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Recent Research In Coastal Louisiana:

Natural System Function and Response to Human Influence

A Symposium
1998

Recent Research in Coastal Louisiana: Natural System Function and Response to Human Influence

A symposium convened by the
Louisiana Universities Marine Consortium
February 3-5, 1998
Lafayette, Louisiana

Edited by

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PREFACE

The coastal barriers, bays, marshes, and swamps of Louisiana compose one of the largest deltaic systems of the world. In contrast to many other deltas on Earth, where wetlands have been drained and barrier islands built upon, the Mississippi Delta Plain is still a highly productive ecosystem. It is, however, a system that had been dramatically altered from its natural state and a place where the challenges of managing and sustaining ecosystem function are faced every day. The research community in Louisiana has for many years been at the forefront in dealing with management issues and ensuring that decisions are made on the basis of the best available science. In the last decade with the advent of a large-scale wetland restoration program under the Coastal Wetlands Planning Protection and Restoration Act, with increased awareness of the potential consequences of river diversions, and with a renewed effort to exploit subsurface mineral resources in the coastal zone, it has become essential that the research community be proactive in disseminating the most current scientific understanding to coastal managers and decision makers.

With this in mind, we convened a broad-based committee from amongst the research community working in coastal Louisiana to design a coastal research meeting. The goal was to provide an opportunity for dissemination and discussion of research findings among scientists, managers, and policy makers. The result was the conference 'Recent Research in Coastal Louisiana: Natural System Function and Response to Human Influences'. The conference was held in February 1998 in Lafayette, Louisiana. Over three days, 48 oral and 20 poster presentations were made to an audience of over 150 people. Plenary presentations by Charles Simenstad on wetland restoration in the Pacific Northwest and Ron Jones on the challenges of ecosystem management in the Florida Everglades provided a context for us to consider the problems we face in coastal Louisiana.

The conference was a great success, and the papers in this volume represent a partial record of the discussions that occurred. Scientific advances will continue and our understanding of system function will only increase. In that sense, the information presented here is a snapshot of the state of coastal science knowledge in the late 1990's. There are few venues where managers and researchers can openly discuss their observations, findings, and doubts. Perhaps more enduring, therefore, is the recognition that we all share a common goal of a sustainable, productive coastal Louisiana and that the best efforts of all are needed to get there.

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HYDRODYNAMICS AND SEDIMENTARY PROCESSES

Historic Patterns of Sedimentation in Grand Lake, Louisiana

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ABSTRACT: When Europeans first explored Louisiana, Grand Lake, formerly called Lake Chitimaches, was a major feature of what is now the Atchafalaya Basin. The natural process of lake sedimentation and succession of Grand Lake has been accelerated through natural and man-induced hydrologic modifications of the Atchafalaya River and Basin. Today, only small remnants of this former lake ecosystem remain. Grand Lake has succeeded into a deep river channel and floodplain swamp ecosystem.

Historic information on the bathymetry of Grand Lake was analyzed along eight surveyed transect lines, termed rangelines. In 1917, during the earliest rangeline surveys, Grand Lake covered an area of 352 km^2 (136 mi^2). The rangeline profiles were surveyed at irregular time intervals over this century, and provide a dynamic picture of how sediments have filled and are filling Grand Lake. In 1917, average elevation above mean sea level (AMSL) of the most upstream and most downstream rangelines was -1.07 and -3.19 m (-3.5 and -10.5 ft), respectively. By 1978 the upstream rangeline had elevated to an average of 0.93 m (3.1 ft) AMSL; and downstream, in the 1980 survey, the rangeline had elevated to an average of -0.34 m (-1.1 ft). The rangeline survey data also document development of a deep channel through the lake.

In a first attempt to describe mechanisms and trends underlying the observed dynamics of bottom surface elevation, we developed a simple annual accretion model. This model assumes that annual accretion is proportional to annual volumetric water discharge, and that as average water depth increases accretion rate increases asymptotically toward a maximum. As bottom elevation rises, the model assumes that an equilibrium elevation is approached. At this equilibrium elevation, rangeline accretion is hypothesized to stop. Following calibration, the model was in close agreement with the observed rangeline elevation values.

The sedimentation of Grand Lake has significant implications not only for the Atchafalaya Basin ecosystem, but also for the Gulf of Mexico littoral ecosystem which it feeds. As Grand Lake filled, a significant mechanism of benthic storage and removal of sediments in the Atchafalaya River Basin disappeared. Temporal patterns of sediment delivery to the Gulf have changed and the fraction of sediment load discharging to the Gulf has increased. These changes can affect not only the growing Atchafalaya Bay deltaic ecosystem, but also human needs for navigation and flood control. The rapid and unusually well documented succession of Grand Lake provides a model of coastal sedimentation patterns which may provide insight into more general mechanisms of sedimentation in shallow inland and coastal waters.

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Introduction

Today, the Atchafalaya River originates near the confluence of Old River and the Red River. Locations along the channel are typically identified as river miles downstream from this confluence which is five river miles north of Simmesport, Louisiana. The present 2190 km² (846 mi²) Atchafalaya Basin Floodway that lies between the East and West Floodway Protection Levees has been termed "America's Greatest River Swamp" (United States Fish and Wildlife Service 1978). The Atchafalaya Basin extends inland 201 km (125 mi) and is roughly 24 km (15 mi) wide. The Atchafalaya River is effectively dammed across its entire lower end by natural levees of long abandoned Mississippi River courses (Russell 1967). Long term patterns of sediment deposition in the Atchafalaya Basin are driven by subsidence and the development of distributary channels (Tye and Coleman 1989a; Tye and Coleman 1989b). Grand Lake, the large lake which formed at the lower end of the Basin, primarily discharges through two outlets, the Lower Atchafalaya River at Morgan City, and the Wax Lake Outlet artificial channel, completed in 1942. In recent decades sedimentation has filled most of the area of Grand Lake (Fig. 1).

