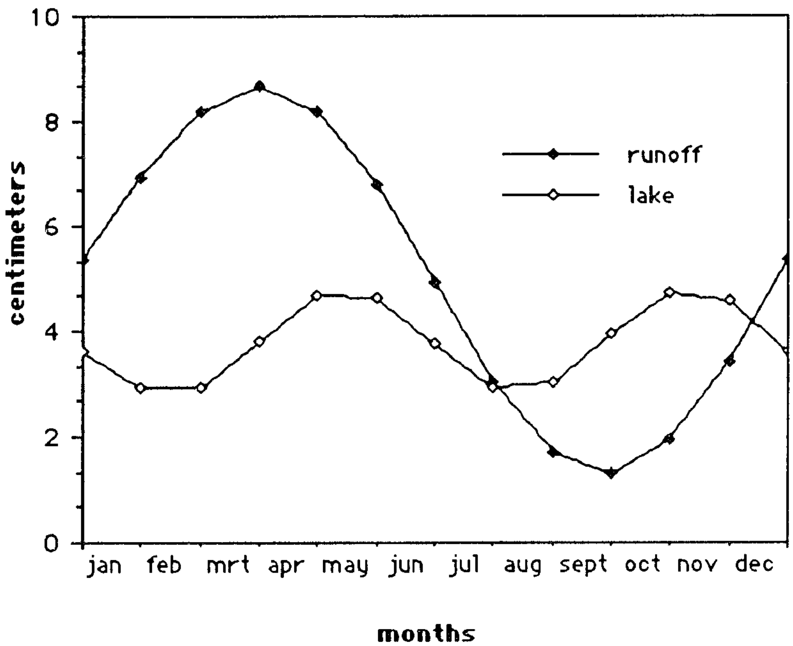

Figure 2.3. The PBS model uses average rainwater surplus (runoff) and average water levels in Lake Pontchartrain (lake), recorded by the U.S. Army Corps of Engineers, as forcing functions. Runoff is the water depth due to excess rainfall as calculated by the Thornthwaite and Mather (1959) water budget. This annual cycle is repeated annually for each of the 28 years of simulation.

$$Flow_{(a\ to\ b)} = (WL_{(a)}-WL_{(b)}) \cdot RES_{(x)} \cdot A_{(a)}$$

where, $WL_{(a)}$ is the water level in polygon (a) and $WL_{(b)}$ is the water level in polygon (b), RES_x is a flow coefficient indicative of the degree of connectivity between polygons (a) and (b), and $A_{(a)}$ is the area of polygon (a). Flows of sediment-poor water from urban runoff were simulated as inputs into the Trapagnier, LaBranche, Walker, and Piquant polygons through flood-control pumping stations (RES_3, RES_4, RES_5, and RES_6) as a function of RUNOFF, the pumping capacity of the station (RES_x), and the urban drainage area ($Area_p$) according to the following:

$$Flow_p = RUNOFF \cdot RES_x \cdot Area_p$$

Habitat succession in the LaBranche PBS model was simulated with a population sub-model such that each polygon "evolved" separately as a function of elevation. We choose this approach because polygons were large and composed of multiple habitats. The original May and McArthur (1972) equations described how the niches of species can overlap in a fluctuating environment. We use the same principle in the form of landscape elements (i.e., swamp, marsh and open water) to explain community overlap and landscape succession in each polygon. Each landscape element had a preferred position in a one-dimensional resource spectrum which we set as the area flood duration. The flood durations were actually land elevations relative to the average tidal fluctuations for the adjacent Lake Pontchartrain, which vary between 0 and 60 cm (Pierce et al. 1985).

To estimate the preferred position of each landscape element along the 0 to 60 resource spectrum (elevation), the percent habitat distributions within each polygon were calculated from remotely sensed images taken in 1952, 1956, 1965, 1972, and 1978 (Pierce et al. 1985) and compared with the theoretical relationships predicted by the following competitive exclusion equations of May and McArthur (1972):

$$\frac{d(S)}{d(e)} = \int_0^{60} 1.7\, S_{(e)} - S_{(e)}^2 - .4\, S_{(e)} M_{(e)} - .9\, S_{(e)} W_{(e)}$$

$$\frac{d(M)}{d(e)} = \int_0^{60} 1.8\, M_{(e)} - M_{(e)}^2 - .8\, S_{(e)} M_{(e)} W_{(e)}$$

$$\frac{d(W)}{d(e)} = \int_0^{60} 2\, W_{(e)} - W_{(e)}^2 - .7\, W_{(e)} S_{(e)} - .74\, W_{(e)} M_{(e)}$$

where:

e = land elevation as cm from 0 to 60.

S = percent swamp area.
W = percent open water area.
M = percent marsh area.

In the May-MacArthur model, changes in the percent dominance of one habitat occurs at the expense of the others. To estimate these relationships we plotted actual habitat areas, based on the remote imagery (Pierce et al. 1985) and theoretical habitat areas, based on the equations above, as a function of relative elevation (Figure 2.4). The theoretical habitat proportions in each cell were calculated iteratively from elevations of 0 to 60 cm. At an elevation (EL) of 60 the stable solution of the May-MacArthur equations produced a polygon of water, marsh, and swamp which was dominated by swamp (1%, 5%, and 94%, respectively). At the minimum elevation of 0, the composition of water, marsh, and swamp was just the opposite (94%, 5%, and 1%, respectively). Proportions were converted to hectares in each cell based upon the size of each cell.

The maximum correspondence between the theoretical and actual habitat distributions were found by adjusting the coefficients of the May-MacArthur equations and by moving the actual habitat data sets (i.e., three data points per polygon) across the x-axis of Figure 2.4. until the greatest degree of correspondence between the two were obtained. The degree of correspondence was calculated as the minimum sum of squares (MSS), where a MSS of zero denotes perfect correspondence as discussed by (Steel and Torrie 1960). The end result was the set of theoretical curves relating habitat distributions to elevation shown in Figure 2.4 with a MSS of 0.03.

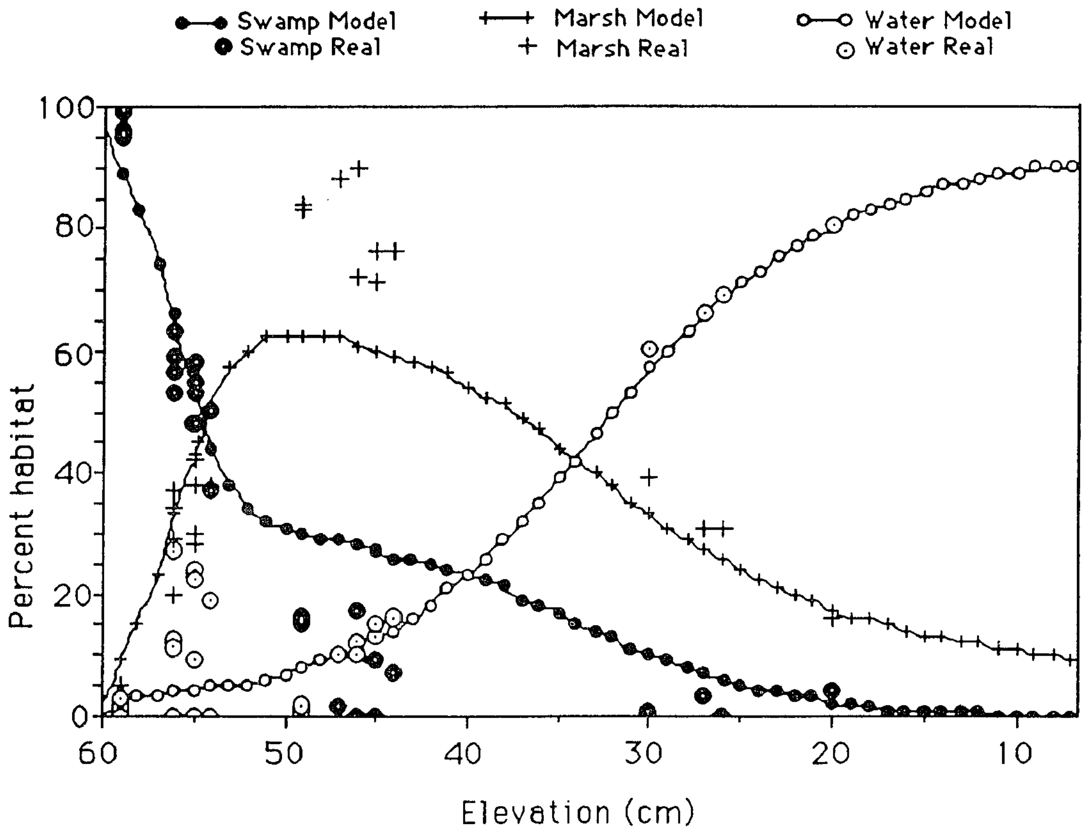

Figure 2.4. The population model of May and MacArthur (1972) was modified to simulate the distribution of swamp, marsh, and open water habitats within a polygon, as a function of wetland elevation. The equations for the three habitat curves are given in the text. The "real" data points for swamp, marsh and water percentages are the values measured for each polygon in 1952, 1956, 1965, 1972, and 1978 by Pierce et al. (1985). Zero on the x axis equals mean low water.

Elevation, in turn, was modeled as a function of subsidence, sedimentation, and biological production according to the following equation:

$$\frac{d\,EL_{(c)}}{dt} = EBM_{(c)} + ESS_{(c)} - \text{Subsidence}$$

where:

$EL_{(c)}$ = Relative elevation (cm) in cell c.

$EBM_{(c)}$ = Contribution of organic matter to the maintenance of relative elevation (cm) in cell c.

$ESS_{(c)}$ = Contribution of suspended sediments to the maintenance of relative elevation in cell c (equals $Ksed \cdot SS_{(c)}$).

Subsidence = Decrease in elevation (0.00587 cm/day) relative to average see level due to:

1) Dewatering and compaction of deltaic sediments (Berner 1980),
2) Subsidence tectonics through sediment loading (Kenneth 1982),
3) Eustatic sea level rise (Nummedal 1983).

The elevation equation responds to community composition as a synergistic loop where primary production increases with elevation and elevation increases with organic deposition. If this were the only synergism however, it wouldn't be long before all Louisiana wetlands rose out of the mud to become hardwood forests. Succession to forested uplands is not occurring in Louisiana wetlands because organic deposition alone can not compensate for subsidence, decomposition, and compaction (Leibowitz et al. 1988). Marshes and swamps must capture suspended sediments to enhance the fertility of the sediments and adjust for erosion and subsidence (Baumann et al. 1984).

The deposition of suspended sediments (ESS) and the burial of organic matter (EBM) were designed as synergistic processes such that land loss tends to increase as elevations begin to decrease (i.e., dEL/dt negative). Organic matter contributions to the elevation of a polygon according to the equation:

$$\frac{d\,EBM_{(c)}}{dt} = S_{(c)}P_{(s)} + M_{(c)}P_{(m)} + W_{(c)}P_{(w)}$$

where:

$EBM_{(c)}$ = Contribution of organic matter to the maintenance of relative elevation(cm) in cell c.

$S_{(c)}$; $M_{(c)}$; $S_{(c)}$;. = Areal extent (m^2) of swamp, marsh, and water, respectively, in polygon c.

$P_{(s)}$; $P_{(m)}$; $P_{(w)}$;. = Coefficients for the accumulation of soil due to primary production ($cm/m^2/d$) in swamps (.003), marshes (.0048), and open water habitats (.0001), respectively.

The P values were based on the swamp production data of (Conner 1986), the marsh production data of Cramer et al. (1978), and the relationship between bio-productivity and elevation according to (Gosselink et al. 1984).

Relatively little suspended sediments enter the marsh or swamps via runoff because the entire LaBranche study area is semi-impounded with a closed levee on the upland side. Although urban runoff can contain significant quantities of suspended sediments, we assume that most, if not all, of the inorganic sediments available for sedimentation in the LaBranche Wetlands comes from Lake Pontchartrain and that the exchange of suspended sediments as $mg \bullet l^{-1}$ is proportional to the hydrologic flows according to the following.

$$\frac{d(ss)_c}{d(t)} = \sum_{c=1}^{m} \frac{F_i}{V_i} \bullet SS_i - \sum_{i=1}^{m} \frac{F_o}{V_o} \bullet SS_o + SS_{c,t=-1} - ESS_c$$

where:

$(ss)_c$ = Concentration of inorganic suspended sediments ($mg \bullet L^{-1}$) in polygon c at time t.

m = The number of polygons that share an interface with polygon c.

F_i = The input of water into polygon c as m^3 per d(t).

V_i = The water volume (m^3) of each polygon with flow (F_i) into polygon c.

SS_i	=	Suspended sediment concentration ($mg \cdot L^{-1}$) of each polygon with flow (F_i) into polygon c.
F_o	=	The output of water from polygon c to all m polygons as m^3 per d(t).
V_o	=	The water volume (m^3) of polygon c.
SS_o	=	Suspended sediment concentration ($mg \cdot L^{-1}$) of polygon c with flow (F_o) to polygons i to m.

The case when suspended sediments enter the study area from pump water or Lake Pontchartrain was calculated as $F_i \cdot kss_{pump}$ and $F_i \cdot kss_{lake}$, respectively. Where, kss_{pump} and kss_{lake} are constants (0.008 g/l and 0.12 g/l, respectively).

RESULTS

WATER: The hydrologic forcing functions RUNOFF and LAKE (Figure. 2.3) were repeated annually producing a steady state water level variation in each polygon. The impact of this hydrologic averaging on the water levels in each polygon are shown in Figure 2.5. Seasonal water level variations were stable over the 28 years that the PBS model was run. However, differences between areas were significant. The Walker polygon had relatively large annual water level variations (14.9 ± 5.5 cm). The Marksville polygon had the smallest water level variations (7.2 ± 1.8 cm). The Trapagnier, LaBranche and Piquant polygons were all similar with moderate variations (10.7 ± 3.4, 10.0 ± 3.1, and 10.6 ± 3.4 cm, respectively).

SEDIMENTS: The average seasonal deposition of suspended sediments was compared with average suspended sediment concentrations over the course of the 28-year PBS simulation (Figure 2.6) and as expected, greater suspended sediment loads leads to larger depositions. Although suspended sediment loads can have a relatively rapid seasonal oscillation within an annual cycle, the rate of deposition tends to be constrained (i.e., is a more

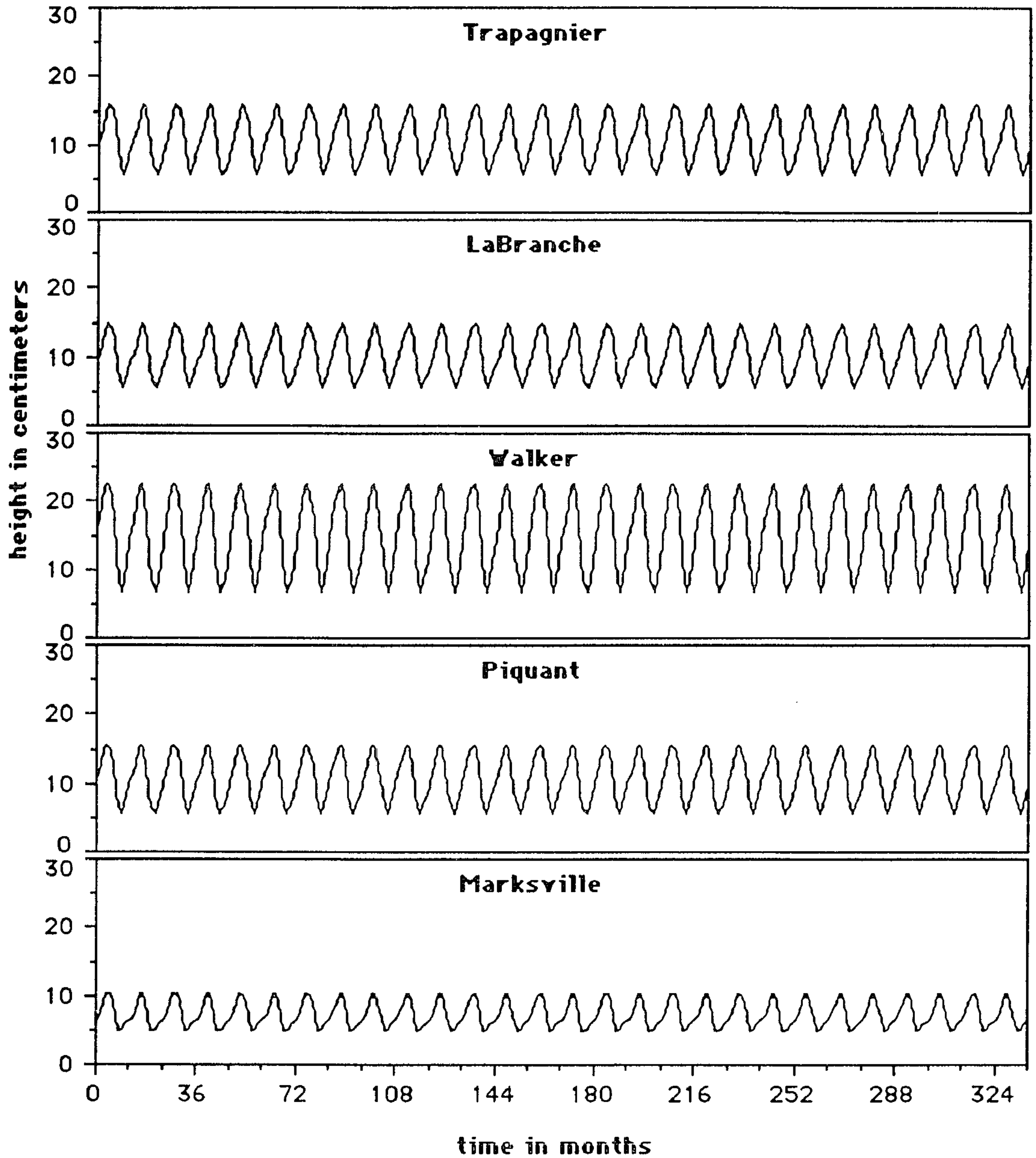

Figure 2.5. The flow of water between polygons in combination with RUNOFF (excess rainfall) and Lake (average water level fluctuations in Lake Pontchartrain) produce seasonal water level changes that are stable within a polygon but which produce large variations among polygons.

moderated process) by the lack of suspended sediments during Summer and Fall and high flow rates (resuspension and erosion) during Winter and Spring.

We conducted a sensitivity analysis to see how the seasonal variation of suspended sediment concentrations (SS), after deposition, varied as a function of changing sedimentation rates and found that the rate of sedimentation had different impacts within different polygons (Figure 2.7A). A high Ksed of 1.0 cm/g/l resulted in suspended sediment loads of zero in all polygons except

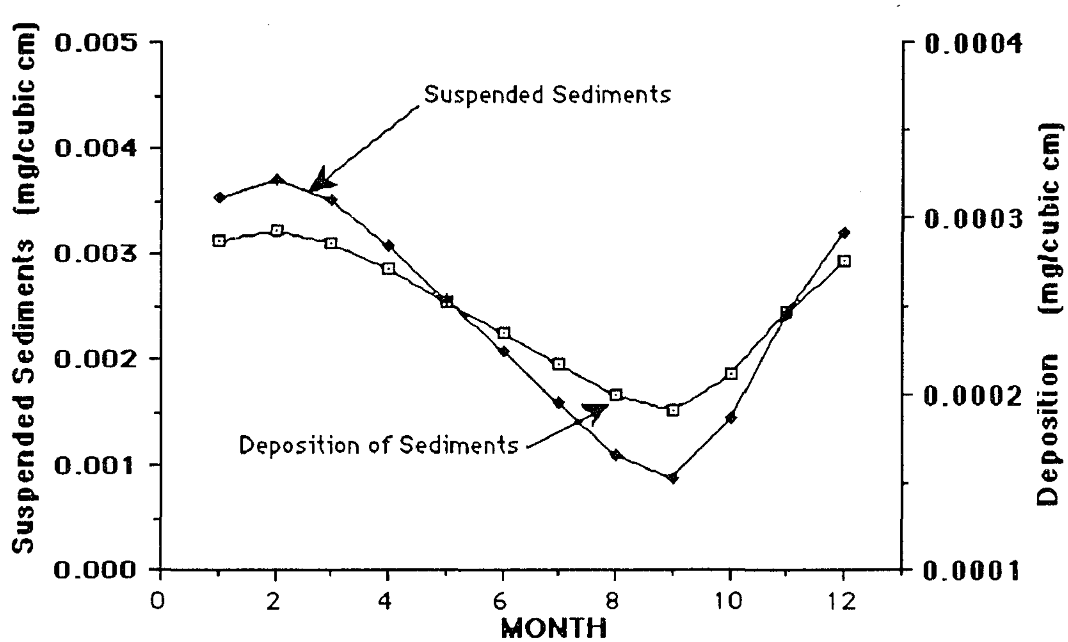

Figure 2.6. Average concentration of suspended sediments and the rate of deposition of these sediments is seasonal with maximum deposition occurring in the spring (month 2= Feb., month 4= April, etc.).

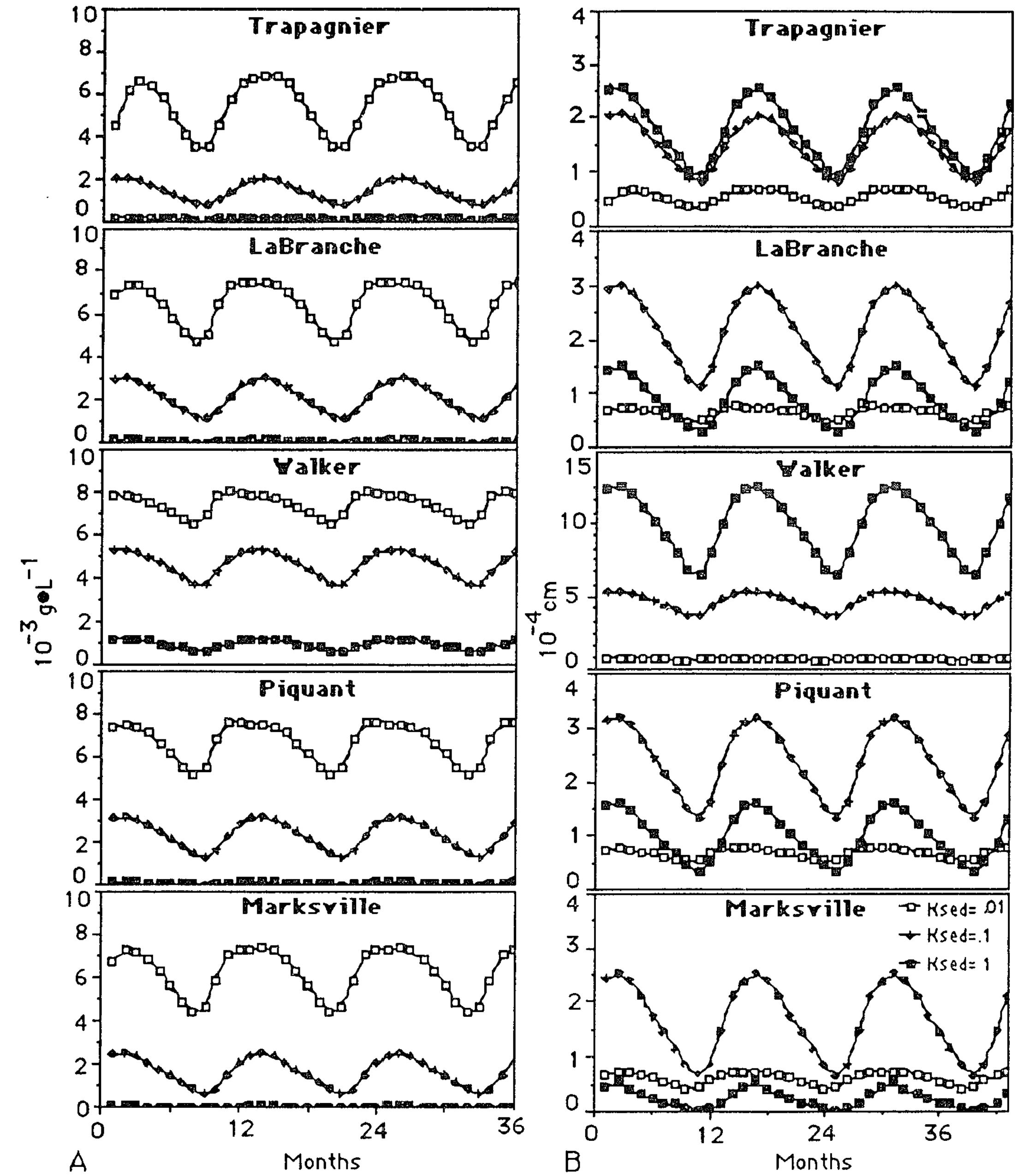

Figure 2.7. Sensitivity of the PBS model to changing Ksed, the sedimentation parameter. (A) Daily suspended sediment loads, after deposition, during 3-year runs of the PBS model. (B) Daily deposition of suspended sediments, as elevation measurements, during 3-year runs of the PBS model.

the Walker polygon. A low Ksed parameter of 0.01 cm/g/l led to high suspended sediment loads in all polygons, with the highest daily concentrations found in the Walker polygon (0.008 g/l) and the lowest suspended sediment concentrations found in the Trapagnier polygon (0.0035 g/l). With a Ksed value of 0.1 cm/g/l, the value used to run the PBS model for 28 years, the daily suspended sediment concentrations varied from a maximum of 0.0053 g/l in the Walker polygon to a minimum of 0.0007 g/l in the Marksville polygon.

The contribution of suspended sediments to the maintenance of wetland elevations (i.e., accretion) was estimated by multiplying Ksed by the daily changes in suspended sediment concentrations (Figure 2.7B). A comparison of the model's response to different Ksed values indicated that the most accretion occurred in the Walker polygon when sedimentation Ksed, was set at the maximum value of 1.0 cm/g/l. The accretion in the Walker polygon varied seasonally, going from a high of 12.7 x 10^{-4} cm/d during winter to a low of 6.4 x 10^{-4} cm/d during summer. Interestingly, with a high Ksed the least amount of accretion occurred in Marksville when in reality, we have observed high sedimentation potential in this area as evidenced by the formation of several flood deltas. This difference occurs because the PBS model is only simulating the past net land loss trends while current trends appear to be reversing due to increasing lake water levels and sediment inputs. When the Ksed value was set very low (i.e., 0.01 cm/g/l), the higher values for deposition (0.8 x 10^{-4} cm/d) occurred in the Piquant polygon while the lower values occurred in Trapagnier (0.3 x 10^{-4} cm/d). Our selection of a final Ksed value of 0.1 cm/g/l resulted in moderate accretion rates with seasonal deposition ranges between 5.3 x 10^{-4} cm/d and 3.7 x 10^{-4} cm/d in the Walker, Piquant and Marksville polygons, and between 2.1 x 10^{-4} cm/d and 0.8 x 10^{-4} cm/d in the Trapagnier and LaBranche polygons.