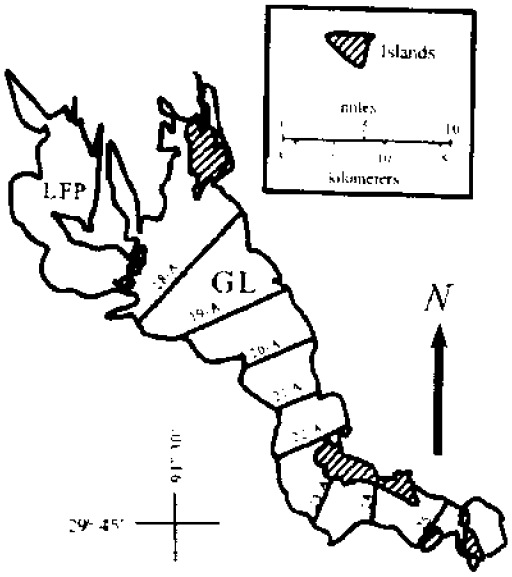

Fig. 1. Grand Lake (GL) and Lake Fausse Point (LFP) shorelines as mapped in 1917. Locations of the 8 Rangelines examined in this study are shown (18-A, 19-A, 20-A, 21-A, 22-A, 23-A, 24, and 25).

Unlike the Mississippi, the Atchafalaya River empties into the Gulf of Mexico through passes into a relatively shallow bay. Thus, in contrast to the Mississippi delta, the Atchafalaya River Delta is actively growing (Adams and Baumann 1980; van Heerden et al. 1983). This accreting delta will provide benefits to Louisiana's fisheries and protection of existing lands and property from erosion and storms. Although this growing delta causes some manageable problems for navigation and potentially impacts local flooding, its benefits vastly outweigh these costs which ultimately may be unavoidable.

The Flood Control Act of 1928 authorized the "Mississippi River and Tributaries Flood Control Plan" and developed the plan to utilize the Atchafalaya Basin as a floodway to reduce flood stages in the Lower Mississippi River. Today, the Atchafalaya Basin serves multiple uses. The principal role, and arguably the most valuable role, of the Atchafalaya Basin Floodway system is to convey one half of the 85,000 m³ s⁻¹ (3,000,000 cubic feet per second, cfs) design flood of the "Mississippi River and Tributaries Flood Control Plan." During such conditions the Atchafalaya Basin conveys half of the total Mississippi River Basin flood discharge to the Gulf. To insure use as a floodway, the federal government has purchased flowage easements on property within the Atchafalaya Basin and regulates development which might interfere with floodway operation. Other important uses of the Atchafalaya Basin include fishery production, sport and non-sport wildlife production, tourism, recreation, agriculture, and silviculture.

At the beginning of the nineteenth century the relationship of the Red River and the Atchafalaya River to the Mississippi River was different from the present configuration (Fisk 1952; Hebert 1967; Reuss 1991; Hale 1996; Reuss 1998). At that time, the Red River was a tributary to the Mississippi River in a large meander loop, and the Atchafalaya River was a very minor Mississippi River distributary originating just downstream of the confluence of the Red and Mississippi Rivers. The head of the Atchafalaya River at this time was blocked by a massive log jam; so massive that cattle were driven across it and live willow trees were

observed growing from it (Hebert 1967; Reuss 1998). The relationship of these rivers was changed dramatically in 1831 when Captain Henry Shreve was authorized to create an 800 ft cut-off through the neck of the Mississippi River's meander loop (later known as Shreve's Cut), which isolated the mouth of the Red River and head of the Atchafalaya River from the main flow of the Mississippi. The upper portion of the severed "Shreve" meander loop soon filled with sediment. The lower portion of the meander loop formed what is now termed Old River, whereupon, the Atchafalaya River began to capture flow from the Mississippi River main channel. The Red River evolved from a tributary of the Mississippi to a headwater of the Atchafalaya River. Demolition of the raft of trees on the Atchafalaya River allowed improved navigation and further changed the regional hydrologic relationships. Between 1900 and 1950, the annual discharge of the Atchafalaya River grew from 12% to 30% of the combined Mississippi River and Red River discharges.

The U.S. Congress, acting to avoid the economic and environmental disaster associated with complete capture of the flow by the Atchafalaya, authorized planning and construction by the U.S. Army Corps of Engineers to ensure that the discharge in these channels remain at the 1950 ratio (Hebert 1967; McPhee 1989; Reuss 1998). It was projected at that time that a 70/30 flow ratio could be maintained in a stable channel configuration, but that if left uncontrolled, the main river flow would naturally reroute to the Atchafalaya by about 1975. The Old River Control Structure (ORCS), completed by the Corps in 1963, controls diversion from the Mississippi River into the Atchafalaya Basin through two dam-like structures. Following flood damage to the ORCS in 1973, the U.S. Congress authorized construction of an auxiliary structure to assist in flow control. Recently, the Sidney A. Murray Jr. Hydroelectric Station generating facility has been constructed and incorporated into the overall control facilities.

Suspended sediment concentrations and loads in the Mississippi River have decreased for several decades (Keown et al. 1986; Dardeau and Causey 1990; Mossa 1990; Mossa 1993). Red River suspended sediment concentration has declined recently, likely caused by the completion of Red River lock and dam projects. This recent decline in Red River suspended sediment concentration is particularly alarming in terms of coastal accretion because historically the Red River contributed loading to the Atchafalaya in a proportion larger than its contribution to volumetric water discharge (Mossa 1990; Mossa 1993). Operation since 1990 of the newly constructed Murray Hydroelectric Power Plant (design capacity 4500 m^3 yr^{-1} or 160,000 cfs), and the Auxiliary Control Structure, completed in 1987, also certainly have affected bedload (and to some degree suspended sediment) distributions from the Mississippi River to the Atchafalaya River.