The PBS model predicted a significant decline in elevations for each polygon after 28 years (Figure 2.8). The rate of the decline in wetland elevations was found to be inversely related to the degree of water level fluctuations. The Spearman Rank Correlation Coefficient between water level fluctuations and elevations was -0.90 with a significance level of 0.05 (one-tailed test). The Walker polygon had the greatest water level variations and the smallest decrease in wetland elevations (16.9 cm/28 years) while the Marksville polygon had the smallest water level variations and the largest decrease in wetland elevation (31.2 cm/28 years). The Trapagnier, LaBranche and Piquant polygons had moderate decreases in wetland elevations in comparison to the other two sites (27.0, 20.9, and 25.7 cm/28 years, respectively) but were still sensitive to the degree of water level fluctuations.

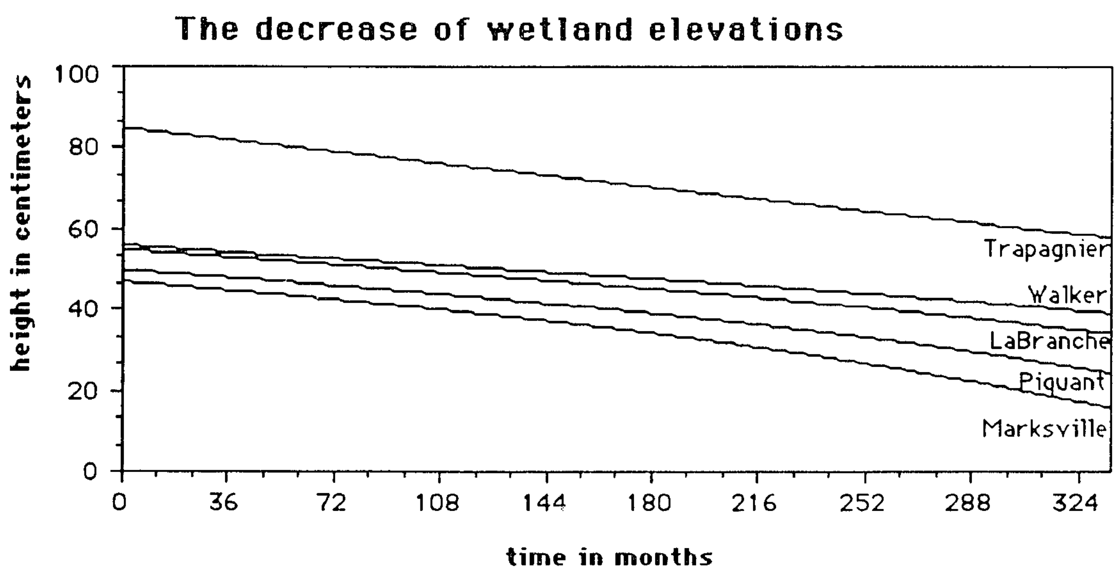

Figure 2.8. Decrease in wetland elevations for each of the five polygons simulated by the PBS model from 1952 to 1980 (daily time steps over a 28-year period).

The wetland elevations decreased in each polygon as a linear function of time (Table 2.2). The slopes of the regression lines

indicate the rates of land loss in each polygon (Elevation) and the relative deposition rates of organic matter (EBM). Table 2.2 also summarizes the long term elevation trends predicted by the PBS model as a function of three different Ksed values. It is significant to note that for a degrading wetland the PBS model predicts very little build up of land as a function of biomass accumulation (EBM). Only the LaBranche and Walker polygons showed a positive contribution to wetland elevation due to bio-productivity. According to the PBS model, the Piquant and Marksville polygons have been slowly losing their ability to produce organic matter.

Although the Trapagnier polygon, with its high elevations and its high percentage of swamp habitat, managed to maintain a steady input of organic matter (i.e. $3.1 \cdot 10^{-3}$ cm/day), it was insufficient to keep up with subsidence. The slope of the land loss curve for Trapagnier (-0.081) was greater than for any other polygon. Suspended sediments reach Trapagnier via pump water but this water is relatively devoid of sediments. Increasing the sedimentation parameter (Ksed) had little effect upon the land loss slopes shown in Table 2.2. This implies that there must be an increase in sediment loading if Trapagnier is to reverse the current trend.

SUCCESSION: Each polygon in the PBS model was initialized with a different mix of habitat types based upon differences seen in aerial images (Pierce et al. 1985). These initial values for swamp, marsh, and water can be seen at year one in Figure 2.9. The predicted habitat changes in each polygon indicate which areas were most susceptible to decreasing wetland elevations, suspended sediment supplies, and organic matter accumulations (Figure 2.9). Trapagnier, the cell with the most swamp area, was stable for 25 years. After 25 years the wetland elevation of Trapagnier dropped rapidly and was affected by the water level fluctuations in Lake Pontchartrain. At this point in time swamps began to flood and were replaced by marsh. This replacement by marsh

proceeded very rapidly. It took only three years for the simulated swamp area to decrease by 25 percent. Although this is probably an unrealistic rate of decline for any individual tree it was realistic at this landscape level model because we assumed that loss of viability was sufficient for reclassification (i.e., persistent flooding prevents cypress regeneration), that time-delay mortality was unimportant (i.e., it was not a component of the May-MacArthur equations) and that landscapes are sensitive to critical threshold values (i.e., catastrophe theory).

Table 2.2. Linear regressions of 3-year runs of the PBS model showing the change in wetland elevations and the buildup of wetland elevations due to biomass accumulation in the sediments as a function of three sedimentation parameter values.

Polygon	Ksed	Elevation	EBM*		
		intercept	Slope	intercept	Slope
		(cm)	(cm/d)	(cm)	(cm/d)
LaBranche	0.01	54.98	-0.058	3.8•10-3	3.0•10-6
	0.1	55	-0.053	3.8•10-3	3.0•10-6
	1	54.99	-0.057	3.8•10-3	3.0•10-6
Walker	0.01	55.97	-0.059	3.7•10-3	5.8•10-6
	0.1	55.99	-0.048	3.7•10-3	5.5•10-6
	1	56	-0.033	3.7•10-3	4.9•10-6
Piquant	0.01	50	-0.059	3.9•10-3	-2.3•10-6
	0.1	50.02	-0.054	3.9•10-3	-2.1•10-6
	1	50.02	-0.058	3.9•10-3	-2.2•10-6
Marksville	0.01	47.01	-0.063	3.8•10-3	-3.6•10-6
	0.1	47.02	-0.06	3.8•10-3	-3.4•10-6
	1	47.02	-0.064	3.8•10-3	-3.6•10-6

*EBM = Wetland elevation buildup (accretion) due to biomass production.

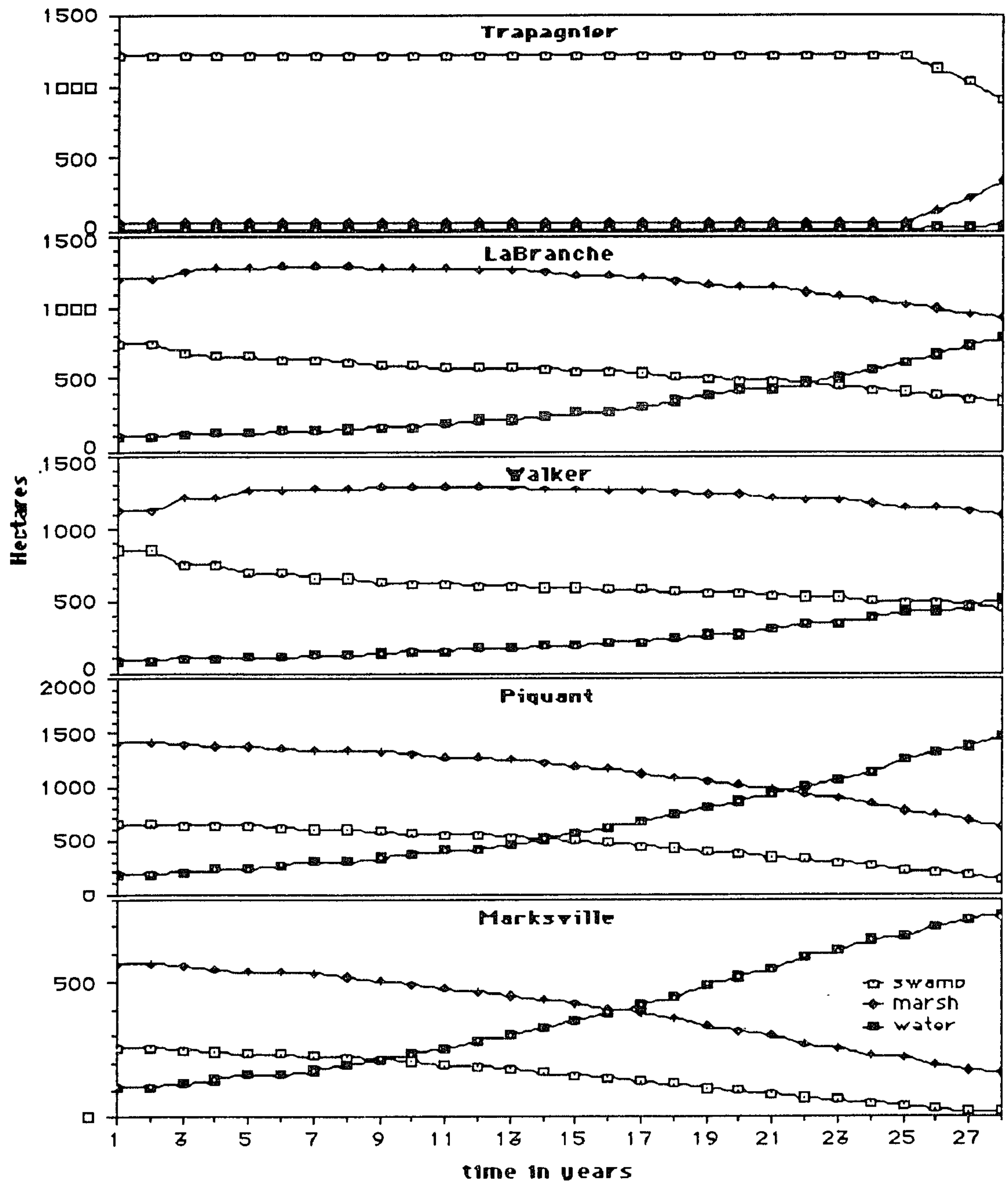

Figure 2.9. The change in community structure in each polygon as predicted by the PBS model from 1952 (year one on the x-axis) to 1980.

The Walker and LaBranche polygons were the only areas to show an initial increase in simulated marsh area. During the first 6 years of the simulation the amount of marsh hectares in LaBranche increased from 1209 ha to 1289 ha. After these first 6 years the marsh area dropped from 1289 ha to 925 ha while simulated swamp area dropped from 748 ha to 344 ha and water area increased from 100 ha to 789 ha. During the first 9 years of the simulation the amount of marsh in the Walker polygon increased from 1119 ha to 1294 ha, after which time the marsh area dropped from 1294 ha to 1095 ha, the swamp area dropped from 856 ha to 454 ha, and the amount of open water increased from 92 ha to 518 ha.

The most dramatic changes in habitat distributions within any of the polygons predicted by the PBS model occurred in Piquant and Marksville. There was a steady decrease in swamp and marsh area and a very steep increase of water area. After 28 years of simulation, the swamp area in Piquant dropped from 660 ha to 146 ha, the marsh area decreased from 1409 ha to 624 ha, and water area increased from 191 ha to 1490 ha. In the Marksville polygon, swamp decreased from 255 ha to 20 ha, marsh dropped from 564 ha to 166 ha, and open water increased from 112 ha to 746 ha.

DISCUSSION

The PBS model was designed to examine marsh succession at the landscape level and to evaluate the utility of the polygon design for landscape simulation. For what it was designed for, the PBS model does a reasonably good, but not an outstanding job. A regression analysis, used as a measure of the goodness-of-fit between predicted and actual habitat change ((Costanza and Sklar 1985), resulted in an r^2 of 0.56 with a $p<0.0001$. Decomposed into its separate habitat components, the r^2 indicated that the PBS model

could account for 62.6%, 50%, and 41.9% of the spatial variation over time for swamp, marsh, and open water, respectively.

Pearson product-moment correlations, in terms of hectares, suggest that the PBS model captured the observed temporal habitat and land loss trends (Table 2.3). For each habitat type, large positive correlations were found between the number of hectares simulated and the number of hectares observed. There were significant negative correlations between swamp and water hectares indicating that land loss or land gain in St. Charles is a dynamic exchange between swamp habitats and open water areas. The PBS model also found that swamps can appear relatively stable for long periods of time, when they could in fact be experiencing degrading environmental conditions which can result in rapid habitat changes after a tolerance threshold is exceeded. Has the threshold been reached for this landscape? The negative correlation matrix implies that it has. Personal observations in 1988 indicate that hardwood swamps in this area, especially in the Trapagnier polygon, are being quickly replaced by marsh and open water habitats. However, photo interpretation of recent aerial photographs is necessary to validate the predicted forest decline.

The problem with this PBS model is its simplicity. This model mostly describes general ecological processes for simple spatial distributions and is not well equipped for predicting short term occurrences. The use of a constant Ksed for the entire area does not reflect the true spatial dynamics of deposition and erosion, and although further study is necessary, Ksed should be made dependent upon habitat distributions and vegetation density. Other problems include the use of generalized forcing functions and a one parameter (i.e. elevation) determination for habitat succession. The May-MacArthur model of succession appears to over-simplify a complex processes and should be enhanced to include more variables. Other improvements should include measurements of elevation gradients in each polygon to better

relate elevations to flood duration and synoptic measurements of sediment concentrations in the water to validate assumptions on suspended sediment loads.

Table 2.3. Correlations matrix comparing hectares of swamp, marsh, and open water habitats as predicted by the PBS model and as measured by Pierce et al. (1985).

	PBS Swamp	PBS Marsh	PBS Water	Swamp	Marsh	Water
PBS Swamp	1					
PBS Marsh	-0.184	1				
PBS Water	-0.736	0.054	1			
Swamp	0.754	-0.111	-0.518	1		
Marsh	-0.314	0.723	0.341	-0.075	1	
Water	-0.747	0.081	0.691	-0.555	-0.413	1

Despite all its shortcomings, a polygon-based simulation model has some real validity and utility. The regression data indicated that the PBS model for the St. Charles wetland landscape simulated the subsidence, accretion, wetland productivity, and habitat dynamics reasonably well. As a result, we can discuss the significant ecological patterns which emerged. That is, accretion from suspended sediment deposition is proportionally more important in those areas where biomass deposition is lowest (Table 2.2), suspended sediment deposition is nonlinear (Figure 2.6), and habitat change is a function of spatial differences in the apparent subsidence rates (Figure 2.9). These simulated relations shed light on what we observe in nature. For years, local fishermen and hunters have noticed that the marsh plants in the Marksville area died very early in the fall while other marshes stayed green through December. According to The Soil Conservation Service (1987), this phenomenon is the result of salt water intrusion. However, why is salt water intrusion occurring? According to the PBS model, the Marksville polygon sedimentation rates are not

keeping up with subsidence (increasing Ksed had little impact). This means that during the Fall dry period, when runoff is at a minimum (Figure 2.3), the hydrologic head on the marsh (i.e. elevation) is not maintaining the height needed to keep brackish lake water (Lake Pontchartrain increases in salinity each Fall, Stone and Hinchee 1980) from moving up into the marshes. To compensate for salt water intrusion there must be more pumping of urban and river waters to increase fresh water, or there must be more deposition to increase the elevation. There is a project now to increase the deposition factor of the suspended sediments by building brush fences (Kamps 1962; Verhoeven and Akkerman 1967; Boumans et al. 1987; Day et al. 1988) on mud flats and by planting marsh grasses (Woodhouse et al. 1974; Joenje 1978; Knudson and Woodhouse 1983; Webb et al. 1984). It will be interesting to see if these techniques can reverse current trends.

Our model found little movement of suspended sediments out of the lake and into the study area. Suspended sediment concentrations followed the upland runoff function and not the function simulating the lake water head. During the 28-year runs of the PBS model the elevations in each of the polygons stayed high enough to keep lake water out of the area. In reality, wind induced tides make the lake water head fluctuate around the average lake water level (Stone and Hinchee 1980; Wang and Scarlatos 1982; Baumann 1987), so that during high winds and high tides the lake water has more of a chance to enter the area. Although the incorporation of wind into our model would increase its realism and its validity, it was not included because the positive effects of wind (i.e., the resuspension and re-deposition of sediments up onto higher ground, (Baumann et al. 1984)), is to a degree balanced by the negative effects of wind (i.e., the erosion and export of sediments downwind, (Mentha 1984)). Even with wind induced tidal inundations it appears from our results that the polygons dominated by swamps (Trapagnier and LaBranche) will continue to degrade because they are either too far from Lake

Pontchartrain or they are cut off from the lake due to man-made impediments (e.g., spoil levees and railroads).

The elevations decreased at different rates in the different polygons. The polygons with the most marsh area (LaBranche and Walker) decreased the least. The polygons with the most water area (Piquant and Marksville) decreased the most. The marsh dominated polygons were able to compensate for subsidence better than the other areas because the higher productivities of marsh grasses added to the elevation as organic deposition (EBM). For this marsh dominated landscape, the correlation between elevation change and biomass deposition (EBM), in three year runs of the model, was found to be more significant ($r=0.825$, $p<0.0001$) than the correlation between elevation change and elevation built by suspended sediments (ESS) ($r=0.681$, $p<0.03$). It seems that as marshes disappear the landscape's ability to compensate for subsidence is significantly reduced.

Elevation change was also related to water level fluctuations. The greater the annual water level variation the smaller the decline in the elevation curve (Figure 2.8). This is consistent with the subsidy-stress hypothesis of Odum et al. (1979) which states that the benefits of energetic processes such as tides, waves, and sunlight to an ecosystem can be plotted as a normal distribution. Maximum subsidy occurs in the mid range of the bell-shaped curve while stress occurs at the two extremes. Impoundments, levees, and roads decrease the annual water level variation (Swenson and Turner 1987; Wang 1987). Decreasing the hydrologic subsidies by partial or complete impoundment of wetlands has lowered tree growth (Conner et al. 1981) and animal diversity (Sklar 1983) in other Louisiana wetlands.

The PBS model is a potential tool for the exploration of generalized wetlands processes at the landscape level. Simulating long term trends using a polygon-based spatial model, has management and

ecological utility because the model can address cumulative impacts (Dijkema et al. 1985; Gosselink and Lee 1987). Point and non-point forcing functions (i.e., perturbations) can be easily incorporated to evaluate impacts and management plans at a regional scale.

There are only a few spatial models of ecosystem processes (Costanza and Sklar 1985). The complexity associated with simulating both spatial and temporal trends with a mechanistic model has deterred its development. Our own experience with the CELSS model of (Dijkema et al. 1985; Sklar et al. 1989) is a case in point. The CELSS model has over 2400 cells, 2000 lines of programming code, and over 100 parameters controlling only eight state variables. In general, an increase in the number of cells and parameters will increase the degree of uncertainty if the data requirements are not met. On the other hand, the PBS model has only 130 lines of code, five polygons and 20 parameters controlling five state variables (i.e. water, suspended sediments, primary production, habitat type, and elevation). The PBS model required a small desktop computer. The CELSS model required a Cray II super computer. The difference was a better goodness-of-fit between the real and simulated data for the CELSS model (Costanza et al. 1986) than for the PBS model (0.89 and 0.56, respectively). The trade-off appears to be more rapid simulation time at the expense of model accuracy and realism. Although the correlation and regression results for the PBS model were not as high as those reported for the CELSS model they did however, indicate that the long term trends and spatial ecological processes in a wetland can be captured by a relatively simple program based on exchanges across polygons. This is a significant finding because it opens the door to the development of dynamic GIS (geographic information systems).

GIS's are rapidly developing throughout the world at local, regional, and global scales. They are polygon-based spatial

databases which, theoretically, could be directly accessed by a dynamic simulation model to predict spatial changes. We believe that the true potential of a GIS will be fully realized when it is somehow linked to a simulation system. The PBS model is only a first step in this new direction.

CHAPTER 3

EFFECTS OF TWO LOUISIANA MARSH MANAGEMENT PLANS ON WATER AND MATERIALS FLUX AND SHORT TERM SEDIMENTATION

Boumans R.M.J. and J.W. Day. Effects of Two Louisiana Marsh Management Plans on Water and Materials Flux and Short Term Sedimentation.

In press in "Wetlands"

INTRODUCTION

For over three decades, one of the most intensively studied topics in coastal ecology has been the interaction between tidal wetlands and adjacent estuarine waters (Odum and Cruz 1967; Gardner and Kitchens 1977; Odum et al. 1979; Kjerfve and McKellar 1980; Nixon 1980; Dame et al. 1986; Stevenson et al. 1988; Day et al. 1989). There has been considerable controversy and discussion over the exact role of coastal wetlands, but undoubtedly there are large and important exchanges of water, suspended sediments, inorganic nutrients, organic matter and living organisms between wetlands and estuarine waters. It has been shown, for example, that some coastal marshes export organic matter and provide habitat for estuarine nekton (Nixon 1980; Day et al. 1989). In coastal Louisiana, input of suspended sediment to wetlands is important to offsetting the high levels of relative sea level rise (Baumann et al. 1984). An important question for coastal managers, therefore, is the impact of management practices on such interactions between coastal wetlands and waters.

A practice which is widely used in the Louisiana coastal zone is marsh management. As used in Louisiana, the term marsh management refers to the use of low dikes and water control structures to create impoundments in order to manipulate inside hydrology. Such marsh management has been practiced for decades in Louisiana to improve marshes for waterfowl and fur bearers. More recently, marsh management has been promoted as a way of combating the high rates of coastal wetland loss (Cahoon and Groat 1990) based on the belief that salinity intrusion and tidal scour were the most important agent leading to land loss (Gagliano and Wicker 1989). At present management impoundments account for 17% of the Louisiana coastal zone (Day et al. 1990).

Control structures include fixed crest weirs, variable-crest weirs and gated weirs and culverts. Fixed crest weirs provide passive control on water levels disallowing the water level to drop under crest elevation. The anticipated benefits include boat access during low tide and prevention of saltwater influx. The adjustable weirs, flap gated weirs and culverts used are more active. Adjustable weirs allows manipulation of the inside water levels because crest elevation can be adjusted. Flap gated weirs and culverts provide means to drain an area at low tide and prevent water inflow at high tide. Hydrostatic pressure opens the gates at higher water levels inside than outside, and closes them when water levels are reversed (Clark and Hartman 1990).

Recently questions have been raised about marsh management concerning both the assumptions underlying justification of use (i.e. the role of saltwater intrusion and tidal scour in wetland loss) and the impact of marsh management on estuarine processes such as flux of materials and sediment, and access by estuarine dependent species (Cowan et al. 1988; Day and Templet 1989; Cahoon and Groat 1990; Herke et al. 1992). Our objectives in this study were to measure the effects of two marsh management plans on water and materials fluxes, short term sedimentation, and soil parameters under different weather, tidal and management conditions.

STUDY SITE

This study was part of a larger, comprehensive evaluation of marsh management practices in the Louisiana coastal zone (Cahoon and Groat 1990). As part of the larger study, 16 areas were selected for the analysis of habitat change from the mid 1950's to the mid 1980's using aerial imagery. The areas were selected by a steering committee composed of landowners, state and federal agency representatives, marsh managers and scientists. Each area included an operational marsh management plan typical of coastal

Louisiana and a nearby non-managed area of similar size, marsh type, degree of hydrologic alteration and degree of wetland deterioration. The 16 management plans included both actively and passively managed marshes. We report only on two of the 16 areas, Rockefeller State Wildlife Refuge (Rockefeller) and the Fina-Laterre Marsh Management Area (Fina), as they were chosen for more intensive study of the effects of management on hydrology, materials flux, vegetation ecology, soil parameters, water chemistry, accretion, and nekton (Cahoon and Groat 1990, (Boumans and Day 1990)).

Fina and Rockefeller are in two geomorphologically different areas of the Louisiana coastal zone; the deltaic plain and the chenier plain (Figure 3.1, for a more complete description of the area see Cahoon and Groat (1990)). The deltaic plain was formed in an area of Pleistocene erosion followed by direct deposition of Mississippi River sediments. In contrast, the chenier plain was formed by reworking of sediments carried to the area by westward littoral drift and deposited in the near shore zone. In each plain, measurements were carried out in a managed area and a nearby unmanaged area.