During the early 1930's the route of the Atchafalaya River through the Atchafalaya Basin was altered. Originally, the major flow of the Atchafalaya River turned sharply eastward near Butte La Rose and followed a course which was in part outside the current East Atchafalaya Basin Protection Levee. Before the 1930's the Atchafalaya River channel did not pass through Grand Lake. Flow through Grand Lake was limited to discharge through a web of small distributaries and overland flows during flood events. In the early 1930's the main Atchafalaya River channel was rerouted to flow through Grand Lake. Although this new channel may have improved channel capacity and created an efficient hydraulic connection to the newly constructed Wax Lake Outlet, it also promoted succession of Grand Lake through the increased deposition of sediments.

In the nineteenth century Grand Lake, also called Lake Chitimaches, was a dominant feature of the Atchafalaya Basin covering approximately 15% of the present day Atchafalaya Basin. Today, only small shallow remnants of the lake exist within the protection levees of the Atchafalaya Floodway. This rapid reduction of lake volume is similar to that documented in some man-made reservoirs which have filled in less than 25 yr (Chanson and James 1998). Protected from sediment load of the Atchafalaya, an arm of the former lake has been preserved as Lake Fausse Point (Tye and Coleman 1989a), located to the west of the West Atchafalaya

Basin Protection Levee. The rapid disappearance of Grand Lake has heightened concern about succession and loss of other features within the Atchafalaya Basin. While sedimentation and delta growth are most often considered desirable processes in the Atchafalaya Bay, sedimentation and associated succession are most commonly considered undesirable within the Atchafalaya Basin. This paper focuses on changes in Grand Lake, documented in records of the eight rangelines (upstream to downstream 18-A, 19-A, 20-A, 21-A, 22-A, 23A, 24, and 25) which cross this lake.

Methods and Materials

In order to identify changes in the capacity of the Atchafalaya Basin to convey flood discharges, the U.S. Army Corps of Engineers has surveyed elevations along transects, termed rangelines, at irregular intervals beginning in 1916. Data from these surveys were obtained from the New Orleans District Office of the Corps of Engineers as large format graphs of elevation plotted against distance along the rangeline. Rangeline data for the entire Atchafalaya Basin for all years were digitized and registered into a three-dimensional georeferenced Intergraph Microstation GIS database. In this 3-d database, the rangelines can be analyzed from many different aspects.

Average elevation was measured using the Microstation polygon area function. A horizontal base line of arbitrary elevation was constructed directly below each rangeline. Vertical lines were then erected from this baseline connecting to the rangeline under consideration at its endpoints or the inner levee toe, and at the main river channel banks. The area of the resulting polygon (or polygons if a section of the main channel crossed the rangeline) was then divided by the total length of the baseline under the polygon to determine the average elevation of the rangelines above the baseline elevation excluding the levees and main Atchafalaya River Channel. These elevations were imported into a spreadsheet for further analysis.

Results

Data from 79 rangeline surveys of Grand Lake in 23 different years over the period from 1916 to 1988 were available for this study (Table 1). Because of budget and logistical considerations, all eight rangelines were surveyed in only one (1974) of these 23 years. River miles listed with rangeline numbers in Tables 1 and 2 provide an approximation of location within the Atchafalaya Basin. As a reference, the Corps of Engineers Butte La Rose gage (03120) is located at river mile 64.8, the Six Mile Lake gage (03645) is located at river mile 105.2, and the gage at Morgan City (03780) is located at river mile 117.7.

Over the entire period of observation the bottom surface elevation sloped toward the Gulf. In the earliest observations in 1917, the minimum elevation was -3.19 m at rangeline 25 near river mile 110. By 1980 this downstream elevation had risen to -0.34 m, an average rate of accretion of 4.5 cm yr^{-1}. From 1917 to 1941 accretion rate at this rangeline was 4.2 cm yr^{-1}, and from 1941 to 1980 accretion rate was 4.7 cm yr^{-1}, an increase of 12%. At the upstream rangeline, 18-A, at river mile 85, elevation rose from -1.07 in 1917, to 0.09 in 1940, to 0.93 m in 1978. This corresponds to an accretion rate of 3.3 cm yr^{-1} over 1917 to 1978, a rate of 5.0 from 1917 to 1940 and a rate of 2.2 cm yr-1 from 1940 to 1978. Thus, at the upstream rangeline accretion rate dropped 56 percent in the later period.

Overall average accretion rate for all rangelines over the entire period of study was 3.6 cm yr^{-1}. This is an unusually high rate of accretion. Referring to canals and wetlands, Dortch (1996) states that "accretion rates are typically on the order of 0.1 to 1.0 cm yr^{-1}. Extensive reservoir sedimentation survey statistics compiled by Dendy (1969) show that typical accretion rates are more than an order of magnitude below the values determined here. The high rate of accretion observed in Grand Lake may be attributed to the high suspended sediment concentration of the Atchafalaya River, the very large flow passing through the lake, and the initial lake depth.

To provide comparative water surface elevations, stage data from three Atchafalaya River Stage gages were analyzed (Fig. 2). Lines of water surface elevation, termed flowlines, at three different discharge values were derived from published stage and discharge data for the 1974 calendar year.

Table 1. Average Grand Lake rangeline elevation in meters above mean sea level (AMSL). Main channel and main channel natural or artificial levees are excluded.