The deltaic plain managed area is the Fina (2768 ha) which is located about 40 km from the coast (Figure 3.1). The goals of the management plan, which was begun in 1985, are to reverse wetland loss, increase freshwater and sediment input, and decrease salinity, stabilize water levels, increase marsh production and allow for immigration of marine fishery species. The area is surrounded by natural ridges or low dikes. Four fixed-crest weirs on the northern boundary and one active structure on the southern are used to control water levels. The southern structure consists of two adjacent bays each 2.8 m wide with individual flap-gates located south of variable-crest weirs. The vegetation ranges from fresh water marsh in the north to a brackish Spartina

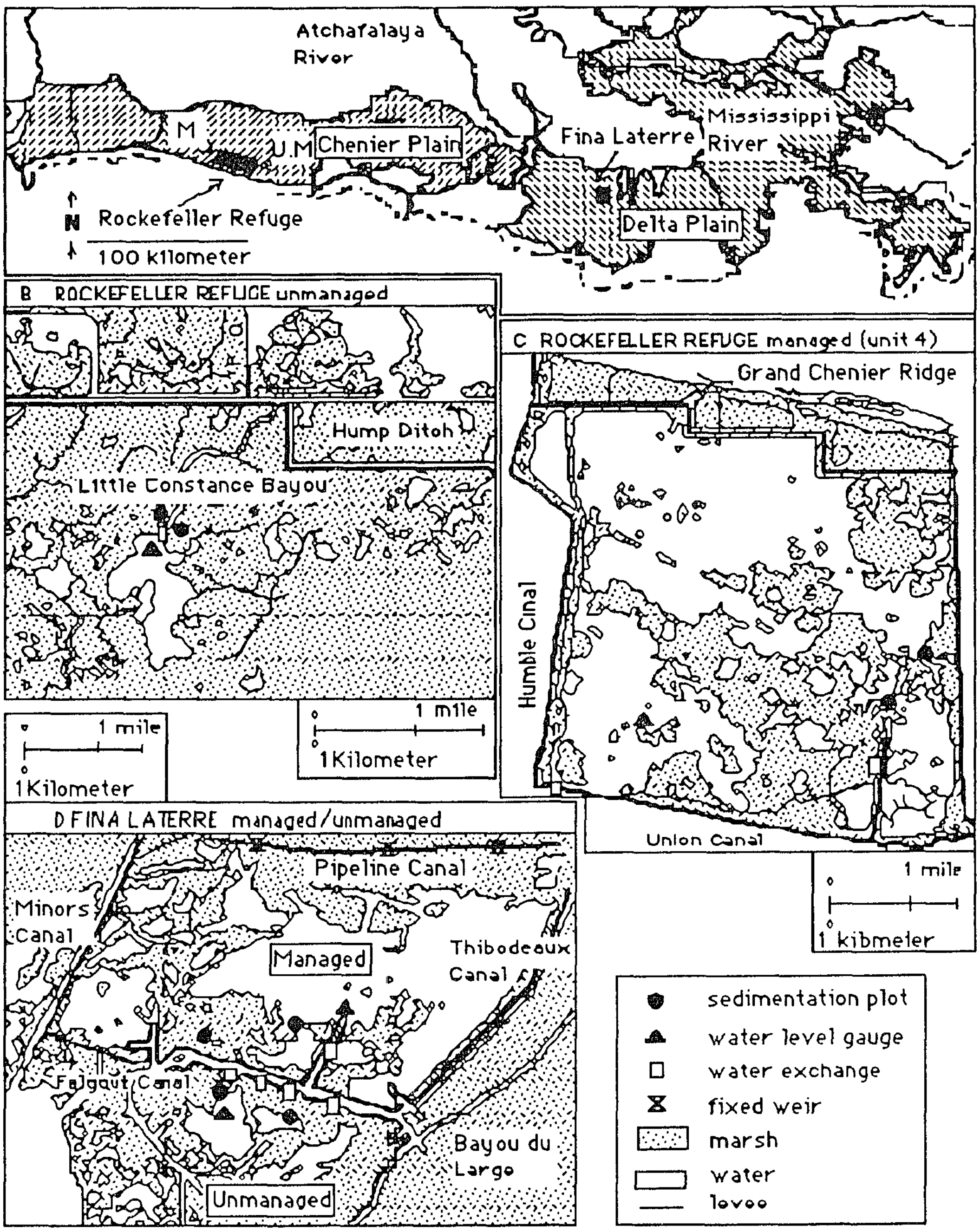

Figure 3.1. Map showing locations of the study sites. The managed and unmanaged areas at Rockefeller are located 15 km apart in the chenier plain. The managed and unmanaged areas at Fina are adjacent in the deltaic plain. Note that Rockefeller is located closer to the coast than Fina.

patens dominated wetland in the south. The unmanaged site (692 ha) is located south of the managed site across Falgout canal.

Water flow is restricted by spoil banks but four openings along Falgout Canal allow water exchange. The dominant vegetation is Spartina patens.

The chenier plain managed site (2084 ha) is Rockefeller (Figure 3.1) which is located about 5 km from the coast. Since the state acquired the refuge in 1920, a program has been undertaken to counteract the effects of saltwater intrusion and altered hydrological regimes. Improving waterfowl habitat and enhancing estuarine fisheries are the primary management objectives. Like Fina, this managed area is bordered on all sides by low dikes. The main water control structure in the southwest corner has seven bays with flap-gated, variable-crest weirs. There is a second structure in the eastern part of the area with five double flap gates. The unmanaged marsh (1604 ha) is 15 km east of the managed area, and borders East Little Constance Bayou. This bayou has direct hydrological exchange with the Gulf of Mexico. The northern border is formed by a canal spoil bank with a variable-crest flap-gated culvert which drains the wetlands north of the study site.

The marsh vegetation in the managed as well as in the unmanaged stations is dominated by Spartina patens, an indicator species for brackish marsh. The managed area is more towards a fresh marsh because of the presence of species like Typha latifolia and Bacopa monnieri. Soils in both the managed and unmanaged areas are classified as Haplaquolls-Hydraquents association.

METHODS

FLUX STUDIES: Measurements of material flux were carried out at the two study areas. Field trips were carried out on the following dates in 1989 to account for seasonal variations in climate, mean sea level and operation of the flap gated structures: Rockefeller - May 31-June 2, September 21-23, and November 2-4; Fina - May 22-24, September 28-30, and October 26-28. At Rockefeller, measurements were carried out at the main control structure in the southwest corner of the managed site, and in East Little Constance Bayou just above Little Constance Lake in the unmanaged site (Figure 3.1). At Fina, measurements were made at the main control structure just north of Falgout Canal and at several inlets to the unmanaged area to the south of the Falgout Canal (Figure 3.1). Measurements of water and materials flux were made each two hours for the duration of each field trip at the control structures, East Little Constance Bayou, and at the largest channel into the Fina unmanaged area. At the four smaller channels at Fina unmanaged, water flux was measured approximately each two hours during daylight to estimate total water exchange between the unmanaged area and Falgout Canal. The cross sectional areas of the channels and control structures were measured. During each trip, field notes were made of weather conditions, structure operation, and other factors which might prove useful in analysis and interpretation of the data.

During each trip water level, and current velocity and direction were measured when samples were taken for the determination of material concentrations at two h intervals for 48-50 hours (2 tidal cycles). At the unmanaged areas samples were collected in the center of the channels at 30-40 cm depth. Five channels (2.40-13 m wide, 0.18-1.22m deep) connect the unmanaged area at Fina to the estuary Only one channel (25.2m wide) connects the unmanaged area in Rockefeller to the estuary. An earlier study of

flux in a similar channel in coastal Louisiana showed that a single sample was sufficient to characterize flux in such a channel (Stern et al. 1986; Stern et al. 1991). At Fina unmanaged, 2 h water samples were taken at the largest and most dominant channel. In addition, water samples were collected with an auto sampler at one additional channel for the analysis of total suspended solids (TSS) only. For the managed areas, current measurements were made and water samples collected from the water flowing through the structures. Routine measurements were made between the gates and the weirs. At several times during each trip, a number of additional current measurements were made north of the weir as well as south of the gates to ensure that measurements in the south bay were sufficient to characterize water flux. At Rockefeller routine measurements were made at the main water control structure in the southwest in the center of bays 2, 4, and 6. Several times during each trip three current measurements were made in each of the seven bays so that total water flux could be calculated.

Water samples were collected for analysis of nitrate plus nitrite (NO_3), ammonium (NH_4), soluble reactive phosphate (SRP), total suspended solids (TSS), and salinity. Samples for NO_3, SRP and NH_4 were filtered through 1.2 μm GF/C filters and frozen on dry ice in plastic auto analyzer vials. All nutrient concentrations were determined according to (Environmental 1979) with a Technion Auto analyzer II. TSS were determined gravimetrically (Banse et al. 1963) and salinity was measured with a chloridity meter.

Current velocity was measured using a Montedoro Whitney, Inc. model PVM-2A current meter. Water levels were read from staff gauges. The staffs were placed in the channels at the unmanaged areas, and inside and outside of the structures at the managed areas. Instantaneous fluxes were calculated from each velocity, cross sectional area and concentration value. Changing cross sectional area due to changing water level was accounted for

(Stern et al. 1986). The net fluxes reported are the average of algebraic sums of instantaneous fluxes for each sampling period at each particular experimental area (Spurrier and Kjerfve 1988). For both instantaneous and net fluxes, negative values refer to transport out of the study area, while positive values refer to transport into the study. Material concentrations and fluxes from unmanaged and managed areas were compared to determine possible effects of management.

SHORT-TERM SEDIMENTATION: Short term sedimentation was measured at two to four week intervals between August 1989 and January 1990 as deposition on petri dishes placed in the marsh at the managed and unmanaged sites. Petri dish collectors were constructed by drilling a hole in a 25•25 cm cedar board slightly larger than a 98 mm diameter glass petri dish. The surface of the board was placed level to the marsh surface. Four galvanized wires about 35 cm long were then pushed through small holes drilled in each corner of the board and into the marsh soil to anchor the board in place. The petri dish was then placed bottom up in the hole in the center of the board over an aluminum wire bent to hold the dish in place as well as to help remove the dish when sampling. At each sampling site 10 petri dishes were set out in two transects of 5 dishes each for a total of 40 dishes (at the see locations, Figure 3.1). Each transect consisted of dishes set out in the marsh at 5, 10, 15, 20 and 25 m from the water's edge. The petri dish technique is a modification of a technique used by (Reed 1989; Reed 1992) who measured short term sedimentation on paper filters wired to the top of plastic petri dishes.

All material which had collected on the surface of a dish collector was placed in a plastic bag. Grass stems and other larger material was picked out by hand, and a razor knife was used to cut grass pieces so that only material which was over the area of the petri dish was sampled. The petri dish was then carefully removed and placed over a funnel. Fine material accumulated on the dish was

scraped off with the razor knife and washed into the plastic bag which was then placed on ice and returned to the lab. In the laboratory, water and collected materials in each bag was placed in a crucible and the water was evaporated at 60 ^{o}C. The crucible was then weighed, heated at 400 ^{o}C for 16 h, and then re weighed. The mass of material after combustion was considered mineral matter and the loss on ignition was organic matter. The data were calculated on a g m^{-2} d^{-1} basis.

SOIL ANALYSES: Sediment samples were taken with a 2.5 cm stainless steel corer at 16 randomly chosen stations in the southern part of the Fina managed area, and in the Fina unmanaged area; and at 20 randomly chosen stations in each of the Rockefeller sites. The upper 15 cm of each sample was homogenized for analysis of total phosphorus, Na^{+}, and % organic matter. Elemental analyses were done using an Applied Research Laboratory inductively coupled Argon plasma quantometer (ICP). Results are reported as the average of duplicate analyses that are within a 10% confidence interval. The results are based on oven dry mass (Soil Survey Staff 1972).

STATISTICAL ANALYSIS: Data for short term sedimentation, soil parameters and water parameters during the flux study were statistically analyzed as a random block design with a 2 by 2 factors treatments arrangement (Appendix A). We tested for differences between management (managed vs. unmanaged), location within the coastal plain (Rockefeller vs. Fina), and also among all individual cells. Where we determined a statistical difference among individual sites we used Tukey's as multiple comparison test. For these tests, we only used the water parameters when the water was flowing out to ensure that we were comparing water masses specific the studied marsh areas.

RESULTS AND DISCUSSION

WATER LEVEL VARIATIONS DURING FLUX STUDIES: The water level variations show a number of interesting results. Outside of the managed areas, we recorded a clear tidal signal for both the Rockefeller and Fina sites, whereas inside the unmanaged areas the water level change was suppressed over the 48-h duration of the flux studies, especially at Rockefeller (Table 3.1, Figure 3.2).

Table 3.1. Tidal range during the three flux studies for each of the study sites.

		Tidal Range (cm)		
Study Site	Measurement*	Spring	Fall	Late Fall
Fina LaTerre	Managed a	5	8	13
	Managed b	17		
	Unmanaged	12	12	14
Rockefeller	Managed a	5	5	3
	Managed b	76	79	74
	Unmanaged	38	16	27

* "Managed a" refers to measurements taken within the water control structure; "Managed b" levels were measured outside the water control structure; "Unmanaged" refers to water levels in the main channel linking the unmanaged marsh to the estuary.

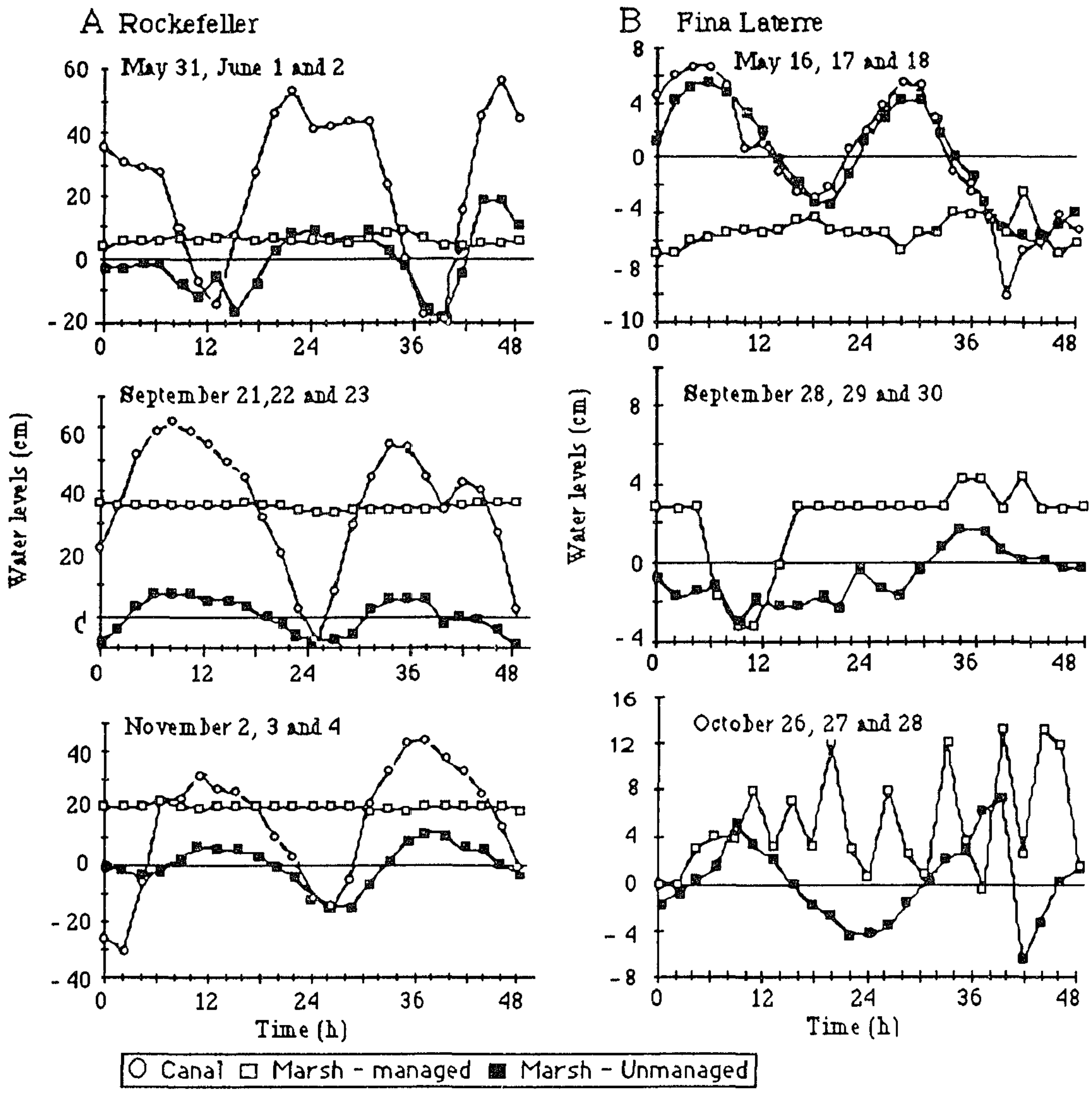

Figure 3.2. Water levels measured at the unmanaged and managed sites during the three tidal exchange studies. A Rockefeller Managed out refers to water levels in the canal outside of the water control structure. The water level scale is relative to an arbitrary datum. Water levels in and out of the managed area are absolute to each other. The curve for the unmanaged area is plotted such that low water coincides with low water for the canal. B. Fina. During the May study, water levels were measured both inside and outside of the water control structure. During the September and October studies, the control structure was open and water levels inside and outside the managed area were the same. The water level scale is relative to an arbitrary datum, but the different curves are absolute to one another.

Fina on October 26-28 experienced water level changes inside the managed area which can not be tied to tidal changes or management, but are possible the result of a sloshing wave with unknown period generated by strong locale winds. Fluctuations in water levels were typical for the northern Gulf, which is dominated by diurnal tides. At Rockefeller, for example, a mixed tide was recorded with both diurnal and semi-diurnal components on September 21-23 and a diurnal tide on November 2-4 (Figure 3.2). The tide range at the two unmanaged sites also were diurnal. The tide range varied from 16 to 38 cm at Rockefeller and from 12 to 14 cm at Fina. This is the expected pattern as tidal amplitude decreases with increased distances from the coast (Baumann 1987).

The flood tide in the canal outside Unit 4 at Rockefeller (Figure 3.2) was amplified because an extensive network of dikes prevented the water entering on the rising tide from spreading out over the marsh. Most of the area in the western part of the refuge is surrounded by levees, in contrast to water in the unmanaged areas, that is not restricted to channels. The tidal range in the canal during the three tidal cycle studies ranged from 74 to 79 cm as compared to 16 to 38 cm in the unmanaged area (Table 3.1). At the Fina site, we did not measure any such amplification of the tidal range (Figure 3.2, Table 3.1). This is likely due to smaller tidal ranges and a much larger unrestricted tidal plain at Fina than in the western part of Rockefeller. The amplified high tide at Rockefeller results in less time per tidal cycle that the main control structure can drain Unit 4. At a non amplified high tide the water level would stay below the marsh surface for more than half of the time. Because the water level in the Humble and Union canals is often higher than the water inside Unit 4, water would flow into the managed area for more time than it would flow out, if the structure were left open at all times. When the water level outside

the structure is below the inside levels, drainage is slow because of the limiting size of the structure openings.

INSTANTANEOUS WATER FLUXES: The water flux measurements show a considerable reduction in total water exchange at the managed sites when compared with the unmanaged sites (Figure 3.3). Instantaneous water flux at the unmanaged sites followed the tidal signal with significant inflows and outflows occurring. Control structures retarded water exchanges in both directions. Inflow was obstructed by the flap gates while outflow was retarded by the limited flow capacity of the structure and altered differentials in water level inside and outside.

At Rockefeller peak fluxes ranged between 4 and 6 $m^3 s^{-1}$ for the unmanaged site and between 1 and 2 $m^3 s^{-1}$ for the managed site (Figure 3. 3). The tidal cycle measurements at the end of May and early June occurred during a period of low winds, and thus water fluxes were mainly tidally driven. At the unmanaged site, the water flux pattern reflected the mixed tide with a considerable flow in and out, but the net flux was relatively low. At the managed site the structure was closed for the entire sampling period except during the final hour when it was opened for a short time to allow post larval shrimp to enter the management unit. This resulted in the strong pulse of inflow at about 1700 (hour 48)on June 2. The measurements at Rockefeller in September occurred during a period of brisk north winds and low tidal range (6 cm). At the unmanaged site this resulted in a net flow out of East Little Constance Bayou were flow out totaled 22 h and flow in 8 h. By comparison, the outflow at the managed area totaled only 14 h due to high water outside the structure. The November sampling also occurred during a period of strong north winds, with a strong net outflow at the unmanaged area. Water flowed out of the bayou for 38 h and in for 8 h. At the managed site outflow was only for 18 h. These results indicate that the amplified high tide in the channel adjacent to the managed site resulted in a

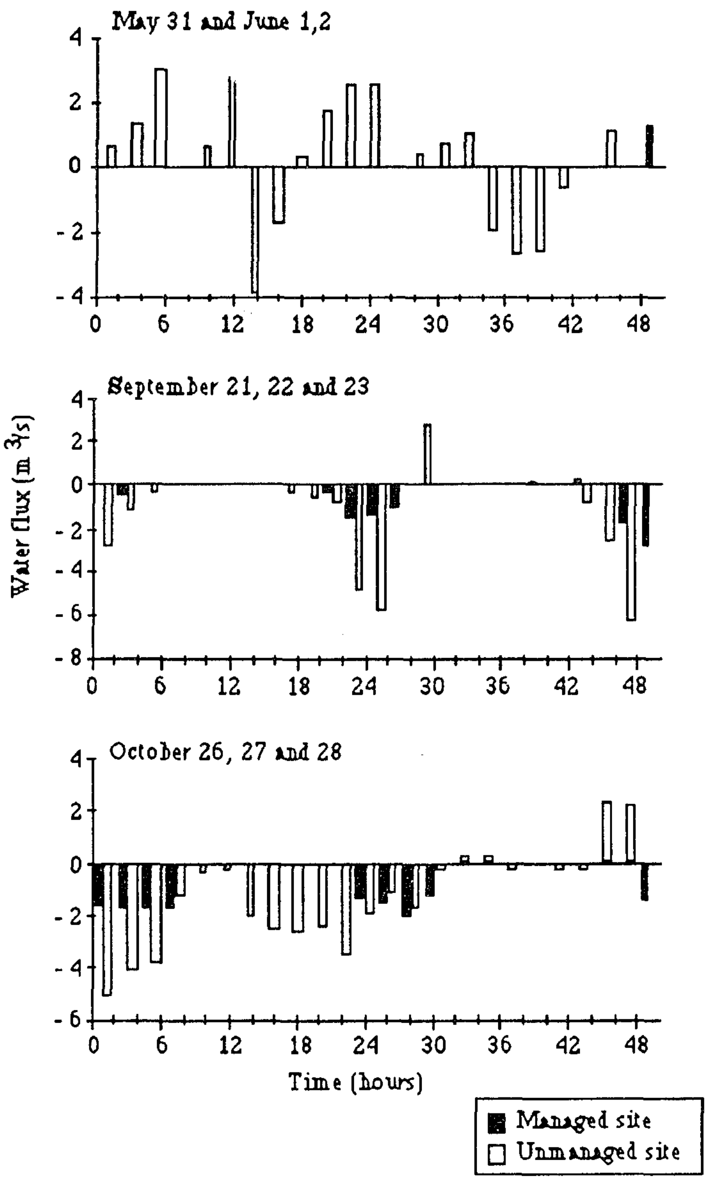

Figure 3.3.A Instantaneous water fluxes at Rockefeller per m^2 drainage area measured each two hours for three tides. Positive values indicate flux into the area and negative values are flux out of the area. Water flux for the managed area was measured at the control structure in the southwestern corner of unit 4. Waterflux for the the unmanaged area is for water exchange in East Little Constance Bayou north of East Constance Lake.

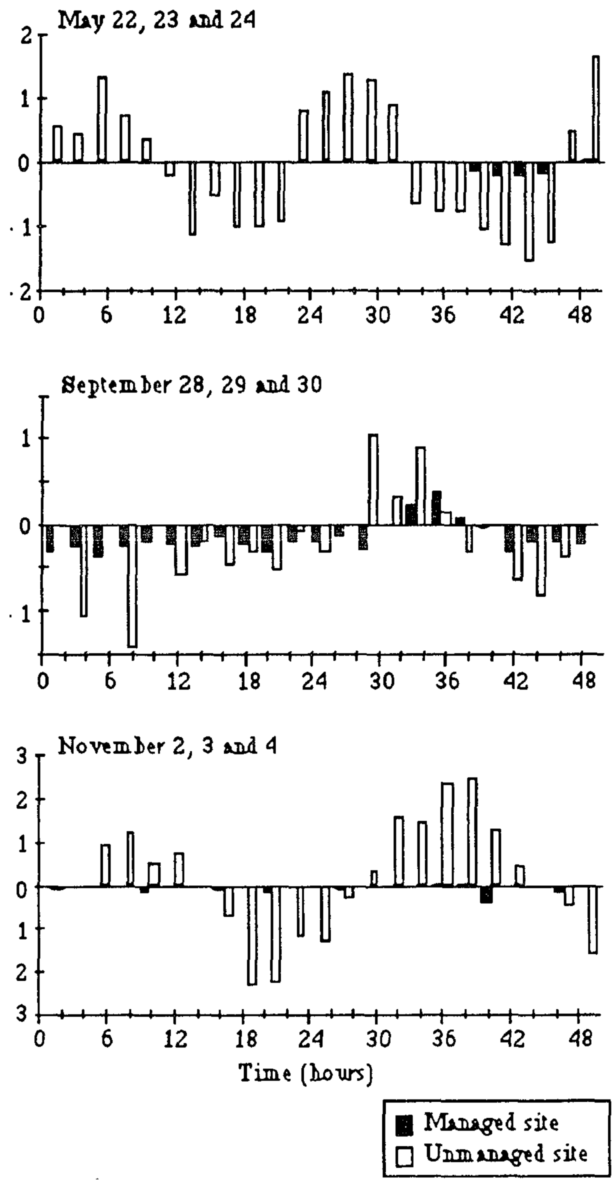

Figure 3.3.B Instantaneous water fluxes at Fina per m^2 drainage area measured each two hours for three tides. Positive values indicate flux into the area and negative values are flux out of the area. Water flux for the managed area was measured at the control structure north of Falgout Canal. The waterflux for the unmanaged area is for water exchange at five points with Falgout Canal.

reduction in the time the managed area drains as compared with the unmanaged site.

Water fluxes at the Fina unmanaged site showed the effects of the ebb and flood of the tide, but the peak fluxes were smaller than at Rockefeller (Figure 3.3). Peak fluxes were between 1.5 and 2.5 m^3 s^{-1}. By comparison, peak fluxes at the managed site were less than 0.5 m^3 s^{-1}. In May no strong winds and a tidal amplitude of 12 cm resulted in a normal diurnal pattern of flood and ebb. The flux data for the unmanaged site, with a regular pattern of inflow and outflow, reflect the tidal signal. The managed area only had outflow for 8 h and no inflow as flap gates were closed by hydrostatic pressure. In September, the flap gates were not operational and therefor inflow was not prevented. The tidal range was again 12 cm but was influenced strongly by winds with variable speeds and shifting directions. This resulted in a somewhat irregular pattern of water flux without the clear diurnal tidal signal. The managed and unmanaged site both had in or outflows, with the larger flows at the unmanaged site. In October, again the flap gates were not operational. Again a more typical tidal signal resulted in a regular pattern of water flux at the unmanaged site. Water fluxes at the managed area were low. Overall, the unmanaged site had a slight net inflow for each of the three sampling trips.