Year	18-A	19-A	20-A	21-A	22-A	23-A	24	25
1916				-1.65	-1.86			
1917	-1.07	-1.35	-1.44			-2.14	-2.42	-3.19
1934	-0.52	-0.91	-1.41	-1.93	-1.68			
1940	0.09	-0.28	-0.77					
1941				-1.34	-1.31	-1.62	-2.09	-2.18
1950		0.47	0.13	-0.64	-0.90			
1951	0.72	0.39		-0.50	-0.91	-1.71	-1.58	-2.62
1953			0.35					
1956							-1.14	
1959				-0.18		-1.19	-0.74	
1963	0.71	0.76			-0.13			-1.24
1964				0.07		-0.70	-0.48	
1967				-0.16		-1.02	-0.51	-0.78
1968	0.78	0.77	0.33		0.00			
1972				0.04		-0.54		
1974	0.98	0.76	0.58	0.34	0.51	-0.27	-0.22	-0.05
1975				0.39	0.56			
1976			0.47			0.01	0.16	-0.14
1978	0.93							
1980				0.07	0.22		0.03	-0.34
1987				1.18	0.40			
1988			-0.17			0.10	0.00	

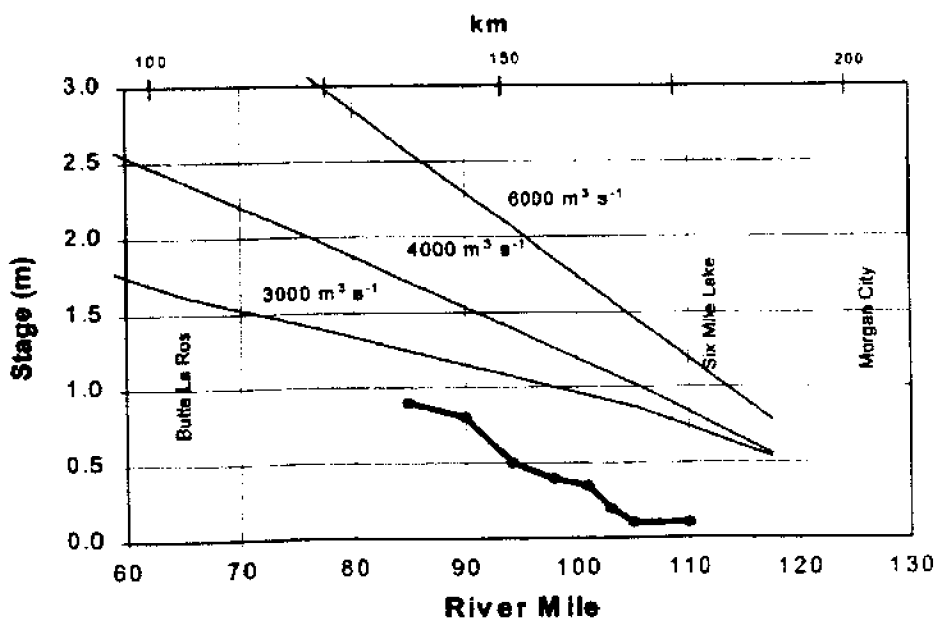

Fig. 2. Model equilibrium elevations (heavy line), and flowlines of water surface elevation at 3000, 4000, and 6000 m^3s^{-1} discharge derived from 1974 USACE stage data.

Elevations at Butte La Rose, Six Mile Lake, and Morgan City were determined for three Atchafalaya River discharges measured at Simmesport. Flowlines for 3000, 4000, and 6000 $m^3 s^{-1}$ (106, 141, 212 thousand cfs) were determined. These discharges represent approximately the 25, 40, and 60 % flow duration (i.e. percent of days that the flow is not exceeded) since the Old River Control Structure began operation (Wells and Demas, 1977).

Model Development

Observations listed in Table 1 are difficult to analyze because of the large number of missing values for any specific year. Additionally, variability in individual observations obscures more general trends. In order to overcome these difficulties, a simple empirical model of annual accretion was developed and compared to the values in Table 1. Sedimentation and accretion models are most commonly based on an assumed particle settling velocity determined through calibration or using Stoke's Law (Cerco and Cole 1995; Dortch 1996). Application of these models requires at a minimum, a knowledge of water velocities and depths; these were unavailable for Grand Lake. In consideration of this limitation, the model developed here is empirical and is based on observed trends and asymptotic reasoning. Annual change in average elevation was modeled as:

$$r_t = E_t - E_{t-1} = \frac{E_0 - E_{t-1}}{K + E_0 - E_{t-1}} = r_{max} \qquad (1)$$

$$r_{max} = r_0 Q_t^a \qquad (2)$$

where r_t is the annual accretion rate over year t ($m\ yr^{-1}$),
E_t is average rangeline elevation AMSL (m) at the end of year t,
E_0 is the equilibrium (maximum) accretion elevation (m),
r_{max} is the maximum accretion rate ($m\ yr^{-1}$),
Q_t is river annual calendar-year average volumetric discharge ($m^3 s^{-1}$),
K is the distance (m) below E_0 at which accretion rate is reduced to half r_{max}, and
r_0 and a are constants.

For Grand Lake these parameters were estimated through a visual calibration to the observed values. An optimal value of the exponent "a" was found to be near unity, and a value of 1 was accepted as the calibrated value. The parameter r_0 was determined to be 1.803 $s\ m^{-2}\ yr^{-1}$ through calibration. For these parameter selections, the average value of r_{max} over the period from 1934 to 1990 is 9.95 $cm\ yr^{-1}$. The parameter K was found, through calibration, to be 0.4 m at all rangelines. The maximum accretion elevation, E_0, was found to depend on the rangeline location, and generally decreased from upstream to downstream locations. For the rangelines 18-A, 19-A, 20-A, 21-A, 22-A, 23-A, 24, and 25, the value of E_0 was estimated to be 0.9, 0.8, 0.5, 0.4, 0.35, 0.2, 0.1, and 0.1 m, respectively (Table 2; Fig. 2).

Because the spatial pattern of flows through the Atchafalaya Basin was radically changed in the 1930's, the calibrated model was initiated in 1934 (Table 2). Initial values were input for 1934, and subsequent years were then calculated on an Excel spreadsheet using the parameters cited above (Table 2). Calculated values were in close agreement (Fig. 3) with the observations listed in Table 1.