NET MATERIAL FLUXES AND MATERIAL CONCENTRATIONS: The data on instantaneous water fluxes and material concentrations were used to calculate total net material fluxes per m^2 of drainage area during each sampling period. These net fluxes at the managed areas were plotted against the net fluxes at the unmanaged areas for time specific comparisons (Figure 3.4). In general, the magnitude of the net fluxes were much greater for the unmanaged areas, due mainly to the greater water transport

Table 3.2. Net fluxes per m^2 of drainage basin for the three tidal flux studies. Negative values are materials leaving the basin while positive values indicate fluxes entering the basin. Unm are values for unmanaged sites while Man are values related to managed sites.

		Water (L m-2 d-1)		NaCl (g m-2 d-1)		NH4+ (µmol m-2 d-1)		NO2/NO3 (µmol m-2 d-1)		PO4 (µmol m-2 d-1)		TSS (g m-2 d-1)	
Study site	Month	Unm	Man	Unm	Man	Unm	Man	Unm	Man	Unm	Man	Unm	Man
Fina	May	6.02	-0.10	0.55	-0.20	35.62	-0.12	117	-0.71	0.42	-0.030	0.35	0
	Sep	5.07	-0.41	11.05	-3.85	30.46	3.80	48	1.7	1.57	0.069	0.26	-0.01
	Oct	14.86	-0.22	11.76	-0.53	129	-0.21	550	-1.55	12.24	-0.040	0.78	-0.01
	Aver	8.65	-0.25	7.79	-1.53	65.04	1.16	239	-0.19	4.74	-0.046	0.46	-0.01
	StE	3.11	0.09	3.63	1.16	32.03	1.32	157	0.98	3.76	0.035	0.16	0.002
Rockefeller	May	1.21	0.42	11.26	2.35	26.32	3.18	-5.89	9.25	0.81	0.76	0.16	0.14
	Sep	-4.50	-1.31	-45.1	-9.65	-1.92	-0.16	-4.41	-0.72	-5.58	-0.79	-0.41	-0.27
	Oct	-7.61	-2.03	-29.3	-57.3	-20.4	-3.17	-12.8	-2.35	-6.40	-1.01	-0.23	-0.25
	Aver	-3.63	-0.97	-21.1	-21.5	1.34	-0.05	-7.7	2.06	-3.72	-0.34	-0.16	-0.12
	StE	2.58	0.73	16.78	18.21	13.58	1.83	2.59	3.63	2.28	0.56	0.17	0.13

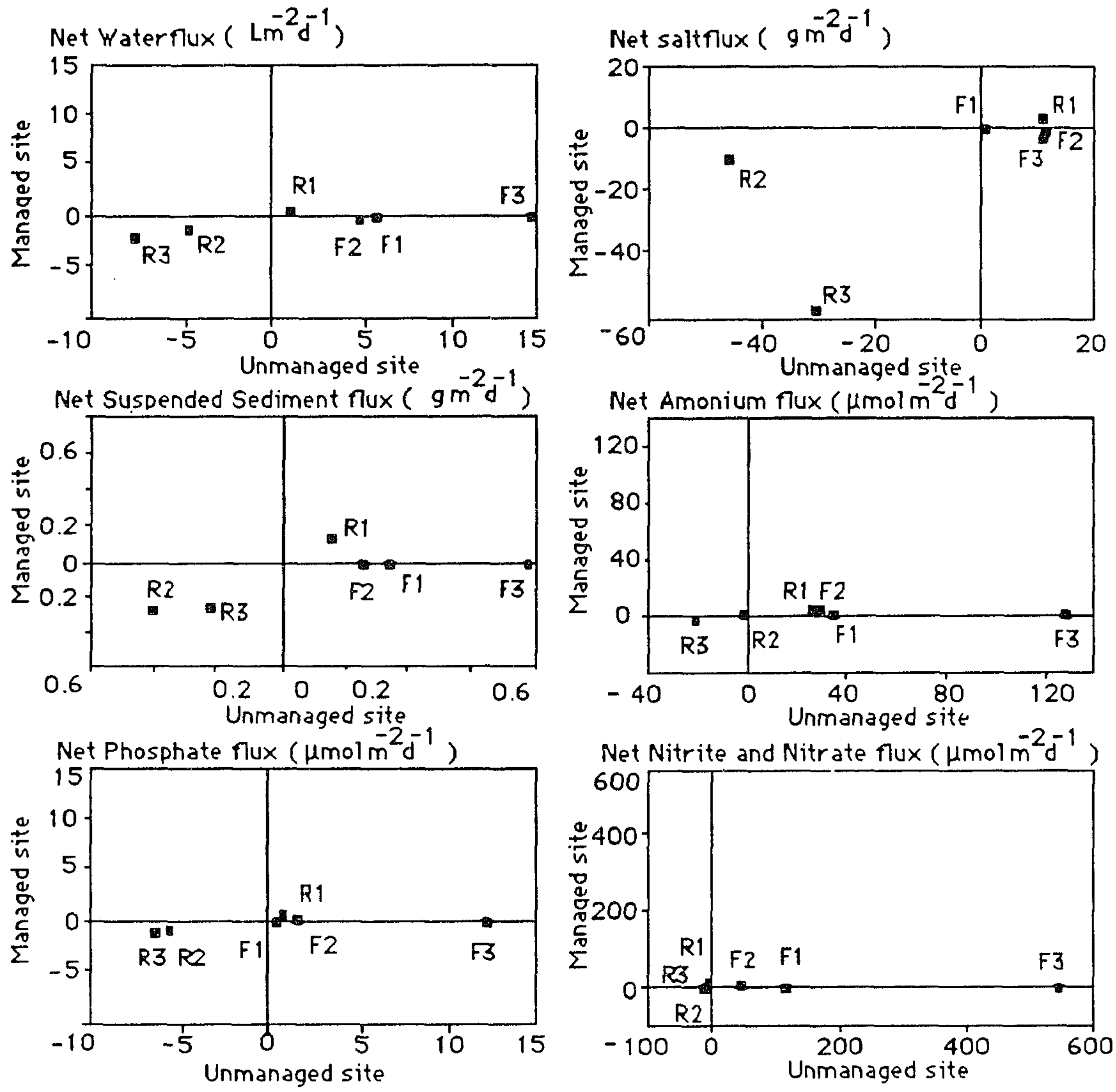

Figure 3.4. A comparison of net material fluxes per m^2 of the study area for the managed and unmanaged areas for each 48 h sampling period. The net fluxes for the unmanaged areas are on the horizontal axis and for the managed areas are on the vertical axis. Each circle compares the net flux between managed and unmanaged areas for a particular constituent (water, salt, suspended sediments, ammonium, phosphate, and nitrate plus nitrite) for an individual sampling trip. Positive values indicate flux into the area and negative values are flux out of the area. R and F stand for Rockefeller and Fina, respectively, and 1, 2, and 3 are for the first, second, and third trips.

(Figure 3.4, Table 3.2). The unmanaged areas had a net material import on some trips and a net export on others. The managed area at Rockefeller had a small import in May while fluxes in September and November were generally near zero or exports. The import during May was a result of management because the gate was opened only for a brief period at the end of the 48 h period and this allowed the only water flow for the entire study period.

The Rockefeller unmanaged area had a strong net export of water during the September and November trips and a slight net import during the May trip. The Fina unmanaged area had a net import of water at the unmanaged site for all three trips. The Rockefeller managed area had a strong net export of water in September and November reflecting the north winds during the sampling trips and a small net water import in May. The managed area at Fina had a small net export of water for each of the trips. These net water transport patterns were the main driving forces affecting materials transport.

The most striking result of TSS transport is the strong net export from both managed and unmanaged areas at Rockefeller during September and November. Fina had a net import of suspended solids into the unmanaged area and a slight export from the managed area for all three trips. TSS were significantly higher in the managed areas ($p<0.0001$) and higher at Rockefeller ($p<0.0001$, Table 3.3). These differences result from the high TSS concentrations at Rockefeller during the two trips with strong north winds. These results suggest that the common management practice of draw down by draining during strong north winds may result in considerable sediment export from managed areas.

The net flux of salt closely followed net water flux because salinity behaves conservatively in the short term and can be used as a

tracer of water masses (Day et al. 1989). Conservative behavior means that the concentration is changed only by dilution. The Rockefeller managed and unmanaged areas had considerable net exports of salt in September and November. The Fina unmanaged area had a net import of salt in September and October and the Rockefeller unmanaged area in late May. On other trips the net flux of salt was close to zero. These results suggest that when there is an net export of water from an area there can be a net export of salt. Salinity levels during the flux studies were higher in the managed areas ($p<0.0001$) and at Rockefeller ($p<0.0001$, Table 3.3). These are reflective only of the period of the flux studies. (Rogers et al. 1992) reported that salinity was significantly higher in the Fina managed area during the draw down period while (Flynn et al. 1990) reported lower salinities in the Rockefeller managed area.

The Fina unmanaged area had a net import of NH_4 and NO_3 on all three trips. The Fina managed areas showed very small net fluxes of these materials The net fluxes of NO_3 at both Rockefeller unmanaged and managed sites were small on all three trips reflecting the low NO_3 concentrations during these periods. However, at Rockefeller during September and November we recorded exports of NH_4 from the unmanaged and managed areas. When expressed on a m^2 basis, the fluxes at the managed areas of both sites are close to zero. The net flux patterns for PO_4 are similar to those of inorganic nitrogen. The Fina unmanaged area had a net import on all three trips, but almost no net flux at the managed area. The Rockefeller managed and unmanaged areas had considerable net exports of PO_4 in September and November reflecting the net water export. On a per unit area basis, the net fluxes for the managed areas were extremely small. Both NH_4 and PO_4 concentrations were higher at Rockefeller but NO_3 levels were higher at Fina (Table 3.3). PO_4 levels were higher at the managed areas.

Table 3.3. Mean and standard error of short term sedimentation rates, soil parameters, and concentrations of materials measured during flux studies for each of the study sites. (Tot Sed=total sedimentation, In Sed=inorganic sedimentation, Org Sed=organic sedimentation, % OM=% organic matter of deposited material).

	Rockefeller		Fina		p
Parameter	Managed	Unmanaged	Managed	Unmanaged	value
Short Term Sedimentation					
Tot Sed (g m-2d-1)	0.57± 0.78	1.02± 0.72	1.68± 0.64	3.81± 0.67	0.21
MinSed (g m-2d-1)	0.08± 0.26	0.10± 0.24	0.35± 0.2	1.57± 0.22	0.02
OrgSed (g m-2d-1)	0.50± 0.55	0.92± 0.51	1.33± 0.46	2.29± 0.42	0.59
% OM	85± 3.5	82± 3.2	79± 3.1	51± 3.0	0
Medium term Sedimentation (Cahoon 1990)					
% OM after 1 y	59± 3	26± 1	28± 5	53± 8	
Soil Parameters					
Phos(mg Kg-1)	101± 7.5	118± 7.7	111± 8.3	232± 7.8	0
Na(g Kg-1)	8.6± 1.5	6.5± 1.5	7.6± 1.6	9.0± 1.5	0.25
%OM	8.7± 0.2	7.2± 0.2	5.6± 0.2	5.8± 0.2	0
Material Concentrations During Flux Studies					
Sus Sed (mg l-1)	28± 5	26± 11	160± 9	63± 7	0
Salinity (ppt)	2.29± 0.26	0.42± 0.59	8.99± 0.45	6.53± 0.34	0.5
NH4 (µM)	3.1± 0.4	3.7± 0.8	5.4± 0.7	4.3± 0.5	0.19
NO3 (µM)	6.9± 0.8	8.8± 1.8	1.6± 1.4	2.0± 1.0	0.57

Flux studies have been used to calculate material budgets for different wetlands (Woodwell et al. 1977; Dame et al. 1986) and to characterize the interaction between wetlands and adjacent waters under varying conditions of such forcing as tide range, river flow, temperature, and storm events (Childers and Day 1990a; Childers

and Day 1990b; Stern et al. 1991). We want to stress that it was not our objective to use the flux results to calculate a long term budget. Rather we wanted to get an idea of the behavior of the system under different conditions of physical forcing (tides and weather events) and structure operation.

Tidal exchange coupled with winds affected the direction and magnitude of flux at the unmanaged areas. Strong north winds (as in September and November at Rockefeller) caused strong net fluxes of water and materials. When water exchange was mainly tidally driven, net fluxes were much smaller. Net fluxes over a single tidal cycle can be very different from day to day and during different seasons (Kjerfve and Proehl 1979; Stern et al. 1991) depending on local climatic and hydrologic conditions. Thus, during the September and November trips at Rockefeller, north winds caused considerable net export of water, TSS, salt, ammonium and phosphate from both the managed and unmanaged sites. We suspect that the TSS loss resulted from resuspension of bottom sediments in shallow ponds and waterways and subsequent export with water flowing out of the areas. By contrast the Fina unmanaged area had a net import of water and TSS on all three trips. Export of TSS has also been related to strong rainstorms at low tide (Settlemyre and Gardner 1975; Ward 1981; Childers and Day 1990a). Net uptake of TSS has also been reported (Wolaver et al. 1988) such as we measured for the unmanaged area at Fina for all three trips.

Net fluxes were much lower at the managed areas, a result of the structures and their operation. Even when flap gates were non operational, the structures in Fina and Rockefeller unit 4 carried less water than the natural channels in the unmanaged areas (Figure 3.3). The operation of the structures further lowered total water flux and, on average, led to a net export of water (Figure 3.5). Thus management, during the periods we studied, substantially lowered interactions between the managed wetlands

and estuarine waters. When expressed on a per unit area basis, these results show, the managed wetlands when compared to the unmanaged areas are functionally uncoupled from the surrounding estuary. A second affect of management was to convert the managed systems, depending on wind and water level conditions, from slightly to strongly exporting systems. These results also suggest that management can be a way to export salt and therefor lower salt in a managed area.

SHORT TERM SEDIMENTATION: The results show generally that short term sedimentation was higher at the unmanaged sites and at Rockefeller (Table 3.3). Total sedimentation rates were significantly higher at Rockefeller than those at Fina ($p<0.05$) and were higher in the unmanaged areas for both sites ($p<0.09$, Table 3.3). Deposition of mineral sediment was significantly higher at Rockefeller ($p<0.01$) and at the unmanaged areas ($p<0.01$), while there was no significant differences in organic matter deposition between areas ($p<0.11$) or management practice ($p<0.21$) The % organic matter of deposited material was significantly higher at the managed sites ($p<0.0001$) and at Fina ($p<0.0001$). Streamside sedimentation was significantly higher than inland only for the Rockefeller unmanaged site (Figure 3.5).

Total sediment deposition at Fina ranged from 0-2.3 g m^{-2} d^{-1} and 0-1.32 g $m^{-2}d^{-1}$, and mineral sediment deposition ranged from 0-0.26 g $m^{-2}d^{-1}$ and from 0-0.38 g $m^{-2}d^{-1}$ for the unmanaged and managed sites, respectively. The organic fraction ranged from 26-100% and from 55-100% for the unmanaged and managed areas, respectively. Total sediment deposition at Rockefeller ranged from 0-11.9 g $m^{-2}d^{-1}$ and 0-3.6 g $m^{-2}d^{-1}$, and mineral deposition ranged from 0-4.22 g $m^{-2}d^{-1}$ and from 0-1.08 g m^{-2} d^{-1} for the unmanaged and managed sites, respectively. The organic fraction values ranged from 9-74% and from 15-100% for the unmanaged and managed areas, respectively. Reed (1989, 1992) reported short term sedimentation rates from near zero to 40 g m^{-2} d^{-1} for a

number marsh sites in the Louisiana coastal zone. Our findings were Rockefeller shows higher sedimentation rates then Fina agrees with Reed (1989, 1992) who showed that sedimentation rates are higher near the coast.

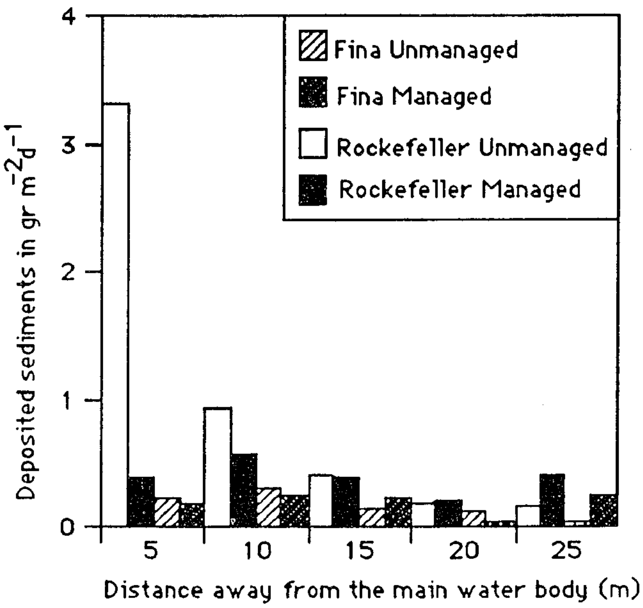

Figure 3.5. Total short term sedimentation rates at the four sites. Values for each station are averaged over the period between August 1989 and January 1990.

These results indicate that management leads to decreased sedimentation of allochtonious materials on the marsh surface. In the same overall study, Cahoon (1990) found that vertical accretion over marker horizons was from 5-10 times lower in the managed areas. Reed (1992) reported lower short term sedimentation at a number of marsh sites in coastal Louisiana which had fixed-crest weirs. New introduced materials to the marsh surface are crucial to offset subsidence and maintain an elevation relative to locale water levels suitable for marsh plants to survive and reproduce (Day and Templet 1989).

Sedimentation was high after the passage of Hurricane Gilbert (September 26 1989) in all areas except Rockefeller managed. The importance of storm-dominated depositional patterns, especially by hurricanes, has been reported by a number of authors for both Louisiana and other coastal areas. (Baumann et al. 1984; Meeder 1987; Rejmanek et al. 1988; Reed 1989). The results of our study and others mentioned above show that the greatest deposition of sediments in Louisiana coastal marshes occurs during storm events when large amounts of sediments are mobilized through resuspension and moved onto the marsh during high tides. The reduced sediment input and sediment deposition at the managed areas in comparison to the unmanaged areas can be explained by a general marsh management objective as it seeks to use water control structures to reduce water inflow and flooding during storm events (Clark and Hartman 1990).

SOIL PARAMETERS: The percentage organic matter in deposited sediments decreased over time. We found the amount of organic matter (OM) in soil samples to be about 10% of the organic level of sediments found on the perti dishes (Table 3.3). Very likely, even less of the original deposited organics had become soil components, because soil organics also originate from in situ produces root mass. The percent OM of materials accumulated in one year over marker horizons (Cahoon 1990) were in between the percent OM values of soil and the recently deposited sediments. These data suggest that the deposition of organic materials to these marshes is mainly lost and has little significance to elevation (Nyman and DeLaune 1990).

The results of the soil analysis showed that phosphorus was significantly higher in the unmanaged areas ($p<0.0001$) and significantly higher at Rockefeller ($p< 0.01$). We did not find significant differences for soil sodium among the different sites.

PLANT PRODUCTIVITY: Flynn et al. (1990) studied productivity of Spartina patens. They reported that productivity was higher at Rockefeller managed but lower at Fina managed as compared to the unmanaged sites. They concluded that the higher productivity at Rockefeller managed was due to an effective draw down which led to greater soil oxidation and higher Eh values. The opposite was true at Fina managed where the soils were more waterlogged and reduced.

FISHERY COMMUNITIES: Rogers et al. (1992) studied the effects of management on fishery communities. At Fina, more grass shrimp and resident minnows (least killifish, western mosquito fish, and golden top minnow) were collected in the managed area, while more marine-transient organisms (gulf menhaden, blue crab, and striped mullet) were collected in the unmanaged area.

CONCLUSIONS

In general the results of this study show that marsh management significantly reduced water and materials exchange at both Rockefeller and Fina. Short term sedimentation was generally higher in the unmanaged areas and at Rockefeller. There were also significant differences in soil organic matter and soil P between managed and unmanaged sites and between Rockefeller and Fina. The reduced sediment input to the managed areas and reduced short term sedimentation are reflected in longer term accretion rates (Cahoon 1990).

We concluded from the flux studies positive and negative implications for marsh management. We found that structure operation converts managed areas to net exporting systems. One of the goals of management is to lower salinity and we showed a net export of salt at both Fina and Rockefeller. Draw down, one of the most important management operations, is most effective when carried out during north winds when coastal water levels are

low, which can lead to a net loss of sediments and nutrients. Management may lead to freshening, a progressive sediment deficit and a loss of fertility. It must be remembered, however, that these measurements were carried out at only two areas and during draw down operation. Further study should be carried out for a wider variety of marsh management plans under different operational phases before broad generalizations are made concerning structural marsh management.

CHAPTER 4

HIGH PRECISION MEASUREMENTS OF SEDIMENT ELEVATION IN SHALLOW COASTAL AREAS USING A SEDIMENTATION-EROSION TABLE

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INTRODUCTION

Sediment elevations in inter tidal and shallow sub tidal coastal areas are constantly changed by such factors as tides, storm activity, subsidence and biotic activity. Annual elevation changes in coastal wetlands ranges from less than 0.01 $cm \cdot yr^{-1}$ in stable sediment-starved areas to about 5 $cm \cdot yr^{-1}$ in areas with high sedimentation rates (Letzsch and Frey 1980; Anderson et al. 1981; Baumann et al. 1984; Rejmanek et al. 1988). There are longer term changes in sea level that lead to changes in the surface elevation in shallow coastal areas. For the last century there has been an average eustatic sea level rise of 1-2 $mm \cdot yr^{-1}$ (Gornitz et al. 1982). In coastal areas with high subsidence rates, such as the Mississippi delta, relative sea level rise can exceed on $cm \cdot yr^{-1}$ (Baumann et al. 1984). Coastal wetlands must grow vertically at the same rate as local water-level rise if they are to survive over the long term.

Precise methods for measuring these changes in the elevation of the sediment surface are necessary to determine the rates of elevation change and to gain an understanding of the processes responsible for the changes. Methods which have been used to study these changes include radio tracers such as ^{137}Cs (DeLaune et al. 1978), marker horizons (Letzsch and Frey 1980; Cahoon and Turner 1989), rare earth horizons (Knaus and Gent 1987), sedimentation pins (Letzsch and Frey 1980; Pethick and Reed 1987), and precision surveying (Anderson et al. 1981). These methods have several limitations and the accuracy of most of them has not been determined.

Marker horizons are not stable reference points and therefore generally not useful to measure elevation change (Baumann and Adams 1981; Cahoon and Turner 1989). Erosion pins measure sedimentation and erosion, but the pins are very sensitive to

disturbance. Stakes and rods have been used to measure elevation changes at least since 1949 (Harbord 1949; Pestrong 1965; Reed 1988) but the accuracy has not been reported. Precision surveying can provide elevation change. (Anderson et al. 1981) compared the means of 93 points on inter tidal mud flats surveyed twice within a two day period, and calculated a 95% confidence interval (C.I) of ± 0.3 cm. The method would likely be much less accurate in wetlands and sub tidal areas where the determination of the surface would be much more difficult.

In this paper we report on the use of a sedimentation-erosion table (SET, Figure 4.1), a non-destructive method for precisely measuring elevation of inter tidal and sub tidal wetlands over long periods. Our design is derived from the SET used by (Schoot and Jong 1982), who reported a standard error of 0.08 cm with a 95% C.I of ± 0.45 cm. In this paper we describe the precision of a more versatile SET for use in a wider variety of environments.

METHODS

STRUCTURE OF THE SEDIMENTATION-EROSION TABLE: The SET has a supporting aluminum base pipe (10 cm diameter, 1 mm wall thickness) placed permanently at each site which is designed to receive the upper portable part of the SET (Figure 4.1). This core pipe was driven into the soil to refusal with either a vibracorer or a hand-held pile driver as near to vertical as possible. The core pipe was then cut off a few cm above the sediment surface and filled to within a meter of the surface with quick-setting cement. The elevation of the top of the pipe after cutting will vary depending on water depth at the site and the tidal range.

Next an aluminum base support pipe (diameter 7 cm, length 60 cm, wall thickness 3 mm) was cemented into the top of the core pipe with mortar mix. No cement was placed inside the top 30 cm of the base support pipe to allow the portable part of the SET to be

inserted. The support pipe extended about 5 cm above the core pipe and was leveled to vertical. The portable part of the SET has four components: a vertical arm, a horizontal arm, a flat plate or table, and nine pins. The vertical arm fits 25 cm into the base

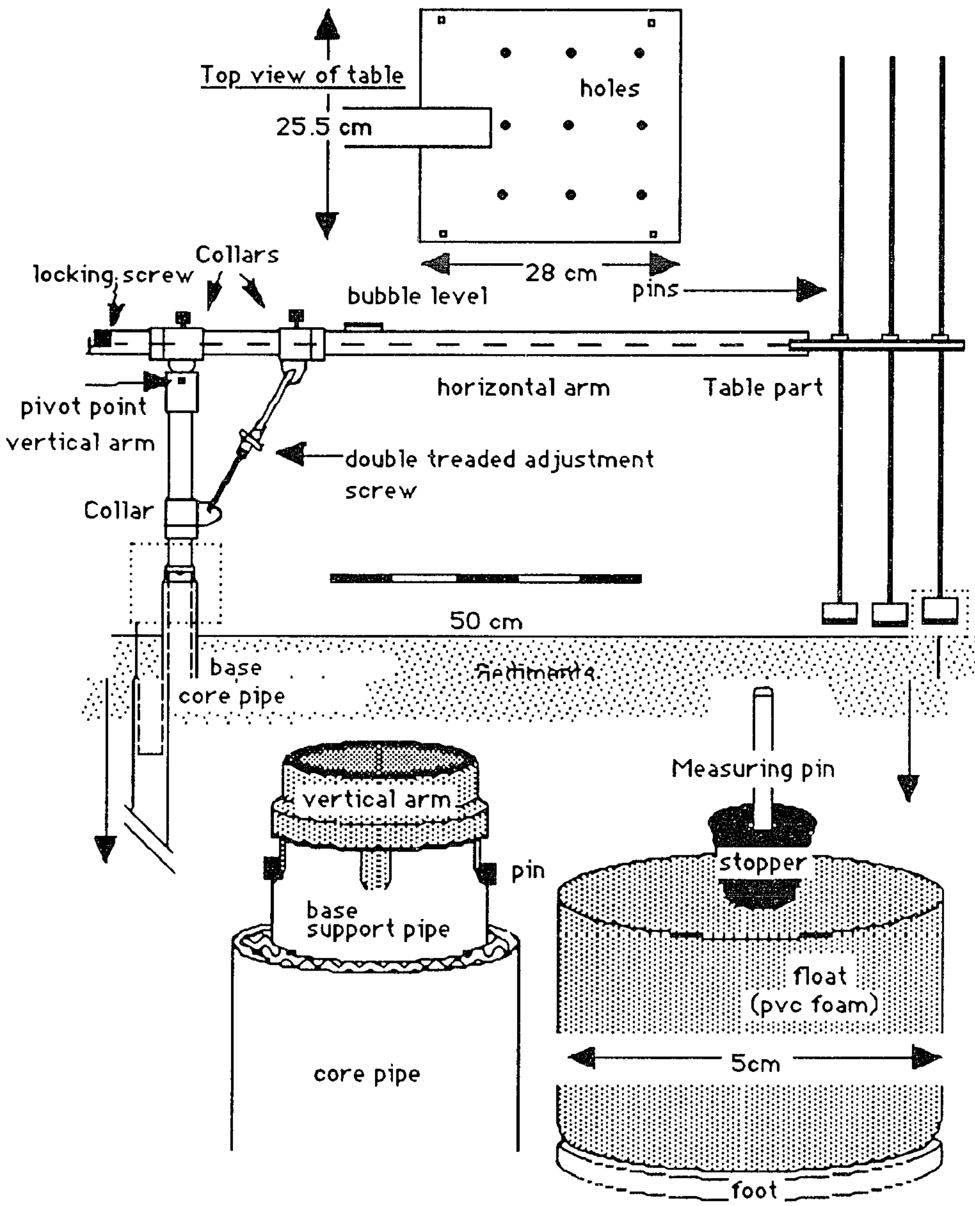

Figure 4.1 Overview of the Sediment Erosion Table (S.E.T.).

support pipe and measurements can be made in four different directions. A horizontal pin fits into notches in the base support pipe so that the four positions are always in the same place.

The horizontal arm of the SET can be leveled in two planes. It attaches to the vertical arm with a pin at the pivot point. This pivot point allows the arm to be leveled up or down with a double threaded adjustment screw. The horizontal arm can be leveled from side to side by rotating the arm in the two collars and tightened with screws. A bubble level determines when the horizontal arm is level in both planes. When leveled, the table on the end of the horizontal arm provides a constant reference plane in space.

The plate measures 255 mm x 280 mm. The distance to the sediment surface is measured with nine pins passing through holes in the table. The holes were drilled in a square design (3 x 3) and 75 mm apart. The number of pins allows for an equal number of replicate measurements. The table has three separate plates with nine holes in each plate. The upper and lower plates are fixed and the middle plate can be moved back and forth. This movement is controlled by the operator, who uses the locking screw at the non-table end of the SET. This particular screw is attached to a threaded rod, which runs through the horizontal arm to the middle plate. Detailed specifications of the SET are available from the authors.

SITE PREPARATION: Before installation of the SET, the site was prepared so that disturbance of the area was minimized. At each site platforms were built consisting of treated boards (2"x12"x 8') on supports permanently attached to treated posts. A movable board was brought to the site and used for SET installation and measurement. For all installation and measurement activities, the site was approached via the same corner of the platform and all measurements were carried out from the platforms.

OPERATION: To operate the SET, the vertical arm is set into the base support pipe so that the horizontal pin fits into the notches. The plates are lined up to insert the pins, then by moving the middle plate slightly, rubber o-rings inserted in the middle plate secure the pins in position to allow the table to be leveled in two directions. After leveling, the pins are unlocked and lowered until they just touch the sediment surface. The pins can be lowered differently depending on whether the site is covered by water or not. When there is no water, the locking screw is loosened slightly and the pins are lowered manually until they just touch the sediment surface. When the site is covered by water, the locking screw is loosened until the pins slide down under the influence of gravity and come to rest on the sediment surface. Flat "feet" with a diameter of 5 cm (surface area = 19.6 cm^2) and pvc sponge floats are attached to the bottoms. These floats retard falling speed through the water column. The combination of a pin and a foot has an average weight of 176 g, The manufacturer specified buoyancy of the floats is 92 g. Therefore, the floats provide a reduction of penetration potential from approximately 9.0 to 4.2 g/cm^2. After the pins sit on the sediment surface, they are locked in place by tightening the locking screw. The length of each pin above the table is measured with a ruler to the nearest mm. This procedure is repeated for each of the four directions to yield a maximum of 36 elevation measurements at each site.

STUDY SITES: We used SET to measure elevations at coastal areas in Georgia and Louisiana. Five sites were in Spartina alterniflora marshes and mud flats on the sound side of Cumberland Island, Georgia. Four sites were also established in shallow sub tidal ponds and mud flats in a low salinity wetland area south of Lake Pontchartrain, Louisiana. Dominant marsh vegetation is Spartina patens. The Georgia location is a stable coast while the Louisiana location is subsiding rapidly (Baumann et al. 1984; Bird 1985). The Georgia sites were never flooded when measured and the pins

were lowered manually. The sites in Louisiana were flooded during measurements and floats and feet were always attached to the pins which were lowered by releasing the locking screw.

SENSITIVITY ANALYSIS: Measurements to estimate the precision of the SET were carried out in Georgia in January, 1991 and in Louisiana in February, 1991. To estimate the standard errors for corners we used a complete randomized design (CRD) nested with replicates, samples, and repeated measures in time (duplicates). Corners of the sites were measured twice (Figure 4.2). Between duplicates we took the table off the stationary pipe, reassembled and re-leveled it. We assumed elevations of the sediment surface did not change between duplicates. Treatment was site; replicates were corners at each site, the samples were the nine pins in each corner, and the response was the height of each pin above the table. The precision calculated is defined as the confidence interval for a 9 pin measurement.

To test the hypothesis that Louisiana and Georgia had the same difference in height between duplicates. We used a CRD with replicates and samples. The treatments were locations, replicates were corners, the samples were pins, and the response was the difference between duplicate measurements for each pin. We estimated the average difference in height for the locations in Georgia and Louisiana and tested whether the mean difference was equal to 0 in each case. We performed this test because pins were lowered manually in Georgia and by gravity in Louisiana.

RESULTS AND DISCUSSION

The SET is the most precise method we know of to measure elevation changes in shallow coastal environments. The method allows for improved estimates of elevation changes as compared with the methods mentioned in the introduction. The standard error of a corner (a nine pin measurement) was 0.08 cm. The 95%

CI for a 9 pin measurement was ± 0.15 cm. The method of lowering the pins by gravitation affected the sediment surface. The elevation in the second set of measurements in Louisiana was 0.25 cm lower than in the first measurements, while for Georgia this value was 0.001 cm (Figure 4.2). The 95% CI for these estimates was ± 0.04 cm,. The disturbance of the sediment surface by penetration needs to be studied more in order to estimate it's influence on the elevation changes over longer periods of time. It

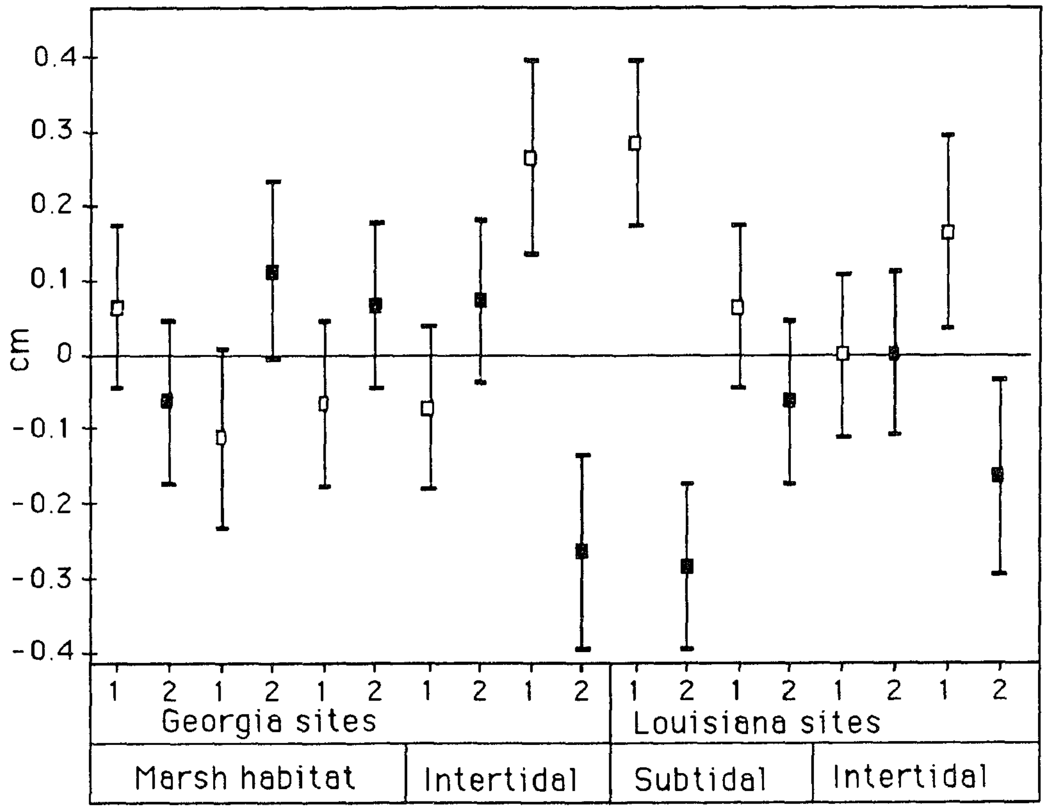

Figure 4.2 Divergence from the duplicate mean of 36 pins for repeated measurement of elevation of marsh and inter tidal habitats in Georgia and Louisiana. An open square is the divergence from the mean of the first measurement and a dark square is for the repeated measurements. Vertical bars are standard errors.

is probable that the compression of the sediment surface by the feet will not be preserved for a very long time, as the processes of sedimentation and erosion tends to even out these small elevation differences.
When monitoring sedimentation and erosion of the sediment surface, it is important to define clearly how the sediment surface is determined. With some sediments, such as sandy inter tidal flats, the surface is very distinct. For other surfaces, it is more difficult to accurately determine or reach the interface such as soft fluid muds, organic mucks, and floating marshes. The bottom of the base support pipe is assumed to be stable and is used as the bench mark. To calculate elevation change, any movement in the base support pipe such as geological down warping has to be accounted for separately.

The SET method is best used within a hierarchical statistical structure. Each site location will represent an experimental units with four samples. Each sample is the average of nine sub samples. An hypothesis is tested by comparing randomly placed sites in areas defined by the initial research question. We are presently using the SET to measure differences in elevation changes along a number of gradients. These include from the edge of the tidal stream to interior marshes located at different distances from a tidal inlet, and either side of a wave dampening structure.

Changes in the original design of Schoot and Jong (1982) were mainly scale enlargements. To measure elevations under water and in marshes with high vegetation, the distance between the table and the sediment surface was increased by approximately a factor of 3. To diminish disturbance in the measurement area, the distance from the stationary pipe to the measuring area was increased by 2.8 times. The introduction of the triple plate design enables the researcher to lower the pins when the sediment surface is not visible (i.e. when it is flooded). In marsh areas, it is

necessary to guide the pins manually to avoid interference of the vegetation and to define the soil surface visually. To enable the table to be leveled while standing at the stationary base, the leveling bubble was placed one-third the distance between the pivot point and the table rather than at the table. This, in addition to the lengthening of the horizontal arm, increased the error introduced by leveling. Even though we used aluminum instead of stainless steel, the enlargement of the design increased the weight of the SET. The added weight demanded a sturdier support base, which proved to be labor-intensive to install. The new support base structures are permanent and function as benchmarks for short as well as long term studies.

Our SET was manufactured at a local commercial machine shop, and took about two weeks to build. The total price was $1600. Site installation costs approximately $100, excluding labor and travel costs, and takes about 1.5 hours (with some experience). A one site reading takes approximately 40 minutes.

The SET method, as described in this study is limited to exposed and shallow water areas with surfaces which are reasonable to reach. Surfaces difficult to reach include those covered by dense beds of vegetation, shell reefs and those that are too deep. We believe that this technique is limited to water depths of less than approximately 0.5 m.

The SET will allow more accurate measurement of surface elevation changes compared to other methods this will lead to more rapid determination of rates of changes as well as a better understanding of the processes causing change. For example, use of the SET should allow a more rapid determination of weather coastal marshes are maintaining their elevation with respect to sea level rise. Current eustatic sea level rise is estimated at between $1\text{-}2 mm \cdot y^{-1}$ (Gornitz et al. 1982). With the SET, a surface elevation change by this magnitude would be determined in less

than 5 y. By comparison, precision surveying would require on the order of 10-20 y. Likewise, individual storm events have been postulated as very important in causing accretion in coastal marshes (Baumann et al. 1984; Reed 1989). The SET could determine accretion of a few mm due to deposition during a storm event.

CONCLUSION

The SET is precise to within a 1.5 millimeter range, and accuracy is different if the pins are lowered manually or by gravity. The SET can be used to measure the elevations of areas less than 0.5 m in water depth. Once installed, sites can be measured over a long period of time.

CHAPTER 5

THE EFFECT OF INTER TIDAL SEDIMENT FENCES ON WETLAND SURFACE ELEVATION, WAVE ENERGY AND VEGETATION ESTABLISHMENT IN TWO LOUISIANA COASTAL MARSHES

INTRODUCTION

Coastal wetlands are being lost through conversion to open water in many estuaries and deltas. This problem is particularly severe in the coastal wetlands built by the Mississippi River in Louisiana (Penland et al. 1990; Evers et al. 1992). Loss rates as high as 100 km^2 per year have been documented during the past six decades largely due to the acceleration of natural wetland degradation processes by man's activities (Craig et al. 1979; Gagliano et al. 1981; Baumann and DeLaune 1982; Turner and Cahoon 1987). Recently, the Mississippi River delta wetlands have become the focus of a major restoration effort (Louisiana 1993).

Inter tidal brush fences modeled after those used for the last century in The Netherlands (Kamps 1962; Glopper 1981; Bouwersma et al. 1986) have been deployed experimentally in more than 50 locations. They have been constructed primarily by volunteer labor to protect eroding marsh shorelines, retain resuspended sediments and promote vegetation development. In a variation on the Dutch model, these fences are "cribs" that are filled with recycled Christmas trees stacked to an elevation of 30 to 60 cm above mean sea level (Figure 5.1). The trees are collected by communities around the state after the holiday season. Once materials are assembled, six volunteers can build 50 m of fence in a day.

"Christmas tree fences" are an alternative to land filling bulky trees and recycling provides an opportunity for the public to play a "hands on" role in coastal restoration (Coalition 1989). Fence construction has become standardized through the efforts of the Louisiana Department of Natural Resources (LDNR) which has provided some funding since 1990. Criteria for systematically siting and orienting a fence have not yet been developed. It is

clear, however, that some fences work better than others but the specific sources of this variability have not been investigated.

Soil surface elevation relative to mean local water level (relative elevation) is the single most important factor controlling the colonization, maintenance or loss of inter tidal wetland vegetation (Sasser 1977; Lyon 1986; McKee and Patrick 1988; Boumans and Sklar 1990). Higher soil surface elevations lead to shorter flood duration and more frequent drainage, which result in less water logging (Howes et al. 1981; Lyon 1986; Burdick and Mendelssohn 1987), a more consolidated substrate (Rosen 1980) and reduced exposure to erosion by waves (Knutson et al. 1990). Accordingly, slightly higher elevations generally enhance production by inter tidal wetland plant species if they do cause succession to non-wetland types. Conversely, exposure of marshes and mud flats to wave action leads to reduced seedling germination and to physical damage of existing young plants (Van Eerdt 1985; Foote and Kadlec 1988).

Relative surface elevation is not fixed, however, particularly in the marshes of coastal Louisiana, but fluctuates in response to many processes operating at a variety of spatial and temporal scales. At least twenty processes are known to affect coastal wetland elevation and sustainebility (Turner and Cahoon 1987), but only three of these have the potential to increase surface elevation relative to mean sea level. These are the (1) deposition of sediments from suspension, (2) deposition of organic matter from above-ground plant production and (3) expansion and incorporation of below-ground plant production to form peaty soils. All other processes work in conjunction with eustatic sea level rise to create a site-specific submergence potential for all of Louisiana's coastal wetlands that typically ranges from 0.5 to 1.0 cm per year (Turner 1991).

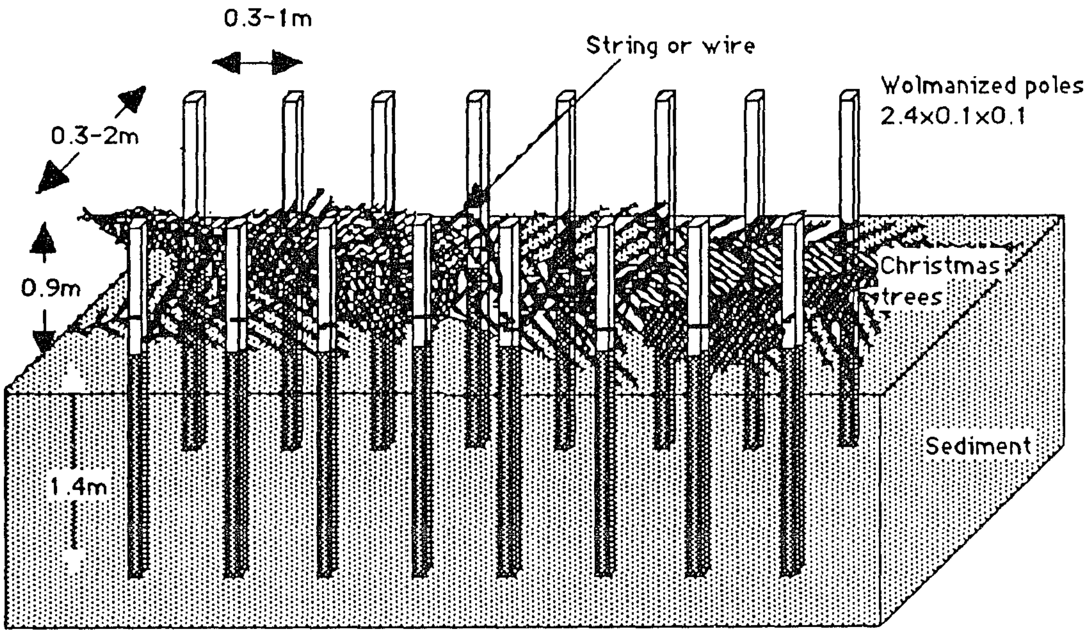

Figure 5.1 Drawing of a Louisiana sediment fence.

Any effective restoration technique must either reduce the submergence potential or enhance the three processes that can increase elevation. Inter tidal sediment fences are designed to locally increase the efficiency of sediment trapping on unvegetated sub tidal surfaces to create an elevated platform that will support colonization by emergent wetland species. Along the northern coast of The Netherlands tidally transported sediments are trapped by brush groins extended seaward of the sea dikes. There, accretion rates of up to 3.5 cm per year have been measured following colonization by marsh plants. This technique has been used to create over 100 km^2 of new marsh land in the past 40 years alone (Bouwersma et al. 1986).

Two Louisiana fence projects at the sites shown in Figure 5.2 have been systematically monitored over the past four years. One, the LaBranche project, is located west of New Orleans just south of Lake Pontchartrain. The other, the Leeville project, is north of

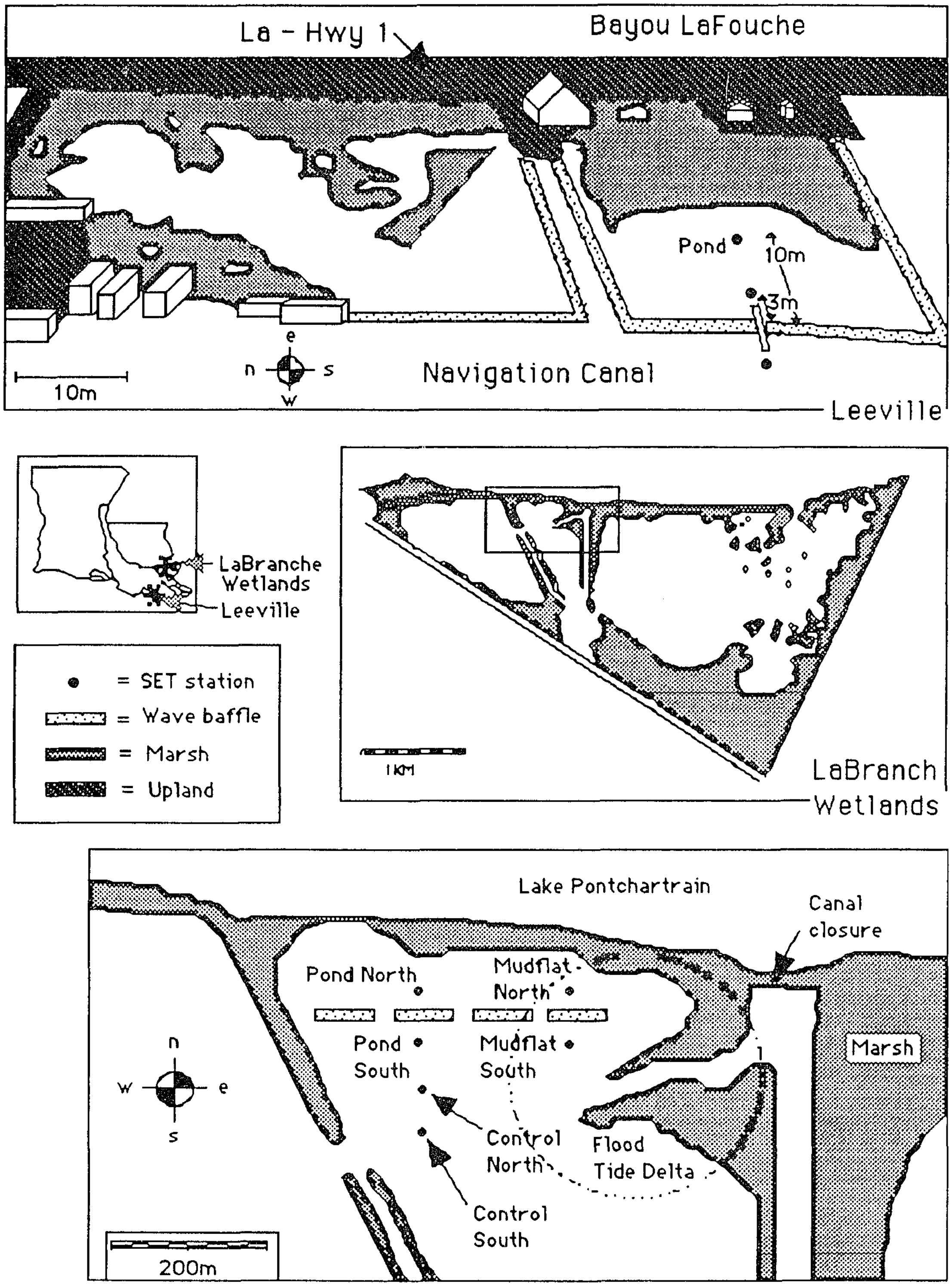

Figure 5.2 Study sites.

Grande Isle adjacent to Bayou Lafourche. Effects of these fences on wave characteristics, sediment accretion and vegetation response are reported here to improve fence design and predictability as well as enhance existing understanding of coastal wetland sedimentation processes.

In principle, sediment fences are expected to work like any sub aerial, permeable breakwater to reduce the erosive effects of waves generated by wind in small, shallow coastal water bodies (Anderson 1972; Håkanson 1977; Letzsch and Frey 1980; Carper and Bachmann 1984; Ward et al. 1984; Knutson et al. 1990). One objective of this study was to provide an understanding of the wave transmission characteristics of this type of low-cost structure to facilitate design of future projects. Transmissivity (K_T) is expressed as the ratio of transmitted to incident wave height (H_t/H_i, USCOE 1984).

Like any permeable breakwater, sediment fences provide a preferred location for deposition of sediment entrained in coastal currents or locally resuspended by waves. Rates of deposition are expected to be a function of sediment supply or flux. A second objective of the study was to quantify the effect of a fence on sedimentation in areas with different rates and modes of sediment supply. The LaBranche and Leeville sites provide this opportunity

Dutch researchers have noted an enhancement of sedimentation following colonization of the unvegetated surface by wetland plants (Bouwersma et al. 1986). A final objective of this study was to record the sequence in which wetland plant species naturally colonize the platform developed around the fence and tie this sequence into observed changes in elevation.

STUDY SITES

The LaBranche and Leeville study sites shown in Figure 5.2 are compared in Table 5.1 and described briefly below. The two projects differ significantly in scale and design. The LaBranche

project involves more than 300 m of fencing placed in a detached breakwater configuration protecting a semi-enclosed area of nearly 40,000 m^2 while the Leeville project consists of two enclosures covering about 20,000 m^2. The LaBranche fence was constructed in January 1989, while that at Leeville was built a little over a year later in March 1990.

LABRANCHE PROJECT:. The LaBranche wetland, located on the south shore of Lake Pontchartrain between the Bonnet Carre Spillway and New Orleans, is among the most productive low salinity habitats in the Lake Pontchartrain estuary (Cramer et al. 1978). The area is characterized by shallow ponds and deteriorating marshes established on a very poorly drained organic soil. The marsh has been classified as intermediate (Chabreck 1972) and consists of a mix of brackish (Spartina patens) and freshwater species (Acnida cuspidata, Scirpus validus, Kosteletzkya virginica, Ludwigia sp., Baccharis halimifolia, Eleocharis sp., Ptilimnium capilaceum, Alternanthera philoxeroides and Ranunculus sceleratus). Spartina alterniflora was introduced into the area in the mid 1980's and this species has slowly increased in abundance.