One goal of this study is to provide a more generalized method for projecting accretion which may be particularly useful in planning river diversions for habitat restoration. It is therefore further conjectured that r_{max} can be related to annual average suspended sediment concentration, c_a and water application rate (discharge per unit area inundated), q_t, by:

$$r_{max} = r^* q_t \frac{c_a}{\lambda} \qquad (3)$$

where
r^* is a dimensionless constant which may be expressed as a percentage,
γ is surficial sediment bulk density, the dry weight of bottom sediments per unit accreted volume ($kg\ l^{-1}$),

and

$$q_t = \frac{Q}{A} \qquad (4)$$

Table 2. Model calculated average Grand Lake rangeline elevation in m above mean sea level (AMSL). Model parameters are displayed near the top of the table.

	Rangeline/River Mile							
Rangeline	18-A	19-A	20-A	21-A	22-A	23-A	24	25
River Mile	85	90	94	98	101	103	105	110
E_0 (m)	0.9	0.8	0.5	0.4	0.35	0.2	0.1	0.1
K (m)	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4
r_0 (cm cms^{-1}yr^{-1})	1.803	1.803	1.803	1.803	1.803	1.803	1.803	1.803
a	1.000	1.000	1.000	1.000	1.000	1.000	1.000	1.000
YEAR								
1916	#N/A	#N/A	#N/A	#N/A	#N/A	#N/A	#N/A	#N/A
1917	#N/A	#N/A	#N/A	#N/A	#N/A	#N/A	#N/A	#N/A
1934	-0.30	-0.70	-1.15	-1.80	-1.90	-3.00	-2.60	-3.10
1940	0.00	-0.38	-0.82	-1.45	-1.55	-2.63	-2.24	-2.73
1941	0.04	-0.33	-0.78	-1.40	-1.50	-2.58	-2.19	-2.68
1950	0.59	0.29	-0.13	-0.68	-0.77	-1.79	-1.43	-1.89
1951	0.64	0.36	-0.05	-0.58	-0.68	-1.68	-1.32	-1.78
1953	0.71	0.45	0.05	-0.46	-0.55	-1.54	-1.19	-1.64
1956	0.77	0.53	0.15	-0.34	-0.42	-1.39	-1.05	-1.49
1959	0.83	0.65	0.29	-0.13	-0.22	-1.13	-0.81	-1.23
1963	0.87	0.73	0.40	0.07	-0.01	-0.84	-0.56	-0.94
1964	0.88	0.74	0.41	0.09	0.01	-0.81	-0.52	-0.91
1967	0.89	0.77	0.45	0.19	0.12	-0.63	-0.38	-0.73
1968	0.89	0.78	0.46	0.23	0.16	-0.56	-0.32	-0.66
1972	0.90	0.79	0.49	0.34	0.28	-0.28	-0.11	-0.38
1974	0.90	0.80	0.50	0.38	0.32	-0.10	0.00	-0.20
1975	0.90	0.80	0.50	0.38	0.33	-0.03	0.03	-0.13
1976	0.90	0.80	0.50	0.39	0.34	0.00	0.04	-0.10
1977	0.90	0.80	0.50	0.39	0.34	0.03	0.05	-0.07
1978	0.90	0.80	0.50	0.39	0.34	0.06	0.06	-0.04
1980	0.90	0.80	0.50	0.40	0.35	0.12	0.08	0.02
1987	0.90	0.80	0.50	0.40	0.35	0.19	0.10	0.09
1988	0.90	0.80	0.50	0.40	0.35	0.19	0.10	0.09

The value of q_i is commensurable with precipitation rate and other area-specific intensive parameters, and may be most conveniently reported as m yr^{-1}.

The parameter r^* can be interpreted as the fraction of the suspended sediment which is removed from the water column in the process of accretion and is related to trapping efficiency defined in other sedimentation studies (Strand and Pemberton 1982; Salas and Shin 1999). Solving Equations 2 and 3 gives:

$$r^* = r_0 A\lambda/c_a \qquad (5)$$

Using parameter values available from the literature, the value of r^* can be roughly estimated. Over the period 1964 through 1974, Atchafalaya River suspended sediment concentration averaged 460 mg l^{-1} (Wells and Demas 1977). Although suspended sediment concentrations in the Mississippi River have declined since the middle of this century (Dardeau and Causey 1990), a constant value is used here for analysis in order to simplify computation, and because very roughly half of the suspended sediment of the Atchafalaya River is supplied by the Red River (Mossa 1990) which is not known to have declined in suspended sediment concentration prior to the relatively recent