Several passes formerly connected Lake Pontchartrain to the LaBranche wetland. Sediment and water was exchanged with the lake in the immediate vicinity of the study site through the pipeline canal shown in Figure 5.2. This canal and all other openings to the lake were either dammed or blocked by weirs in 1987 as part of an early marsh restoration project. The LDNR collected hourly water level data in the LaBranche wetland

between June 1992 and January 1993 which indicated that the average tidal amplitude in this largely isolated area had been reduced to approximately 8 cm.

Table 5.1 Site and monitoring specifics.

Parameter	Leeville	LaBranche
Vegetation Type	Salt marsh	Intermediate marsh
Daily water level range ± 1 Stdev *	0 ± 10 cm	8 ± 4.5 cm
Water level monitoring period	10/'86 to 1/'91	6/'92 to 1/'93
Water depths at monitoring sites	0 to 60 cm	0 to 60 cm
Concentration of TSS samples ± 1 StE **	93±8 mg•L-1	61±2 mg•L-1
Potential Source of TSS	Storms and Boat traffic resuspension	Lake Pontchartrain
Number of SET stations	3	6
Monitoring period for SET stations	9/'90 to 7/'93	3/'90 to 7/'93

*Louisiana Department for Natural Resources-the Coastal Restoration Division

** 500 ml bottles taken at irregular intervals throughout the sampling period.

delta in the study area which was colonized by vegetation before the canal was closed in 1987. Some sediment continues to enter the area from the lake though the flux has been significantly reduced. The Christmas tree fences were constructed by the LDNR in January 1989 in a shallow pond starting on the east in this shoal area and extending west into somewhat deeper water. The fences have not been subsequently refurbished.

LEEVILLE PROJECT: The Leeville site is located in a deteriorating <u>Spartina alterniflora</u> marsh adjacent to Bayou Lafourche about 20

km from the Gulf of Mexico. The site is located next to a small navigation canal in the middle of the Leeville Oil Field, one of the oldest and largest fields in coastal Louisiana. The sediment fences were constructed by the LDNR and the Lafourche Parish Coastal Zone Advisory Committee in 1990 to enclose a shallow open water area which had opened up along the east bank of the canal. The enclosed area becomes somewhat deeper toward the canal. The canal is continually stirred by boat wakes and is the primary source of suspended sediment. The fences have been refurbished each year with new Christmas trees.

METHODS

Nineteen sets of wave data were acquired during the study at the two areas at the times shown in Table 5.1. Each data set consists of two simultaneous wave records 26 minutes long sampled at a frequency of 4 hertz with pressure transducers (PDCR 830-0576, 2.5 psi range) positioned on the seaward and land ward sides of the fence. Long period (tidal) trends were removed from the time-series and the power spectra were computed using a Fast Fourier Transform algorithm (FFT, IGOR). Summary statistics for wave energy, significant wave height (Hs) and wave period were derived from the power spectra following (Khandekar 1989). Suspended sediment concentrations were determined from water samples collected at the time that the wave records were acquired.

Sediment surface elevation was measured at each station shown on Figure 5.2 every three months using a sedimentation-erosion table (SET, Boumans et al. 1993). With the SET, surface elevation is measured at nine points in each of four quadrants relative to a benchmark set in the base support pipe for a total of 36 measurements. A small platform was constructed at each SET location to minimize disturbance of the sediment surface during sampling. Six stations were established at LaBranche. Four were positioned 25 m from the centerline of the fence, while controls

were placed at 100 m and 150 m seaward of the fence. Three stations were established at Leeville. Two were placed three meters from the fence centerline on the canal and land ward sides, while the third site was positioned in the center of the enclosed area 10 m land ward of the fence.

Each SET station was treated as an experimental unit while each pin reading was treated as a sub sample. An ANOVA model was constructed to test the effect of fetch, proximity to the sediment fence, field site, season and Hurricane Andrew (August 1992) on the change in elevation and the effect of hurricane Andrew on total elevation (Appendix A). Linear regression was used to establish the rate of elevation change in $cm\text{-}yr^{-1}$ at all sites.

Observations of new vegetation growth were recorded during site visits. Plant species were noted as present even if only a few individuals were seen. More diverse species assemblages were assigned to a typical plant association

RESULTS

Water depths monitored at the two field sites were similar, ranging from 0 to 90 cm at LaBranche and from 0 to 70 cm at Leeville. Suspended sediment concentrations exhibited little variation between visits at a single site but were, on average, 50 percent higher at the Leeville site (93 ± 8 $mg\text{-}l^{-1}$) than at LaBranche (61 ± 2 $mg\text{-}l^{-1}$).

WAVE EFFECTS: Time-series of the smallest amplitude waves recorded at Leeville, the highest amplitude waves recorded at LaBranche, and a wave train recorded at Leeville which was partially generated by a boat wake are shown in Figure 5.3 Dominant periods in the wind wave band were estimated from the spectra to range from 0.6 to 5 seconds with a mean of 1.9 seconds

at both sites. Periods did not differ significantly between field sites and were not affected by the fences.

Significant wave heights measured at both sites were very small as would be expected for such shallow water depths, ranging from a few mm to 1.7 cm. It is clear from the percent exceedence curve plotted in Figure 5.4, however, that the incidence of larger waves is far greater at the Leeville site than at LaBranche.

A paired t-test was used to test the effects of the sediment fence on wave energy and significant wave height. The fence was found to significantly reduce wave energy and wave height at both field sites. The sediment fences caused an average loss of significant wave height of 50±3 % (R^2= 0.95) as a wave train passed through the fence ($p< 0.001$). This corresponded to an average energy loss of 72± 3% (R^2=0.84).

SURFACE ELEVATION RESPONSE: Results of SET measurements at all stations are presented in Table 5.2 and are shown in Figure 5.5. Mean elevation change at all stations monitored ranged from +3.3 to -0.8 cm yr^{-1}. The sediment fences caused elevation increases at all SET stations with the largest effect at stations with the longest fetch seaward of the structures (1.38 ± 0.38 cm-yr^{-1}; $p< 0.0007$) and a somewhat lower effect at fence stations with small fetch (1.18 ± 0.38 cm-yr^{-1}; $p< 0.0032$). Control stations distant from the fences decreased in elevation (-0.73 ± 0.38 cm-yr^{-1}; $p< 0.0624$). Elevation increases proved to be significant, while stations that decreased in elevation did not differ significantly from a 'no change' hypothesis. Elevation changes measured at stations in Leeville were not significantly different from those measured at LaBranche. No significant seasonal elevation trends were identified although hurricane Andrew produced obvious short-term increases at most monitoring stations.

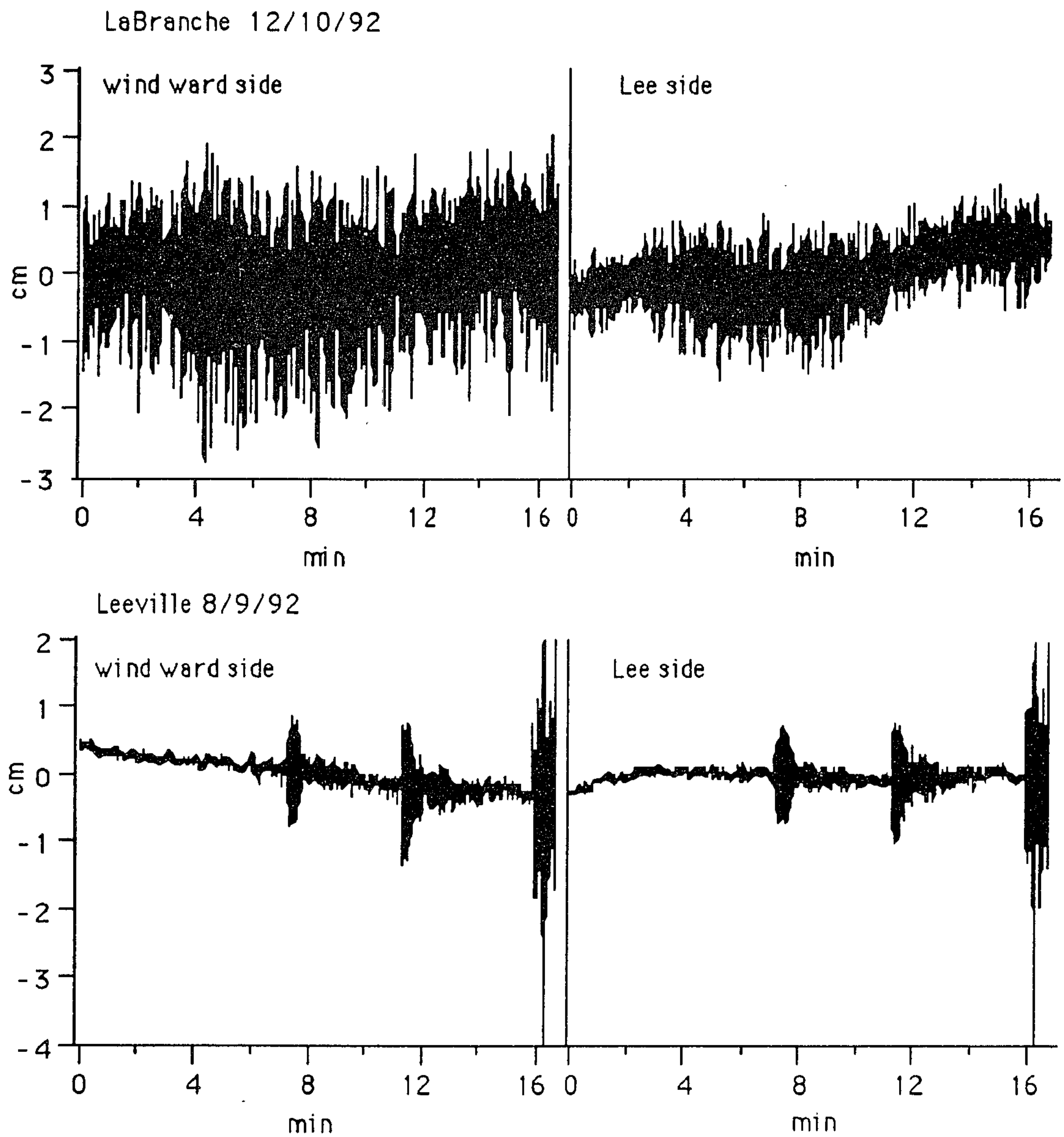

Figure 5.3 Two examples of the 17 Minute wave records sets sampled simultaneously at opposite sides of the fences. The laBranche 12/10/92 records show the effect of the sediment fence on the largess waves measured. The Leeville 8/9/92 records show the effect of the fence on Boat wakes.

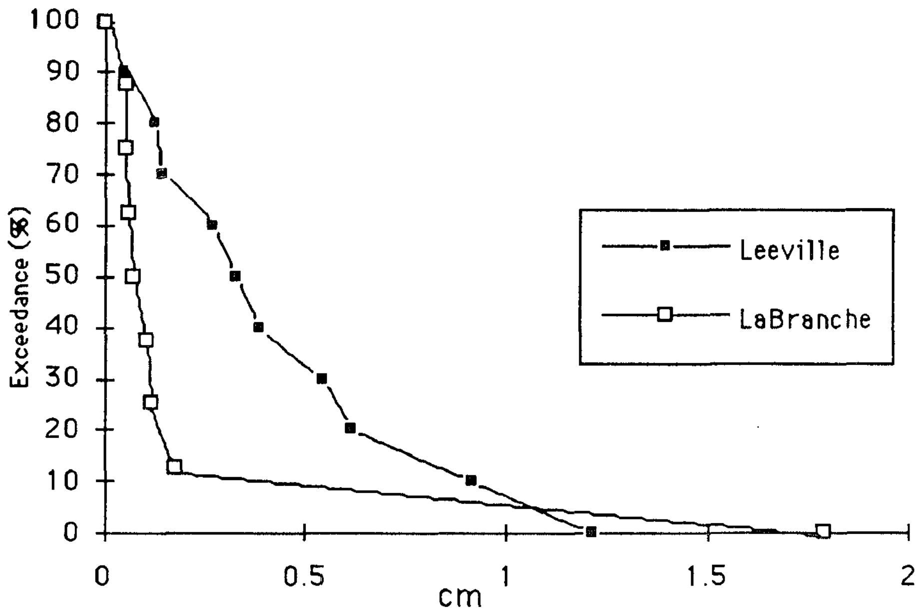

Figure 5.4 The percentage of wave measurements larger than the wave height plotted on the x-axis for Leeville and LaBranche.

Table 5.2 Linear increases in cm•y $^{-1}$ estimated for individual SET stations.

		R^2	elevation change (cm/yr)	Standard error
LaBranche				
	Control North	0.32	-0.4	0.08
	Control South	0.21	-0.7	0.17
	Pond North	0.01	0	0.07
	Pond South	0.49	0.7	0.09
	Mud flat North	0.6	2.8	0.3
	Mud flat South	0.71	1.7	0.14
Leeville				
	Outside	0.88	2.2	0.37
	Inside	0.56	3.3	0.23
	Pond	0.6	0.7	0.1

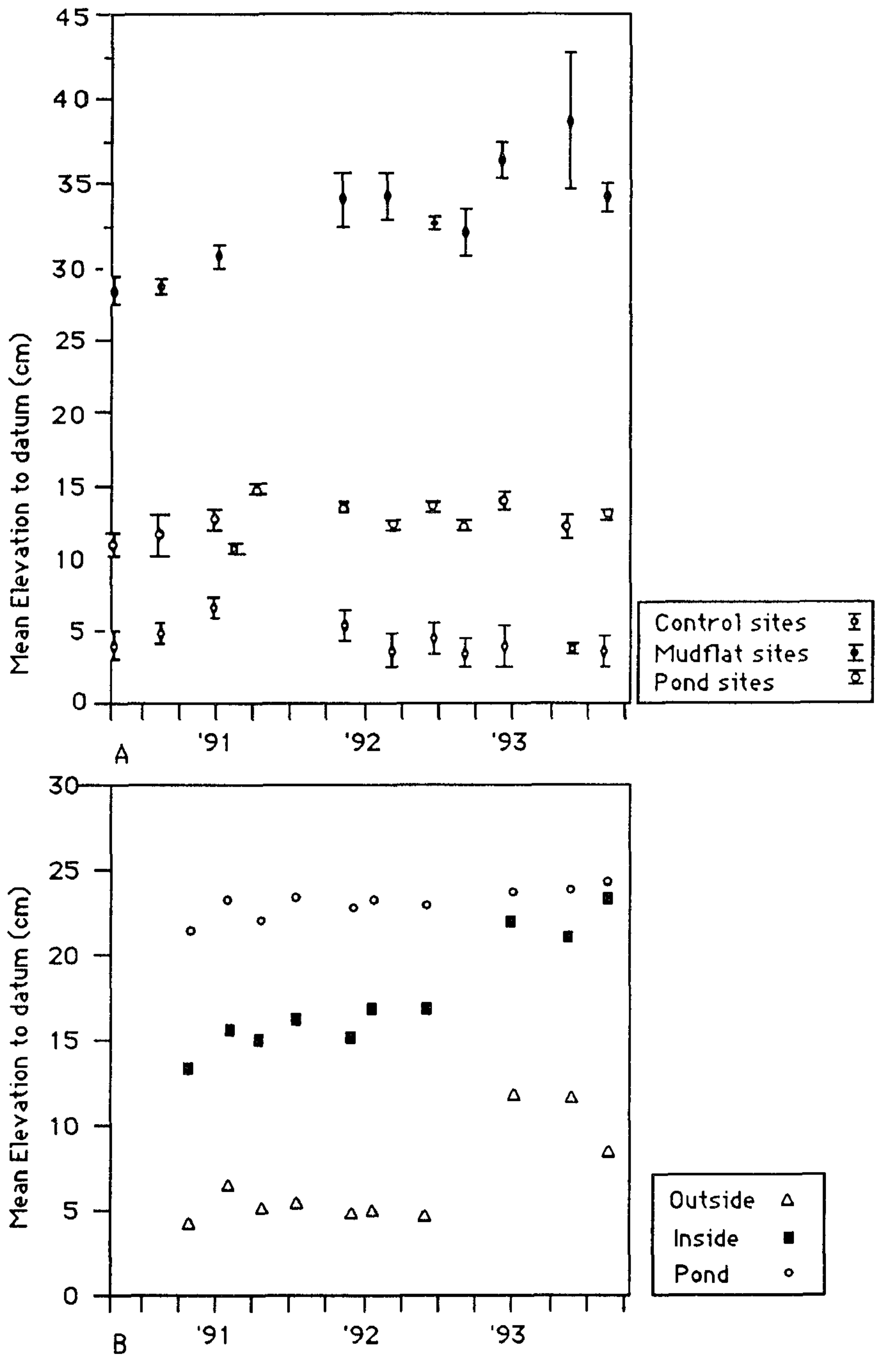

Figure 5.5 SET Elevations measured at the LaBranche (A) and Leeville areas.(B).

VEGETATION: No emergent vegetation has appeared at the Leeville site four years after construction. In contrast, the first evidence of re-vegetation was observed at the LaBranche site in May, 1992, about three years after initial construction. A fresh and intermediate marsh community colonized part of the LaBranche fence that was built on the former flood delta at the eastern end of the structure. Later that year in August, patches of Spartina alterniflora were observed on the mud flat on the north side of the fence. Hurricane Andrew (August 26 1992) removed trees from the western part of the fence in deeper water which had no vegetation. In November, 1992, the vegetation on the previously colonized parts of the fence had been replaced primarily with dwarf spike rush (Eleocharis parvula) and, to a lesser extent, with Spartina alterniflora. At the same time, an abundant growth of submerged aquatics was observed in the areas around the empty cribs on the west end. By the spring, 1993, the submerged aquatics had spread and were present throughout the whole area. By July, 1993, however, the submerged aquatics had disappeared while the Eleocharis parvula covered the entire expanse of the previously unvegetated shoal area on the east end. On September 1993 it was observed that most of the Eleocharis parvula had been replaced with sparsely distributed Pluchea camphorata. Over time walking on the former flood delta became easier because of a more compacted sediment substrate

DISCUSSION

The results show that sediment fences can reduce wave energy, cause increases in surface elevation, and potentially lead to marsh vegetation establishment. It is believed that the reduction in wave energy results in reduced sediment resuspension, enhanced deposition, and consolidation of the surface sediments (Anderson et al. 1981; Nagai et al. 1984; Ward et al. 1984; Shibayama et al. 1986)

WAVES: Wave transmission through any structure is a complex function of the wave conditions and structure width, size, permeability, materials and water depth (USACOE 1984). Accordingly, a plot of the wave transmission coefficient ($H_{incident}/H_{transmitted}$) is more useful than any single estimate. The data obtained at the two sites are shown in Figure 5.6 following USACOE (1984). It is important to note that despite some minor differences in construction, both fences functioned similarly with respect to transmissivity. The following empirical relation was derived and can be used for future design :

$$K_T = H_i/H_t = 0.92 - 0.13 \log(H_i/gT^2)$$

where K_T is the transmissivity coefficient.

It can be seen that sediment fences function quite similarly to more traditional permeable breakwaters in that they function most effectively to dampen the steep, locally generated wind waves that are responsible for most erosion in the shallow water bodies in which these structures are placed. In contrast, waves with small heights and low frequencies are hardly affected.

ELEVATION: Changes in elevation at the SET stations followed either a general increase at stations adjacent to the fences or were subject to erratic elevation fluctuations. Erratic short term fluctuations with no long term trends were observed away from the fences and on the elevated mud flat in the LaBranche area (Table 5.2).

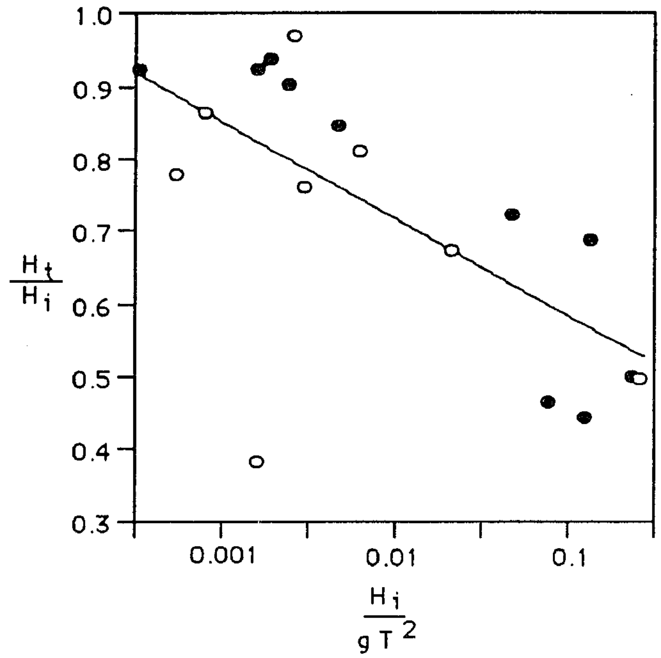

Figure 5.6 Wave transmission for sediment fence breakwaters in Leeville(•) and St. Charles (o). H_i is the wave height of the approaching wave and H_t is the wave height after passage. T is the wave period and g is gravity (USCOE 1984).

Most studies report sediment accretion rates rather than actual changes in sediment elevation. These rates, based on artificial or natural marker horizons are biased against negative elevation changes as they fail to record erosional events and subsidence (Anderson et al. 1981) this in contrast to the SET (Boumans and Day 1993). SET elevation changes can be compared to accretion records if no erosion is assumed and an estimate for subsidence is added. A typical subsidence rate for the Louisiana coastal zone is 1 cm•yr^{-1} (Day and Templet 1989; DeLaune et al. 1990; Nyman and DeLaune 1990). The estimated accretion rate for the non-fence stations (0.27 cm/yr) was similar to the low accretion values reported by (Childers et al. 1993),Letzsch and Frey (1980) and (Day and Templet 1989), while the deposition values at the fence stations (2.3 cm/yr) approached the higher reported accretion rates. The extremely high accretion rates of greater than 10

cm/year, occasionally reported with sediment trapping management strategies were not observed. (Glopper 1981; Stevenson et al. 1985 ; Bouwersma et al. 1986; Dijkema et al. 1990).

The fences were most likely to succeed in enhancing plant colonization when placed on mud flats with an elevation around mean water level within the tidal range. Elevation increases were slow at the low, barely inter tidal elevations such as the LaBranche pond sites (Table 5.2), but also on the mud flat sites at LaBranche once they were elevated to sustain vegetation (Figure 5.2). The high initial elevation within the tidal range of the north mud flat combined with the 2.8 $cm \cdot y^{-1}$ increases raised the LaBranche mud flat in two years high enough to allow colonization of marsh plants. In the spring of 1991 the elevation increase slowed down and changes became more erratic (Figure 5.5).

A higher elevation in the tidal range improves the drainage crucial for plant colonization (Joenje 1978; Eerdt 1985; Burdick and Mendelssohn 1987; Shaffer et al. 1992)] in spite of the associated loss of elevation because of compaction (Meade 1966). Evidence for elevation loss due to compaction was observed at the mud flat in LaBranche where vegetation colonized the area after an episode of elevation decrease (Figure 5.5).

The Leeville inside station eventually will be colonized because of the on going increase in substrate elevation. It has not yet reached the conditions for plant colonization because it lacks the distinct episodes of elevation loss through compaction. Similar episodes of elevation loss before plant colonization were observed by (Bouwersma et al. 1986). The increase in elevation after hurricane Andrew for the fence stations in Leeville could only be sustained by the inside station which in contrast to the outside station is sheltered by the fence from most of the waves.

VEGETATION: Similar elevation requirements for plant colonization are not expected for the LaBranche and Leeville areas as they are different located within different wetland habitat zones. The SET stations will have to be revisited and surveyed with a GPS system to estimate the elevation at which to expect vegetation to colonize a newly elevated mud flat for different plant communities associated with these wetland ecosystems (Sasser 1977).

MANAGEMENT IMPLICATIONS: Inter tidal sediment fences do not of themselves reverse the larger scale processes that have led to wetland loss, nor do they affect submergence potential in any way. It is likely that the most widespread use of inter tidal sediment fences in the future in Louisiana will not be in a stand-alone mode for low-cost shoreline protection but to hasten the appearance of vegetation in areas receiving new inputs of suspended clays. In this context, sediment fences can be designed to promote deposition at preferred locations and sculpture the landscape. If placed well, such fences will become vegetated and buried before refurbishment is necessary and the marsh formed on the platforms created will continue to expand as the plants capture sufficient sediment to offset the subsidence processes.

WATER DEPTH. Potential areas for fences need to have a high enough initial sediment elevation to expect re-vegetation before deterioration of the fence. In LaBranche the required elevation increase was 5 cm which was accomplished after two years. In Leeville the increase in three years was 10 cm and the elevation for re-vegetation is not reached yet. In Louisiana, Christmas trees decompose in approximately three to four years. Consequently, the fences could be restocked with trees until vegetation on the fences is observed. Because of the apparent non-linear nature of the processes involved for choosing the optimum elevation, simulation modeling (Wiegert et al. 1975) is advised to assure an elevated mud flat before the decomposition or destruction of the fence.

SEDIMENTS: For sediment fences to increase elevation there must be a supply of suspended sediments. The measured suspended sediment concentrations at the two sites were rather low (Table 5.1) and equaled concentrations reported for estuaries during fair weather conditions which are not directly fed by river discharge (Anderson 1972; Stumpf 1983). Neither of the two sites are recipients of river discharges, so the only regular source of sediments is local resuspension during storms (Stevenson et al. 1985). The reduced wave energy behind the fences created a spatially preferred location for sedimentation. The effect of sediment fences on marsh creation can be improved by providing sediment sources in addition to local resuspension.

Re-vegetation in these examples occurred without planting. Potentially, one could consider planting after a certain elevation and sediment consolidation has been reached and re-vegetation has not been observed. Other studies have shown further elevation increases when vegetation was present due to organic matter production (Nyman and DeLaune 1990) and consequently more reduction of the wave energy (Knutson et al. 1981; Benner et al. 1982; Knutson et al. 1990; Nyman and DeLaune 1990). .

CHAPTER 6

MODELING IMPACTS OF MANAGEMENT MEASURES ON FRAGILE COASTAL WETLANDS

INTRODUCTION

The alarming rate of wetland loss in Louisiana mentioned in previous chapters has generated many ideas on marsh management strategies two of which, have been discussed in more detail. These two strategies, marsh management and the construction of sediment fences, differ in the processes which they tend to regulate. Marsh management favors the idea that habitat can be restored by regulating the hydrology, while brush fences alter wetland sediment dynamics. Other strategies, like fresh water diversion and distribution of dredged materials (Suhayda et al. 1991) are designed to increase suspended sediment loads.

I constructed a computer simulation model (Wiegert et al. 1975) to test the effect of each of these four strategies individually and in combination on marsh habitat improvement in a Louisiana coastal wetland. In the model, marsh habitat is favored against open water at higher elevations (Chapter 2) with fairly compacted or mature substrates of clays and organic materials (Chapter 4; (Chabreck 1972)).

MODEL HISTORY: The model was developed to elaborate on the processes summarized by Ksed in the PBS model (Chapter 2), and was updated following new observations in conjunction with the flux (Chapter 3) and sediment fence experiments (Chapter 5). The model was adapted to the General Ecological Model (GEM) conventions (Fitz et al. in press) to include the option for spatial simulation. GEM is a landscape modeling concept presently being developed by a group of scientists from different universities in the United States. The effort is based on earlier work of spatial modeling published by (Costanza et al. 1988). Although GEM is still in the development stage, the eventual result will be a generic spatial modeling package where processes are initiated by dynamic STELLA™ models, and initial conditions are imported by

geographical information systems. GEM output will be in a GIS format to allow spatial calibration and verification of the simulated results.

METHODS

MODEL DEVELOPMENT: The general conceptual model representing the dynamics of wind, water, soil and vegetation is presented in Figure 6.1. It includes five state variables: Soil organic matter, Soil mineral matter, suspended minerals, pore space and wave energy. It was translated for mathematical representation in the STELLA™ modeling language (Appendix B). State variables were calculated in units of weight (KG for suspended and deposited sediments), volume (m^3 for pore space) and wave energy (Joule per m^2).

DYNAMICS: Two state variables, wave energy and pore space represent hydrodynamics and soil formation in the model. Wave energy, calculated using the Linear Wave Theory (USCOE 1984), was lost to bottom friction, and was gained by wind stress. Pore space was lost to compaction, and gained with newly deposited sediments and bottom friction.

Wave energy generates a wave shear stress which is the main potential erosional force on the deposited substrates (Grant and Madsen 1979). Loss of wave energy through bottom friction occurs mainly under shallow water conditions, reached when the wave height to water depth ratio exceeds 0.78 (USCOE 1984). An additional 73% of wave energy is dissipated when sediment fences are modeled (Chapter 5). Wave energy export is proportional to the wave group velocity, the wave energy and traveling distance within a cell (eq 3-39 in USCOE 1984).

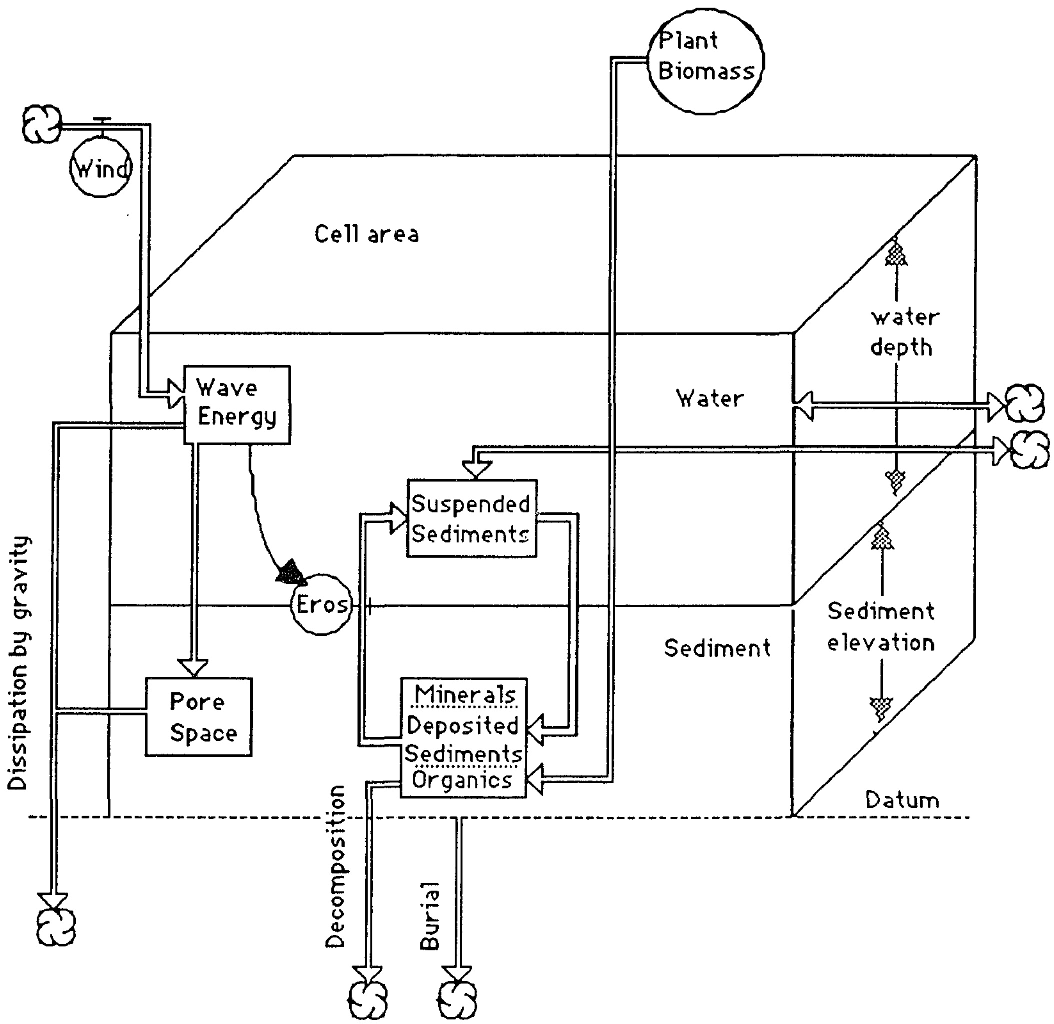

Figure 6.1 The conceptual representation of the GEM hydrodynamic and flux modules.

Pore space in wetland soils occupies a large percentage of the total soil volume (Meade 1972). Increases in pore space are associated with new deposits, while decreases occur through compaction as the clay sediments mature. Drainage accelerates maturation during non flooded conditions (Pons and Molen 1973). Energy transfer from wave energy into the soil counteracts compaction by keeping soil particles in motion (Clukey et al. 1985; Chou et al. 1991).

Erosion was estimated proportional to the wave shear stress exceeding the soil-shear-resistance. The soil-shear-resistance (τ_r in kg/m^2) incorporates root mat density (%R), the organic content of the soil (%OM), and the percent pore space or porosity (%P) (eq 1). Further research will have to test and calibrate this relationship against appropriate data sets.

eq 1

$$\tau_r = 0.5 \bullet \frac{1 + \%R}{1 + \%OM + \%P}$$

Deposition occurs when the strength of the suspended sediment concentration exceeds the shear stress (McCave 1984; Kemp 1986).

FLUXES: The model includes fluxes for water and for suspended and deposited sediments. The detailed hydrological module in GEM, which simulates water fluxes between adjacent cells, was replaced with a single water level forcing function to simulate water exchanges. Positive discharges import water with constant suspended sediment concentrations, while negative discharges export water with simulated concentrations. Deposition and erosion regulate the vertical sediment flux between the suspended sediments and the deposited sediments. An alternative vertical drain on deposited sediments is sinking below the datum, due to subsurface processes such as subsurface compaction faulting and geological down warp.

The budgets for soil organic matter and soil mineral matter were similar, except for below ground decomposition. Organic matter added to the soil is habitat dependent and represent typical litter fall values. Eroded organics are exported from the model and not further accounted for.

MODEL APPLICATION: I calibrated the model parameters to achieve the system's behavior observed in the earlier chapters.

Model output was total change after a 100 h period, calculated in hourly time steps. Simulated elevation change and sedimentation were extrapolated to yearly rates and compared to SET readings (Chapter 4) and short term sedimentation rates (Chapter 3) respectively. Water level changes were simulated based on typical tidal ranges (Chapters 3 and 5), while the wind speed and wind direction data sets were collected at the New Orleans airport, and made available by the Louisiana State Climatology office. A previous run of the model (Coastal zone 1993) included data collected during Hurricane Andrew (August 26 1992) and covers the period between 2:00 pm August 12,1992 and 12:00 am August 30 1992 at using hourly water levels and wind data collected in the LaBranche wetlands. Some results of this run will be used to discuss the pulsed effect of storms on wetland elevation.

CALIBRATION: The model was calibrated to not exceed 10 cm/yr in elevation changes to resemble the same order of magnitude measured with the SET on inter tidal mud flats (Chapter 5). Table 6.1 shows the initial conditions and rate constants after calibration. This type of calibration represents the "ballpark" level, or preliminary phase of model development (Fitz et al. in press). A model calibrated at this stage should allow the initiation of sensitivity analyses and the process of refining the model with more specific data sets.

SIMULATION: The sensitivity analyses was directed to test the effects of the previously mentioned management options on changes in elevation, porosity and the total amount of sediments trapped. Simulations with a 76% wave energy reduction mimicked the impact of brush fences (Chapter 5), eliminating tidal range represented the Louisiana marsh management options (Chapter 3), while increased levels of suspended sediments represented sediment diversion techniques (Table 6.2). I also compared the effect of lower water levels and decreased porosities on further

developments of sediment elevation and soil compaction, as discussed in chapter 5.

RESULTS

After calibration, the results of the model indicate a fairly realistic representation of elevation change in inter tidal coastal wetlands. Sensitivity analysis proved to be useful in understanding the importance of the individual dynamics and fluxes at different elevation levels within the tidal range.

INCREASED SEDIMENT FLUX: The modeled system proved to be very sensitive to increased fluxes of suspended sediments. Elevation increases could only occur with the addition of new suspended materials, which made the sediment diversion and distribution techniques the most successful (Table 6.2). This conclusion is in agreement with chapters 2 and 3 and with the conclusions of (Baumann et al. 1984; Templet and Meyer-Arendt 1988; Penland et al. 1990). The newly deposited sediments increased sediment porosity and created unmature substrates. The simulated rates of sediment deposition were an order of magnitude larger then the short term deposition rates observed in chapter 3, but fitted well within the findings of (Reed 1992). The suspended sediment concentrations at Rockefeller and Fina were lower than the higher values initiated in the model. (Reed 1992) reported short term depositions similar to the model values at times when increased suspended sediment loads are expected.

DECREASED TIDAL RANGE: The elimination of the tidal range had the most negative effect on elevation (Table 6.2), which indicates that the Louisiana marsh management technique is the least effective option to promote sedimentation. Sediment flux did not occur, and no new sediments were trapped. A general loss in elevation occurred through compaction. The decrease of sediment flux that results in a decrease of trapped sediments because of

smaller tidal ranges is supported by the findings in chapter 3 and also by (Reed 1992).

Table 6.1 Initial conditions and rate constants after calibrating the GEM hydrodynamic and sediment flux module for shallow inter tidal water bodies. at hourly time steps. GEM is a modeling approach with the use of computer simulation (Wiegert et al. 1975)

Model Component	Parameter	value	units
State variables:	Wave_Energy	0.5	J/cell
	Suspended Matter	0	kg/cell
	Soil mineral matter	65000	kg/cell
	Soil organic matter	5500	kg/cell
	Pore space	20	m3/cell
Soil sediment	Sediment elevation	50	cm
	cell_area	100	m2
	Mineral matter density	2.6	$kg \cdot l^{-1}$
	Organic matter density	1.1	$kg \cdot l^{-1}$
Flux	Mean water depth	30	cm
	Tidal range	15	cm
	TSS of in coming water	200	$mg \cdot l^{-1}$
Rate constants	Compaction:		
	Loss of pore space (flooded)	$6.7 \cdot 10^{-5}$	$\% \cdot h^{-1}$
	Loss of pore space (drained)	$6.7 \cdot 10^{-4}$	$\% \cdot h^{-1}$
	Decomposition:		
	Loss of Soil organics	$1 \cdot 10^{-6}$	$\% \cdot h^{-1}$
	Downwarp	0	$cm \cdot y^{-1}$

FENCE EFFECTS: Simulating the effect of a fence as only a decrease in wave energy (Figure 6.2, Table 6.2) caused slightly accelerated

increases of deposited sediments and consequently increased sediment elevation. This agrees with the general trend of accelerated increases in elevation adjacent to fences (Chapter 5) but does not account for the much larger differences that were measured. The output also did not indicate any maturing of the substrate under influence of the fence as was observed in chapter 5. Probably the 100 h time period was is too short to show such an effect (Meade 1966).

Elevation increases accelerated after the fence treatment included decrease of wave energy and a more compacted or matured substrate (Chapter 5, Table 6.3). The lower subsidence rates associated with the more compacted soils allowed for accelerated elevation increases.

The effect of the fence simulated as the decrease in wave energy plus an elevated surface was simulated by initiating unconsolidated substrate at lower mean water depths. This slowed the elevation increase, lowered the sediment load trapped and increased pore space in the substrate (Table 6.4). Similar findings of low elevation increases or even decreases were observed in chapter 5 and by (Bouwersma et al. 1986) and (Anderson et al. 1981). The lower water depths cause a change in erosional potential by the smaller waves while shorter flood durations and lesser floodwater volumes lower the suspended sediment flux. The increased compaction during the longer dry periods in the simulation was not sufficient to offset the increased erosion. The higher elevation could not be sustained.

Simulating the sediment fence effect as the elevated mud flat with mature substrate and lower wave energies showed lower accretion rates and lower elevation increases than non-fenced areas, but not as dramatic compared to the unmature substrate (Table 6.5). The sediments during this simulation period matured and became less susceptible to erosion.

Table 6.2 Simulations of total sedimentation, changes of elevation and soil porosity were designed to test aspects of three management approaches. Different total suspended sediment concentrations (TSS 0-200 mg•L^{-1}) compare high versus low new sediment input through freshwater diversion or dredge spoil distribution. A typical versus no tidal range (30-0 cm) tests the reduced water level change effect of marsh management. High versus reduced wave energies (100-28%) tests the impact of sediment fences.

	Model conditions							
TSS (mg/l)	0	0	0	0	200	200	200	200
Wave energy (%)	100	100	28	28	100	28	28	28
Tidal range (cm)	30	0	30	0	30	30	0	0
Model output:								
Elevation change cm/y	-0.4	-0.4	0	-0.4	6.3	6.4	-0.4	-0.4
Porosity change %10-4	-6	-6	10	-6	10	10	-6	-6
Trapped sediments g/m2/d	0	0	0	0	27	28	0	0

Initial soil porosity was 40 % and initial mean waterlevel was 30 cm

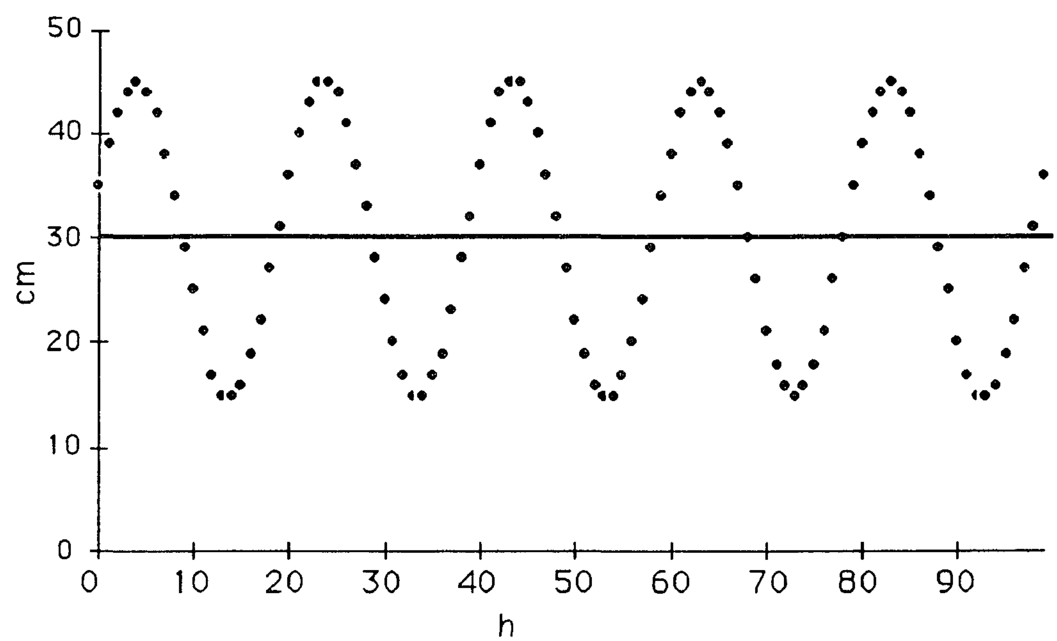

Figure 6.2 Simulated water levels with tides (•) and without tides(-).

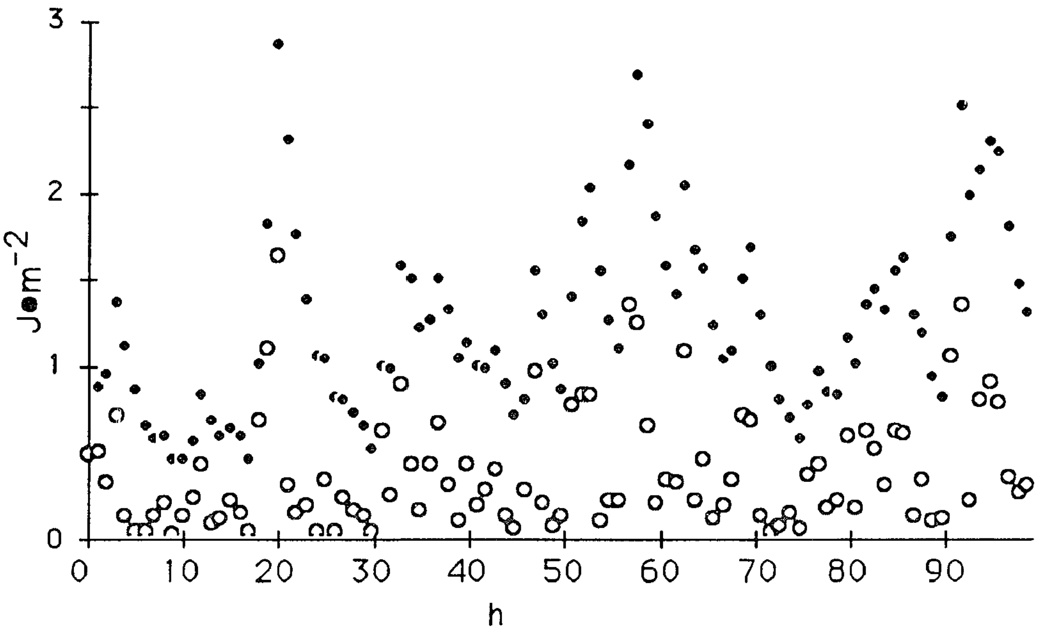

Figure 6.3. Simulated wave energies without sediment fences (•) and with sediment fences (◦).

Next to light and nutrients, the capability to withstand the physical energy in the water column during flooding periods is the most important environmental condition necessary to ensure colonization and the survival of marsh vegetation (Van Eerdt 1985; Foote and Kadlec 1988). I would argue that this resistance is greatly improved with a more compacted soil to provide matrix for the plant roots therefore, plant growth and survival is enhanced

THE EFFECT OF STORMS: The model, initiated to simulate hurricane Andrew (Boumans and Day 1993), supported the idea that the more densely vegetated marshes and wave protected mud flats are the most effective habitat with respect to trapping sediments during storms. The maximum simulated elevation increase in the model was 13.6 cm/year, similar to our observations in Leeville after hurricane Andrew (Chapter 5). Considering the simulations presented here, these storm deposits can only be incorporated into the substrate if a period for maturing occurred, as stimulated by sediment fences and drainage during non-flooding conditions.

Table 6.3 Simulations of total sedimentation, changes of elevation and soil porosity were designed to test aspects of three management approaches. Different total suspended sediment concentrations (TSS 0-200 $mg \cdot L^{-1}$) compare high versus low new sediment input through freshwater diversion or dredge spoil distribution. A typical versus no tidal range (30-0 cm) tests the reduced water level change effect of marsh management. High wave energies with increased pore space versus reduced wave energies with reduced pore space test the impact of sediment fences.

	*Model conditions								
	TSS (mg/l)	0	0	0	0	200	200	200	200
	Wave energy (%)	100	100	28	28	100	100	28	28
	Tidal range (cm)	30	0	30	0	30	30	0	0
	Init.por. (%)**	60	60	20	20	60	60	20	20
Model output:									
Elevation change (cm/y)		-1.1	-1.1	0	-0.1	5.8	-1.1	6.7	-0.1
Porosity change (%10-4)		-2	-2	-34	-10	36	-2	-59	-10
Trapped sediments (g/m2/d)		0	0	0	0	27	0	27	0

*Initial mean waterlevel set to be 30 cm.

**Includes lesser pore space as a fence effect.

Table 6.4 Simulations of total sedimentation, changes of elevation and soil porosity were designed to test aspects of three management approaches. Different total suspended sediment concentrations (TSS 0-200 $mg \cdot L^{-1}$) compare high versus low new sediment input through freshwater diversion or dredge spoil distribution. A typical versus no tidal range (30-0 cm) tests the reduced water level change effect of marsh management. High wave energies at lower elevation versus reduced wave energies higher elevation test the impact of sediment fences.

	*Model conditions							
TSS (mg/l)	0	0	0	0	200	200	200	200
Wave energy (%)	100	100	28	28	100	100	28	28
Tidal range (cm)	30	0	30	0	30	30	0	0
Init.WL (cm)**	30	30	10	10	30	30	10	10
Model output:								
Elevation change (cm/y)	-0.4	-0.4	-2	-0.4	6.3	-0.4	3.9	-0.4
Porosity change (%10-4)	-21	-2	1	-10	-15	-2	8	-10
Trapped sediments (g/m2/d)	0	0	0	0	27	0	21	0

*Initial soil porosity set to be 40%

**Includes lower waterlevels as a fence effect.

Table 6.5 Simulations of total sedimentation, changes of elevation and soil porosity were designed to test aspects of three management approaches. Different total suspended sediment concentrations (TSS 0-200 $mg{\bullet}L^{-1}$) compare high versus low new sediment input through freshwater diversion or dredge spoil distribution. A typical versus no tidal range (30-0 cm) tests the reduced water level change effect of marsh management. High wave energies with increased pore space at lower elevation versus reduced wave energies with reduced pore space at higher elevation test the impact of sediment fences.

	*Model conditions							
TSS (mg/l)	0	0	0	0	200	200	200	200
Wave energy (%)	100	100	28	28	100	100	28	28
Tidal range (cm)	30	0	30	0	30	30	0	0
Init.WL (cm)*	60	60	20	20	60	60	20	20
Init. Por(%)*	30	30	10	10	30	30	10	10
Model output:								
Elevation change (cm/y)	-0.4	-0.4	-2	-0.4	6.3	-0.4	3.9	-0.4
Porosity change (%10-4)	-21	-2	1	-10	-15	-2	8	-10
Trapped sediments (g/m2/d)	0	0	0	0	27	0	21	0

*Includes lower waterlevels and lesser pore space as a fence effect.

CONCLUSIONS

1) Model simulations predicted the largest increases of elevation when initiated with a tidal range and a high suspended sediment influx.

2) Model results suggest that sediment fences are effective in creating new marsh habitat by enhancing soil formation through compaction, which was necessary to ensure elevation increases.

3) Simulated water level management with constant water levels now influx of new suspended sediments lead to elevation decrease and habitat loss

CHAPTER 7

GENERAL CONCLUSIONS

Elevations of inter tidal coastal wetlands are constantly changed by the processes of sediment flux, hydrodynamics, soil formation, plant growth and subsurface dynamics. The results of the study are in agreement with others (Baumann and DeLaune 1982; Templet and Meyer-Arendt 1988) who report that the loss of marsh habitat in Louisiana is caused by a general decrease in relative surface elevation of the coastal plain. Two factors which are critical to the restoration of marsh habitat are wave turbulence and the level of suspended sediments. Decreased turbulence and increased suspended sediment concentrations lead to increased marsh elevation and marsh re-establishment.

The sediment surface elevations of marshes, ponds and mud flats fluctuate around an equilibrium elevation determined by the wave climate and the tidal range, where the wave climate is limited by fetch. Wetland elevation is lowered by erosion and subsidence, and is raised by deposition and root mass production. A change in habitat (land gain or land loss) occurs when a change in wave climate tidal range or suspended sediment load alters the equilibrium elevation at a particular location.

Although a wave climate is generated by the atmospheric conditions of wind speed and duration, the wave erosion at a particular site depends on the geomorphology of the area because wave shear stress at a location depends on water depth and fetch.

Erosion and deposition occur at different ends of the wave shear stress distribution. Typically, higher wave stresses at the sediment surface during shallow water conditions cause erosion, while reduced stresses allow particles to settle for deposition.

Erosion is a function not only of amplitude and frequency of wind waves on pond bottoms, inter tidal flats and marshes, but also the shear resistance of the sediment substrate. Elevations higher in

the tidal range experience more severe wave stress when flooded under shallow water conditions, but also are more likely to increase in shear resistance through drainage and the growth of plant roots.

When wave turbulence is low, sedimentation strongly depends on the availability of suspended materials. The primary source of sediments in the areas studied was via tidal advection from surrounding areas. When tidal exchange was decreased, by water control structures, for example, sedimentation also decreased. The larger deposits are at lower elevations because of longer flood duration, lower wave shear stresses, and a higher suspended sediment supply associated with greater water depth.

Vegetation colonization was greatest on the more compacted substrate typical of higher elevations within the tidal range. Once vegetation is established it alters the wave climate which raises the elevation. The associated root mass production increases the soil shear resistance and the soil volume.

Storms had a pulsed effect on sediment elevation causing an increase above the equilibrium elevation. Such increases were not sustained unless fetch was decreased, as by sediment fences, for example.