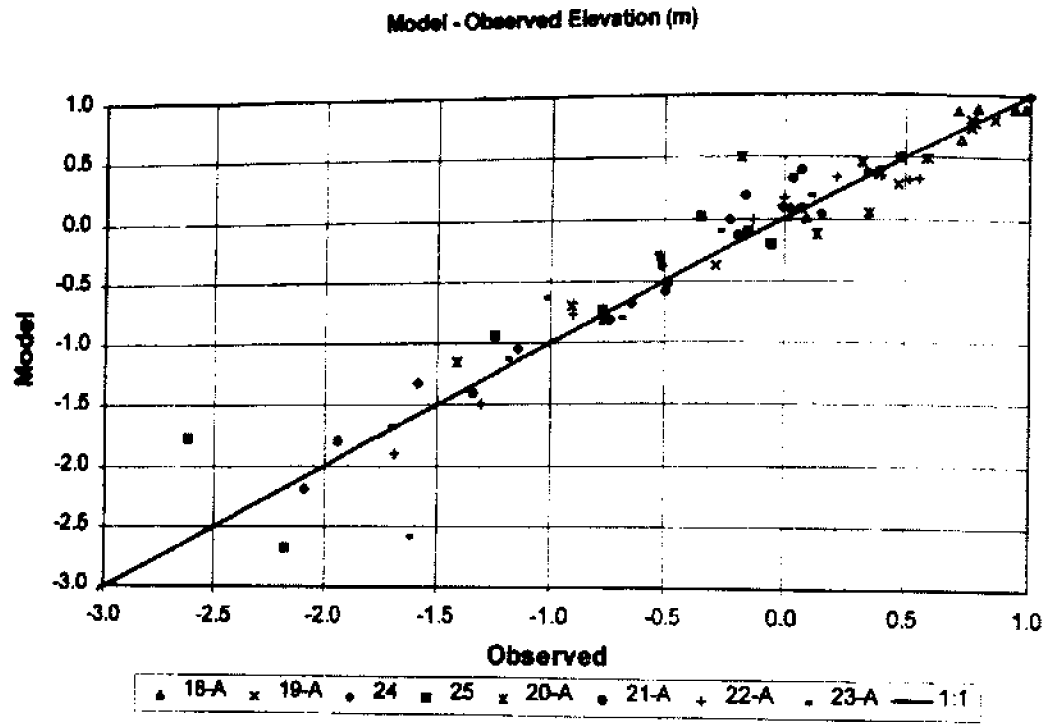

Fig. 3. Comparison of model and observed average rangeline elevations. Solid line is line of perfect fit (1:1 slope).

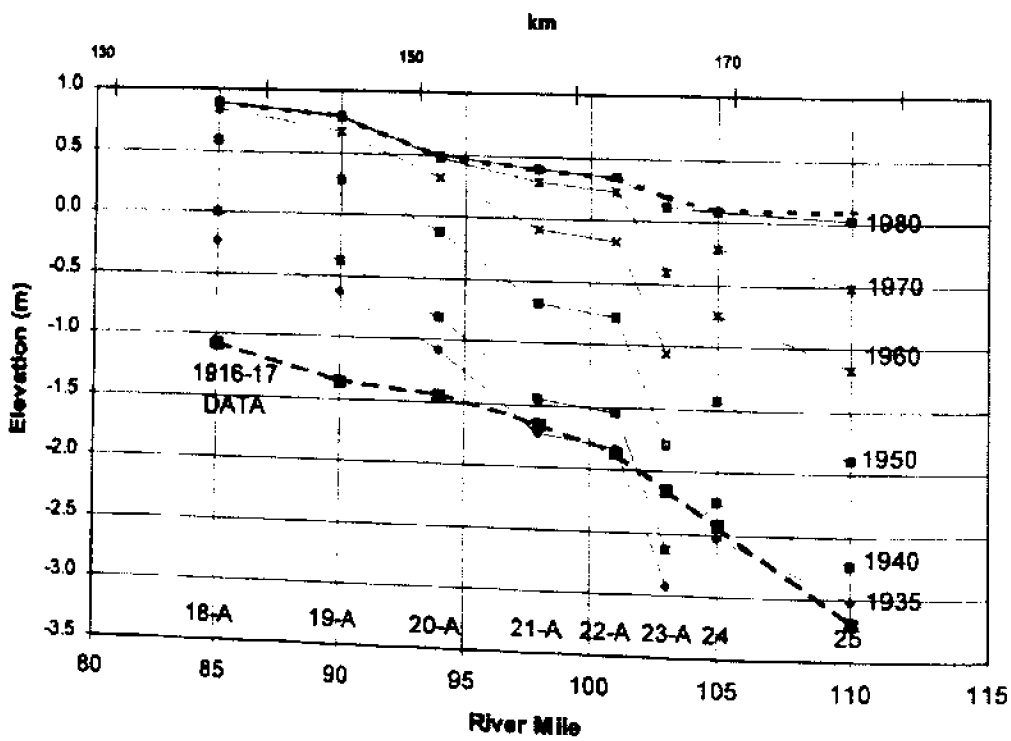

Fig. 4. Average elevation along Grand Lake rangelines. Points from 1916-1917 are observed values, solid lines are model projections, upper dashed line is equilibrium elevation.

navigation improvements. Bottom surficial sediment densities have been compiled for a large number of North American lakes by Dendy and Champion (1969). A typical value of 1.12 kg l^{-1} (70 lb ft^{-3}) is used here. Combining these values with the original area of Grand Lake provides an estimate of 49% for r^*.

This estimate may be further compared to the peak percent of load removal projected in the model. Wells and Demas (1977) report that the average suspended sediment load passing Simmesport is 86 x 10^6 t yr^{-1} (260,000 t d^{-1}). The peak modeled annual accretion rate (Fig. 5) is 38 x 10^6 m^3, which, using 1.12 t m^{-3}, converts to 43 x 10^6 t yr^{-1}, or 50% of the average suspended sediment load. Average annual accretion projected by the model from 1965 through 1971 is 11.8 x 10^6 m^3 yr^{-1}. This would correspond to an annual removal of 13.2 x 10^6 t. Wells and Demas (1977) report that although data are limited, it is estimated that over this period approximately 75% of the suspended sediment load at Simmesport was transported through the two outlets. They also report that net scour of the Basin was observed during the floods of 1973, 1974, and 1975.

Discussion and Conclusions

Accretion in Grand Lake peaked around 1950, and has generally diminished since that time (Fig. 5). Sedimentation and succession of Grand Lake will continue in isolated areas, but at a greatly reduced rate. In the future, sedimentation will likely focus on the remaining low-lying areas of the former lake as upper rangelines equilibrate at a mean elevation of <1 m. During the disappearance of deep open water, the river channel extended itself through its growing floodplain. Development of a more efficient channel must, to some degree, reduce the sediment load flowing over the elevated floodplain, and affect final equilibrium elevations. Succession is approaching a final equilibrium condition with a typical channel, natural levee, and flood plain morphology. Today, the rate of sediment trapping in the former Grand Lake has greatly slowed or stopped, and Grand Lake is approaching an equilibrium surface elevation.