RECOMMENDATIONS FOR FUTURE RESEARCH

FIELD STUDIES: Future efforts to add to the understanding of marsh deterioration and colonization versus the elevation developments should include next to elevation changes, the properties of stress resistance and rates of local subsidence. Additional research is needed to confirm or reject the use of linear wave theory in inter-tidal shallow waters

MODELING: The GEM hydrodynamic and flux modules need more sophisticated calibration. The sensitivity of the subsurface needs to be tested on the elevation development within GEM. Spatial representation of a calibrated GEM unit model is needed to test landscape specific management designs.

MANAGEMENT RECOMMENDATIONS

In compliance with my findings I formulated the following criteria to manage an area for encouraging submerged vegetation re-establishment in a deteriorated coastal marsh:

1) Increase the suspended sediment flux load by tapping a river source and allowing large water level variations.
2) Consider sediment fences to discourage erosion and enhance deposition and compaction on unvegetated mud flats.

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APPENDIXES

APPENDIX A

Table A.1 ANOVA tables to test the effect of coastal area and marsh management on water quality parameters.

	Source	Df	Type III sum of squares	F ratio	P>F
TSS (gr/l)	area	1	0.78771363	75.2516	0.0001
	treatment	1	0.18882644	18.0389	0.0001
	area*treatment	1	0.32307812	30.8642	0.0001
NaCl (gr/l)	area	1	5316.5658	108.4654	0.0001
	treatment	1	964.8823	19.6849	0.0001
	area*treatment	1	970.0013	19.7894	0.0001
NH4 (μMol/L)	area	1	487.92075	16.518	0.0001
	treatment	1	0.09079	0.0031	0.9558
	area*treatment	1	41.9809	1.4212	0.2343
$NO_{2/3}$ (μMol/L)	area	1	2757.2718	30.9713	0.0001
	treatment	1	918.4898	10.317	0.0015
	area*treatment	1	2043.1695	22.9501	0.0001
PO_4(μMol/L)	area	1	73.872825	44.239	0.0001
	treatment	1	10.354382	6.2008	0.134
	area*treatment	1	3.764825	2.2546	0.1344

Table A.2 ANOVA tables to test the effect of coastal area and marsh management on fluxed materials.

	Source	Df	Type III sum of squares	F ratio	P>F
H_2O	area	1	126.94508	9.0466	0.0169
(l/d•m^2)	treatment	1	29.17201	2.0789	0.1873
	area*treatment	1	100.24832	7.1441	0.0282
TSS	area	1	0.40911054	7.2906	0.0271
(g/d•m^2)	treatment	1	0.13977367	2.4908	0.1532
	area*treatment	1	0.1877751	3.3462	0.1048
NaCl	area	1	1789.3926	3.0092	0.121
(g/d•m^2)	treatment	1	71.8796	0.1209	0.7371
	area*treatment	1	58.5937	0.0985	0.7616
NH_4	area	1	3179.9106	2.37	0.1623
(µMol•d•m^2)	treatment	1	3174.4043	2.3659	0.1626
	area*treatment	1	2909.6438	2.1685	0.1791
$NO_{2/3}$	area	1	44646.657	1.8948	0.206
(µMol•d•m2)	treatment	1	39315.821	1.6685	0.2325
	area*treatment	1	46302.434	1.9651	0.1986
PO_4	area	1	58.233662	3.9405	0.0824
(µMol•d•m2)	treatment	1	1.401448	0.0948	0.766
	area*treatment	1	49.506438	3.3499	0.1046

Table A.3 ANOVA tables to test the effect of coastal area and marsh management on short term sedimentation.

	Source	Df	Type III sum of squares	F ratio	P>F
SSL(g/m2)	area	1	3418.1289	3.8558	0.0511
	treatment	1	2585.9527	2.9171	0.0894
	area*treatment	1	900.8537	1.0162	0.3148
CH2O(g/m2)	area	1	1063.4277	2.5267	0.1137
	treatment	1	666.0229	1.5824	0.21
	area*treatment	1	206.2613	0.4901	0.4848
Min(g/m2)	area	1	680.4314	6.8079	0.0098
	treatment	1	621.51502	6.1284	0.0142
	area*treatment	1	237.51802	2.3764	0.1249

Table A.4 ANOVA tables to test the effect of coastal area and marsh management on soil chemistry.

	Source	Df	Type III sum of squares	F ratio	P>F
PO4 (mg/kg)	area	1	40743.58	1.7887	0.1838
	treatment	1	175140.82	7.6888	0.0065
	area*treatment	1	94940.12	4.1679	0.0435
NaCl (mg/kg)	area	1	150873245	2.6782	0.1045
	treatment	1	59678623	1.0594	0.3056
	area*treatment	1	350574437	6.2232	0.0141
Organic matter (%)	area	1	55.58115	74.5268	0.0001
	treatment	1	5.980849	8.0195	0.0065
	area*treatment	1	16.60656	22.2671	0.0001

Table A.5 The ANOVA Table to test the effect of fetch, proximity to the sediment fence, field site, season and Hurricane Andrew (August 1992) on the change in elevation and the effect of hurricane Andrew on total elevation.

Source	DF	Type III SS	Mean Square	F Value	Pr > F
Time	1	1.3812809	1.3812809	0.43	0.5183
Treatment	2	14.2999839	7.1499920	2.21	0.1257
Time*Treat	2	37.0747946	18.5373973	5.72	0.0072
Season	3	18.3772952	6.1257651	1.89	0.1499
Time*Season	3	9.3039804	3.1013268	0.96	0.4243
Treat*Season	6	6.8493874	1.1415646	0.35	0.9037
Time*Treat*Season	6	12.9509494	2.1584916	0.67	0.6776
Area	1	0.0039116	0.0039116	0.00	0.9725
Time*Area	1	1.6074545	1.6074545	0.50	0.4861
Area*Treat	2	3.0141287	1.5070643	0.46	0.6321
Time*Area*Treat	2	8.1502541	4.0751270	1.26	0.2974
Area*Season	2	10.2588299	5.1294149	1.58	0.2203
Time*Area*Season	2	7.0707748	3.5353874	1.09	0.3475
Area*Treat*Season	4	12.2157552	3.0539388	0.94	0.4516
Time*Area*Trea*Seas	4	12.8103841	3.2025960	0.99	0.4273
Site(Area*Treat*Seas)	12	118.7997820	9.8999818	3.05	0.0052
Hurricane	1	13.1708294	13.1708294	4.06	0.0518
Area*Hurricane	1	9.6340436	9.6340436	2.97	0.0938

APPENDIX B

Stella equations associated with the GEM hydrodynamics and Flux modules

$initial conditions:

cell_size = 100	{M^2}
Cell_width = SQRT(cell_size)	
DayMod = TIME	
Hab = 1	Spatial habitat distribution
hrs_in_timestep = 1	Definition of the time step
ic_porosity = 0.4	
ic_Sed_elev = .5	
ic_V_perc_O = .1	
ic_Wave_Energy = 0	
mean_water_depth = 0.3	
min_density = 2600	
Org_density = 1100	

PhBio_GPP = IF Hab=1 THEN open_water ELSE IF Hab =3 THEN Emmerged_vegetation ELSE submerged_veg
rc_compaction = IF Water_depth >0 THEN 0.00000067 ELSE 0.0000067
rc_downwarp = 0
Out_TSS_C = 20 {equals 200mg/l}

Seasonal growth curves for the different habitat types:

BMP Emmerged_vegetation = GRAPH(DayMod)
BMP open_water = GRAPH(DayMod)
Root_mat_density = GRAPH(Hab)
submerged_veg = GRAPH(DayMod)
Wind_direction = GRAPH(DayMod)
Wind_speed = GRAPH(DayMod)

```
$DYNAMICS
PORE_SPACE(t) = PORE_SPACE(t - dt) + (Pore_sp_inc - Pore_sp_dec)
* dt
INIT PORE_SPACE = ic_porosity*ic_Sed_elev*cell_size

INFLOWS:
Pore_sp_inc = ((.8*DIS_frm_SIS)/2600)+0.01*Pore_exp
OUTFLOWS:
Pore_sp_dec = ((Porosity*rc_compaction)*PORE_SPACE)
+(Eros_Min*Porosity)/2600
WAVE_ENERGY(t) = WAVE_ENERGY(t - dt) + (gain - loss) * dt
INIT WAVE_ENERGY = .5

INFLOWS:
gain = If Obstruction=0 THEN
(Wave_Ex_E@W+Wave_Ex_S@N+Wave_Ex_N@S+Wave_Ex_W@E)+(1.
225*Fluid_density*(Loc_Wave_height^2)) ELSE
.27*(Wave_Ex_E@W+Wave_Ex_S@N+Wave_Ex_N@S+Wave_Ex_W@E)
+(1.225*Fluid_density*(Loc_Wave_height^2)))
OUTFLOWS:
loss = if Obstruction =1 THEN .7*WAVE_ENERGY
+max(to_friction+Wave_Export,0) ELSE
max(to_friction+Wave_Export,0)
current_corr = If orbital_velocity > Current_velocity THEN
Current_velocity/orbital_velocity ELSE 0
Current_direction = 0
current_velocity = 0
D_less_depth = (Water_depth*9.8)/(Wind_speed^2)
D_less_fetch = (Fetch*9.8)/(Wind_speed^2)
Eros = MAX(Pot_Eros-DELAY(Pot_Eros,1), 0)*Cell_size
Eros_Min = Eros*V_perc_M*2600
Eros_Org = Eros*V_perc_O*1100
Fetch = IF (Wind_direction = 1)OR(Wind_direction =
3)OR(Wind_direction = 5)OR(Wind_direction = 7) THEN
SQRT(cell_size) ELSE SQRT(cell_size)*SQRT(2)
```

group_velocity = IF (Wave_period <=0) OR (Wave_Length <=0) THEN 0 ELSE (.5*(Wave_Length/Wave_period)*(1+((4*PI*Water_depth)/((EXP((4*PI*Water_depth)/Wave_Length)-EXP(-(4*PI*Water_depth)/Wave_Length))/2))))
Loc_Wave_height = (.283*(Wind_speed^2)*depth_H_corr*fetch_H_corr)/9.8
Obstruction = 0
orbital_velocity = IF Wave_Length<=0 THEN 0 ELSE ((Wave_height*9.8*Wave_period)/(2*Wave_Length))*(1/((EXP((2*PI*Water_depth)/Wave_Length)+EXP(-(2*PI*Water_depth)/Wave_Length))/2))
Porosity = PORE_SPACE/Vtot
Sed_bulk_dens = sed_tot_wt/Vtot
sed_elev = Vtot/Cell_size
sed_tot_wt = SOIL_ORGANICS+SOIL_MINERALS
Shear_res = .2*(1+Root_mat_density)/(1+V_perc_O+Porosity)
Shear_stress = If orbital_velocity > Current_velocity THEN (.5*fric_coef*Fluid_density*(1+current_corr^2+2*current_corr*COS((ABS(Current_direction-Wind_direction)*.8)))*SQRT((orbital_velocity)^2)) ELSE 0
stress_effect = IF Shear_stress > Shear_res THEN Shear_stress-Shear_res ELSE 0
Surface_water = Water_depth* cell_size
Tide = .15*SIN(DayMod/(PI))
to_friction = IF Wave_height/Water_depth >.78 THEN .4*WAVE_ENERGY ELSE .2*WAVE_ENERGY
Vm = 1/((1/SOIL_MINERALS)*(min_density))
Vo = 1/((1/SOIL_ORGANICS)*(Org_density)) {volume of organic sediment with no pore space.}
Vtot = Vo+Vm+PORE_SPACE {m^3}
V_perc_M = Vm/Vtot
V_perc_O = Vo/Vtot
Water_depth = mean_water_depth+Tide
Wave_Export = group_velocity*WAVE_ENERGY

```
Wave_Ex_E = IF (Wind_direction >45) AND (Wind_direction<=135)
THEN Wave_Export ELSE 0
Wave_Ex_N = IF (Wind_direction >315) OR (Wind_direction<=45)
THEN Wave_Export ELSE 0
Wave_Ex_N@S = 0
Wave_Ex_S = IF (Wind_direction <=225) AND (Wind_direction<135)
THEN Wave_Export ELSE 0
Wave_Ex_S@N = 0
Wave_Ex_W = IF (Wind_direction >225) AND
(Wind_direction<=315) THEN Wave_Export ELSE 0
Wave_Ex_W@E = 0
Wave_height = SQRT((.835*WAVE_ENERGY)/Fluid_density)
Wave_Length = 1.56*(Wave_period^2)*SQRT(depth_L_corr)
Wave_period =
(7.54*(Wind_speed)/9.8)*depth_T_corr*fetch_T_corr
depth_H_corr = GRAPH(.53*(D_less_depth^.75))
depth_L_corr = GRAPH((4*Water_depth)/(Wave_period^2))
depth_T_corr = GRAPH(.833*(D_less_depth^.375))
fetch_H_corr = GRAPH( .00565*(SQRT(D_less_fetch))/depth_H_corr)
fetch_T_corr = GRAPH(.0379*(D_less_fetch^.333)/depth_T_corr)
fric_coef = GRAPH(current_corr)
Pore_exp = GRAPH((Shear_stress*Porosity)*.8*Vtot)
Pot_Eros = GRAPH(stress_effect)
rc_decomp = GRAPH(sed_elev)
Wave_Ex_E@W = GRAPH(DayMod)

$InorgSediments
SOIL_MINERALS(t) = SOIL_MINERALS(t - dt) + (DIS_frm_SIS -
DIS_dn_warp - SIS_frm_DIS) * dt
INIT SOIL_MINERALS = (1-
ic_V_perc_O)*ic_Sed_elev*Cell_size*min_density*(1-ic_porosity)

INFLOWS:
DIS_frm_SIS = IF Min_deposition =1 THEN .05*SUS_INORG_SED
ELSE 0
```

OUTFLOWS:
DIS_dn_warp = rc_downwarp*Cell_size*V_perc_M*(1-Porosity)*min_density
SIS_frm_DIS = Eros_Min
SUS_INORG_SED(t) = SUS_INORG_SED(t - dt) + (SIS_frm_DIS + SIS_X_in - SIS_X_out - DIS_frm_SIS) * dt
INIT SUS_INORG_SED = 0

INFLOWS:
SIS_frm_DIS = Eros_Min
SIS_X_in = Sed_to_S@N+Sed_input+Sed_to_W@E+Sed_to_N@S
OUTFLOWS:
SIS_X_out = MIN((Sed_to_S+Sed_to_E+Sed_to_N+Sed_to_W),SUS_INORG_SED)
DIS_frm_SIS = IF Min_deposition =1 THEN .05*SUS_INORG_SED ELSE 0
Fluid_density = SiS_conc +1000*(1-(SiS_conc/min_density))
Fl_mud_yield = 0.000049*((SiS_conc*10)^2.5)
Min_deposition = IF Shear_stress < Fl_mud_yield THEN 1 ELSE 0

Sed_input = MAX(Water_depth-DELAY(Water_depth,1),0)*Out_TSS_C
Sed_to_E = MAX((DELAY(Water_depth,1)-Water_depth),0)*SiS_conc*cell_size
Sed_to_N = 0*SiS_conc
Sed_to_N@S = 0
Sed_to_S = 0*SiS_conc
Sed_to_S@N = 0
Sed_to_W = 0*SiS_conc
Sed_to_W@E = 0
SiS_conc = IF Surface_water=0 THEN 0 ELSE SUS_INORG_SED/SURFACE_WATER
tot_sed_dn_warp = DIS_dn_warp+Sed_OM_dn_warp {kg/t}

$Org/Sed/Soil
SOIL_ORGANICS(t) = SOIL_ORGANICS(t - dt) + (Sed_OM_depo - Sed_OM_dn_warp - Sed_OM_decomp - Sed_OM_susp) * dt
INIT SOIL_ORGANICS =
ic_V_perc_O*ic_Sed_elev*Cell_size*Org_density*(1-ic_porosity)

INFLOWS:
Sed_OM_depo = GRAPH(Hab)
(0, 0) (0.8, 1e-07) (1.6, 1.5e-07) (2.4, 7e-07) (3.2, 2e-06) (4, 4.15e-06) (4.8, 6.6e-06) (5.6, 7.9e-06) (6.4, 8.35e-06) (7.2, 8.05e-06) (8, 7e-06)
OUTFLOWS:
Sed_OM_dn_warp = rc_downwarp*Cell_size*V_perc_O*Org_density
Sed_OM_decomp = rc_decomp*SOIL_ORGANICS
Sed_OM_susp = Eros_Org

APPENDIX C

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Estuaries Vol 16, No. 2, p 375–380 June 1993

High Precision Measurements of Sediment Elevation in Shallow Coastal Areas Using a Sedimentation-Erosion Table

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ABSTRACT: The use of a sedimentation-erosion table (SET) for the measurement of small changes in sediment surface elevation in intertidal and shallow subtidal areas is described. The SET provides a constant reference plane in space from which the distance to the sediment surface can be measured by lowering pins to the surface. The precision of the method was determined by repeated measurements in coastal marshes, mudflats, and subtidal areas in Louisiana and Georgia. The confidence interval of the SET is about ±1.5 mm. The SET is more accurate if pins are lowered manually and the sediment surface is determined visually than if pins are lowered by gravity and the sediment surface is determined by soil resistance.

Introduction

Sediment elevations in intertidal and shallow subtidal coastal areas are constantly changed by such factors as tides, storm activity, subsidence, and biotic activity. Annual elevation changes in coastal wetlands range from less than 0.01 cm yr^{-1} in stable sediment-starved areas to as much as about 5 cm yr^{-1} in areas with high sedimentation rates (Letzsch and Frey 1980; Anderson et al. 1981; Baumann et al. 1984; Rejmanek et al. 1988). There are longer term changes in sea level that lead to changes in the surface elevation in shallow coastal areas. For the last century there has been an average eustatic sea level rise of 1–2 mm yr^{-1} (Gornitz et al. 1982). In coastal areas with high subsidence rates, such as the Mississippi delta, relative sea level rise can exceed one cm yr^{-1} (Baumann et al. 1984). Coastal wetlands must grow vertically at the same rate as local water-level rise if they are to survive over the long term.

Precise methods for measuring these changes in the elevation of the sediment surface are necessary to determine the rates of elevation change and to gain an understanding of the processes responsible for the changes. Methods which have been used to study these changes include radiotracers such as ^{137}Cs (DeLaune et al. 1978), marker horizons (Letzsch and Frey 1980; Cahoon and Turner 1989), rare earth horizons (Knaus and Gent 1987), sedimentation pins (Letzsch and Frey 1980; Pethick and Reed 1987), and precision surveying (Anderson et al. 1981). These methods have several limitations and the accuracy of most of them has not been determined.

Marker horizons are not stable reference points and therefore generally not useful to measure elevation change (Baumann and Adams 1981; Cahoon and Turner 1989). Erosion pins measure sedimentation and erosion, but the pins are very sensitive to disturbance. Stakes and rods have been used to measure elevation changes at least since 1949 (Harbord 1949; Pestrong 1965; Reed 1989) but the accuracy has not been reported. Precision surveying can provide elevation change. Anderson et al. (1981) compared the means of 93 points on intertidal mud flats surveyed twice within a two-day period and calculated a 95% confidence interval (CI) of ±0.3 cm. The method would likely be much less accurate in wetlands and subtidal areas where the determination of the surface would be much more difficult.

In this paper we report on the use of a sedimentation-erosion table (SET, Fig. 1), a nondestructive method for precisely measuring elevation of intertidal and subtidal wetlands over long periods. Our design is derived from the SET used by Schoot and de Jong (1982), reported a standard error of 0.08 cm with a 95% CI of ±0.45 cm. In this paper we describe the precision of a more versatile SET for use in a wider variety of environments.

Methods

STRUCTURE OF THE SEDIMENTATION-EROSION TABLE

The SET has a supporting aluminum base pipe (10 cm diameter, 1 mm wall thickness) placed permanently at each site that is designed to receive the upper portable part of the SET (Fig. 1). This core pipe was driven into the soil to refusal using either a vibracorer or a hand-held pile driver as near to vertical as possible. The core pipe was then cut off a few cm above the sediment surface and filled to within a meter of the surface with quick-setting cement. The elevation of the top of the pipe after cutting will vary depending on water depth at the site and the tidal range.

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A polygon-based spatial (PBS) model for simulating landscape change

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Keywords: landscape ecology, wetland, swamp, marsh, succession, Lake Pontchartrain, Louisiana

Abstract

A spatial model of long term habitat succession at a degrading Louisiana wetland was constructed based upon simulating exchanges across irregularly shaped polygons. Polygons represented the natural morphology which is indicative of the natural landscape. The PBS model was partially successful in simulating spatial habitat changes over a 28-year period when more than 1000 ha of wetland loss occurred ($r^2 = .56$). General landscape trends did, however, emerge from the model development. Areas of high annual water level fluctuations, and high primary productivity were less likely to change from wetlands to open water and were most likely to recover if altered. We discuss the potential for predictive improvement and for integration with polygon-based geographic information systems, and conclude that a PBS model demonstrates the need for spatially explicit landscape management.

Introduction

Explicit incorporation of spatial parameters into ecological simulation models is important because of the strong spatial aspect of many ecological processes. As a result: decisions made by environmental regulatory agencies often require information on spatial and temporal responses (Weinstein and Shugart 1983, Risser *et al.* 1984, Sklar *et al.* 1985, Forman and Godron 1986, Turner 1987). For example; how will the placement of dikes and weirs alter fish migration, or how does the alteration of water flow patterns impact marsh productivity, or what types of coastal developments lead to habitat modifications (Chabreck 1972, Mendelssohn *et al.* 1983, Steiner 1983, Costanza *et al.* 1986)? These decisions are difficult to make without at least a conceptual model of spatial interactions. Then an important question is, how does one incorporate spatial interactions? To address this question we present a spatial model of a case study which uses wetland loss in Louisiana as the focus for analysis, irregular shaped grid cells (*i.e.*, polygons) as the spatial classification, and population interaction equations (May and MacArthur 1972; Maynard Smith 1974) as the basis for habitat succession.

Numerous researchers have documented that wetlands in Louisiana are diminishing at a rate estimated to be as high as 100 km^2 a year (Gagliano and van Beek 1970; Adams *et al.* 1976; Craig *et al.* 1979; Hopkinson and Day 1980; Gagliano *et al.* 1981; Boesch 1982; Scaife *et al.* 1983; Leibowitz *et al.* 1988; Turner and Cahoon 1987). Factors influencing land loss include: (1) rising sea levels, (2) subsidence of deltaic sediments, (3) nutrient and inorganic sediment depletion, (4) canal construction (5) salt water intrusion, and (6) decreasing bioproductivity. Attempts to model land loss as the

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The Journal
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Wetland Scientists

INSTRUCTIONS FOR AUTHORS

Authors are invited to submit manuscripts to the Editor-in-Chief to be peer-reviewed and considered for publication in *Wetlands*. Original articles dealing with freshwater, saltmarsh, or estuarine wetlands research from the viewpoint of any appropriate scientific discipline or from a management or regulatory viewpoint will be considered. The cover letter should either state that the information has not been published elsewhere or describe the form of such publication and provide justification for consideration.

Three copies of manuscripts, in English, must be double-spaced on one side of numbered 21.5 x 28 cm white paper, with 4 cm margins on all sides. The title, running head, abstract, keywords, headings, tables, and figures should all be clear and concise. Figures and tables should be on separate pages and referenced in the text. Do not send original figures or prints with the manuscript. Scientific names should accompany the first use of common names for all plant and animal species; taxonomic authorities should accompany all scientific names, either upon first use or in existing tables. **Literature citations in the text and in the Literature Cited section should be in the format used by the journal.** Examples are given below.

Text:

Weller (1988) or (Gosselink and Lee 1989) or Euliss et al. (1991) or (Wilcox and Simonin 1988, Wieder 1989, van der Valk et al. 1991)

Literature Cited:

Grace, J. B. and R. G. Wetzel. 1982. Niche differentiation between two rhizimatous plant species: *Typha latifolia* and *Typha angustifolia*. Canadian Journal of Botany 60: 46-57.

Lee, L. C. 1983. The floodplain and wetland vegetation of two pacific northwest river ecosystems. Ph.D. Dissertation. University of Washington, Seattle, WA, USA.

Mitsch, W. J. and J. G. Gosselink. 1986. Wetlands. Van Nostrand Reinhold Company, New York, NY, USA.

Richardson, C. J., R. Evans, and D. Carr. 1981. Pocosins: an ecosystem in transition. p. 3-19. *In* C. J. Richardson (ed.) Pocosin Wetlands. Hutchinson Ross Publishing Company, Stroudsburg, PA, USA.

Zedler, J. B. 1982. The ecology of southern California coastal salt marshes: a community profile. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC, USA. FWS/OBS-81/54.

The general style for all manuscripts should follow that for papers published in recent issues. The format for headings and sub-headings should be as follows:

MAIN HEADINGS

Sub Heading On Own Line

Sub-Sub Heading. The text then continues....

Sub-Sub-Sub Heading. The text then continues....

Instructions for preparation of the final version of manuscripts to be published will be provided upon acceptance. There are no page charges on final, accepted manuscripts submitted on computer disks in suitable word-processing format. **The preferred software for submission is IBM-compatible WordPerfect 5.1,** although other common software may be acceptable. High quality hard copies, scannable by computer, will have a $10 per page charge. Lengthy tables or poor letter quality manuscripts that cannot be scanned will necessitate additional charges.

Proofs will be sent directly to the author from the printer, together with an order blank for reprints. The author who receives the order blank is responsible for ensuring that orders are also placed for any coauthors. Corrected proofs should be returned within 48 hours to the Editor-in-Chief; reprint orders should be returned to the printer.

All manuscript submissions and correspondence regarding manuscripts, revisions, manuscript status, and editorial decisions should be directed to

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VITA

Roelof was born in Heerlen, The Netherlands, on August 26 1957. He received his high-school diploma from Het Zuid Limburgs Avond College, Heerlen, June 1979. He graduated with a Drs. in Biological Sciences from the University of Amsterdam, Amsterdam, The Netherlands, in October 1986, and entered The Marine Sciences Department at Louisiana State University in January, 1987.

DOCTORAL EXAMINATION AND DISSERTATION REPORT

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Major Field: Oceanography and Coastal Sciences

Title of Dissertation: Factors Affecting Sediment Transport, Deposition and Erosion in Intertidal Wetlands in Louisiana.

Approved:

Major Professor and Chairman

Dean of the Graduate School

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Date of Examination:

January 28, 1994