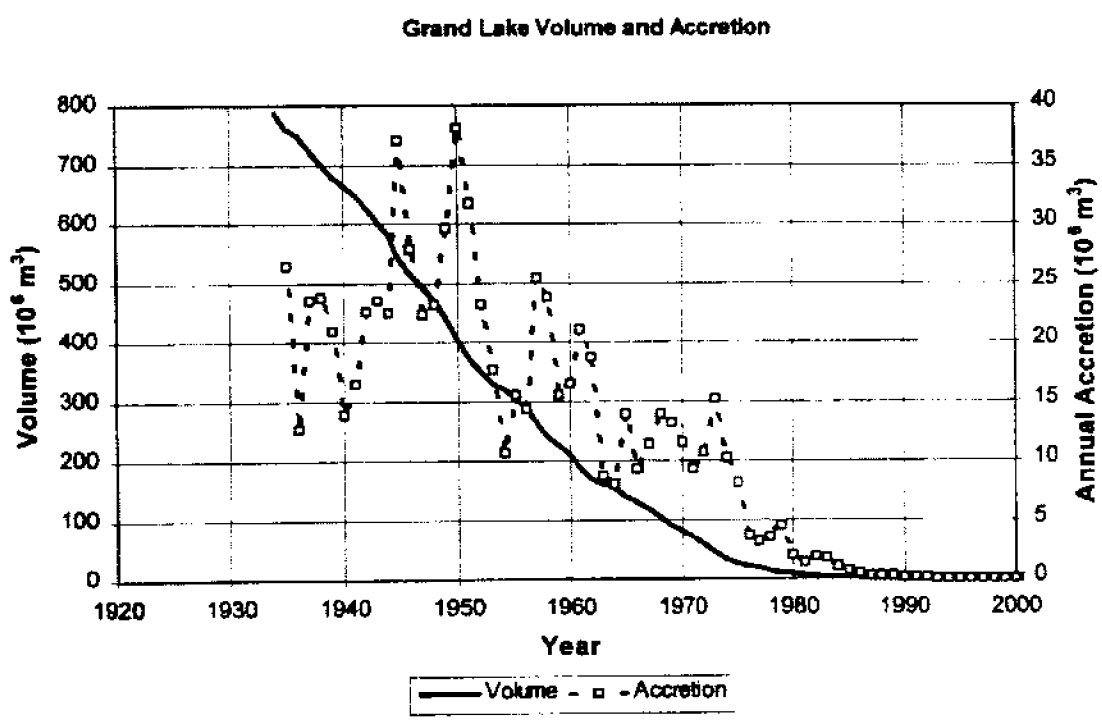

Fig. 5. Model projected Grand Lake volume (below equilibrium elevation) and annual accretion rate.

During the decades of the 1930's and 1940's a significant fraction, perhaps 20-40%, of the suspended sediment load entering the Atchafalaya Basin remained in the Basin and was unavailable for delta accretion in Atchafalaya Bay. Fisk (1952) projected that the Atchafalaya Basin would be filled with sediment by the early 1970's, and a new marine delta would then build in Atchafalaya Bay (Tye and Coleman 1989b). Since the mid 1970's, Grand Lake has been effectively filled. Today, the sediment trapping efficiency of the lake has declined to near zero, and nearly all suspended sediment entering Grand Lake is delivered to Atchafalaya Bay. Similar successional patterns have been observed in man-made reservoirs (Lajczak 1996; Kern and Westrich 1997).

Patterns of sedimentation have been identified in this study. Accretion slows and finally stops as elevation approaches an equilibrium elevation. Areas of deep water accrete sediments more rapidly than shallow water bodies. Lakes also may preferentially fill with sediments from upstream to downstream segments. Expression of these patterns to the point that they can be identified and quantified depends on local hydraulic conditions and suspended sediment availability.

Accretion occurs preferentially in deep areas. This observation is consistent with observations and models of reservoir sedimentation (Strand and Pemberton 1982). In Grand Lake, we found that accretion slows to 33% of the maximum rate at a bottom elevation 0.2 m below the equilibrium elevation. For a depth of 2 m below the equilibrium elevation, sedimentation occurred at >80% the maximum rate determined by suspended sediment load. Thus, the rate of sediment accretion is relatively uniform and higher in deeper waters.

The model developed here does not consider the longitudinal reduction of suspended sediment concentration which results from upstream accretion. Here this is appropriate because detention times are sufficiently short. In other situations it may be appropriate to consider upstream trapping and loss of suspended sediment by subdividing the study area into interconnected segments.

A better understanding is needed of the factors that determine the equilibrium surface elevation. It is likely that this elevation is related to some statistic (e.g. mean annual low water) of local water-surface elevation. Mechanisms which reduce observed accretion as the equilibrium elevation is approached include loss of sediment source, increased scour and erosion, and enhanced consolidation of higher elevation surfaces.

Preferential accretion in deep water may be one mechanism contributing to indirect land loss from excavation of canals in wetlands. In situations where suspended sediment supply is limited, sediments may preferentially settle in deep canals and therefore are unavailable to maintain surface elevations of surrounding wetlands.

The methodology reported here could provide a quantitative basis for projecting and analyzing river diversion projects which divert flow to open water. It may also provide a basis for design of accretion monitoring studies in these areas. These simple modeling techniques also may provide insight in interpretation of other wetland sedimentation studies using different data collection methodologies (Hupp and Morris 1990; Hupp et al. 1993).

Much information about sedimentation and accretion can be extracted from the Atchafalaya Basin rangeline data. Only the data concerning Grand Lake have been considered in this paper. We plan to continue examining these valuable records, and to extend the analysis to the entire Atchafalaya River Basin.

ACKNOWLEDGMENTS

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Sedimentation, Consolidation and Surface Elevation Change in Two Salt Marshes of the Mississippi River Deltaic Plain: Geotechnical Aspects of Wetland Loss

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ABSTRACT: A geotechnical process acting to reduce the strength of the marsh soil substratum is proposed to explain observed patterns of salt marsh loss in the Mississippi River deltaic plain. Factors affecting marsh surface elevation, including sedimentation, accretion and consolidation, were investigated at two sites about 60 km apart, one adjacent to, and one distant from direct fluvial influence. The sites are end members on a sustainability scale. The isolated marsh at Bayou Chitigue (BC) is rapidly submerging and converting to open water. The Old Oyster Bayou (OB) marsh has remained stable as it has been increasingly affected by the Atchafalaya River, a Mississippi River distributary now carrying about 30 percent of total flow. Elevation at BC, 10 cm lower on average than at OB, did not increase at a sufficient rate to offset any relative sea level rise (RSLR) greater than zero, much less the 1 cm-y^{-1} estimated for this site, even though it is receiving sediment at more than twice the rate at OB. Shallow subsidence (0-4 m) was estimated at 3.6 and 1.6 cm y^{-1} at BC and OB, respectively, or more than three times RSLR estimated for these sites. A detailed analysis of marsh surface soil (< 25 cm) properties disclosed only subtle differences between the two sites, suggesting that the marsh cap itself remains relatively unchanged as it declines toward the lower bound of marsh plant viability. Shallow subsidence is taking place in the marsh substratum beneath the marsh cap. Sediment availability is enhanced at BC as a consequence of the higher frequency and duration of flooding, but does not result in a proportional increase in accretion or marsh aggradation. The BC marsh captures less than half of the available sediment as this material apparently remains in flux under the prolonged flooding characteristic of this site. Marsh vegetation persists at close to the lower elevation limit and breaks up as individual plants die. The OB marsh, in contrast, captures virtually all of the smaller input of sediment available and is aggrading at a rate that appears to approximate local RSLR (0.5 cm y^{-1}). A theoretical geotechnical analysis of the load imposed by the marsh cap under normal drainage conditions

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is used to show that shallow subsidence should, like sedimentation, be a function of position within the tidal frame. A decrease in loading of the marsh substratum drives a reduction in strength in this section, leading to an increased potential for sudden settlement that can precipitate marsh loss. Soil strength profiling is suggested as a basis for predicting the restorability and response of tidal marshes to river diversions and other restoration projects.

INTRODUCTION

Dunbar et al. (1992) estimated that 400,000 ha of estuarine wetlands associated with the Mississippi River delta in Louisiana have submerged and converted to open water since the 1930s. Much evidence points to waterlogging as a primary mechanism for this marsh die-off (Naidoo et al. 1992, Ernst 1990, Pezeshki et al. 1991, Pezeshki et al. 1988, Mendelssohn and McKee 1988, Morris and Dacey 1984). Rates of relative sea level rise (RSLR) for this area range from 0.5 to more than 2.0 cm y^{-1} (Penland and Ramsey 1990). For coastal marshes to survive, the rate of aggradation, or absolute surface elevation gain, must equal or exceed RSLR (Baumann et al. 1984, Reed and Cahoon 1992, Reed 1995a).

In contrast to the general pattern of loss, salt marshes continue to flourish adjacent to the unleveed mouth of the Atchafalaya River, a distributary of the Mississippi that carries approximately 30 percent of the total flow (Fig. 1). The exact mechanisms through which the salt marsh is affected by river discharge, however, remain poorly understood (Turner 1997). Such detailed information is required to optimize design and operation of the river diversion structures and channels that are proposed as the primary means for restoring the wetlands of the Mississippi River deltaic plain (Reed 1995b, Boesch et al. 1994).

Aggradation has been shown to track sediment accretion for salt marshes with mineral soils (Pethick 1981, Stumpf 1983, Stoddart et al 1989, French 1993, Allen 1994). Stevenson et al. (1986) examined accretion in marshes from a variety of tidal ranges and concluded that mineral sediment input was critical even in relatively peaty salt marsh settings. Others have found, however, that accumulation of organic matter rather than mineral sediment is better correlated with vertical accretion (Hatton et al. 1983, Bricker-Urso et al. 1989, Craft et al. 1993, Nyman et al. 1993, Callaway 1994).

A collapsible soil matrix results when much of the soil volume is contributed by plant material and void space, both highly compressible elements (McCaffrey and Thomson 1980, Hackney and Cleary 1987, Krone 1987, Oenema and Delaune 1988, Cahoon et al. 1995a). Salt marsh soils of the Mississippi River deltaic plain are not as organic as some at higher latitudes on the U. S. east coast (Bertness 1988, Turner 1976). But void space and organic matter together take up 95 percent of the volume of typical Louisiana salt marsh soils (Nyman et al. 1990, Delaune et al. 1983).

Cahoon et al. (1995a) compared precise, direct measurements of marsh surface elevation change using a sediment erosion table (SET) (Boumans and Day 1993) with standard measures of vertical accretion. They found that accretion was not a reliable predictor of aggradation in some salt marshes, particularly those experiencing high rates of RSLR. Despite high rates of accretion in these marshes, the elevation of the marsh surface either did not change or decreased. Cahoon et al. (1995a) attributed this to the offsetting effect of autocompaction or shallow subsidence occurring between the marsh surface and the base of the SET support pipe, generally 3 to 5 m deep. They distinguished this settlement from deeper subsidence caused by consolidation and tectonic movement below the base of the SET.

Terzaghi (1936) proposed that most changes in soil consolidation, bearing strength and shearing resistance can be attributed to changes in the loading force or effective stress imposed on that soil.

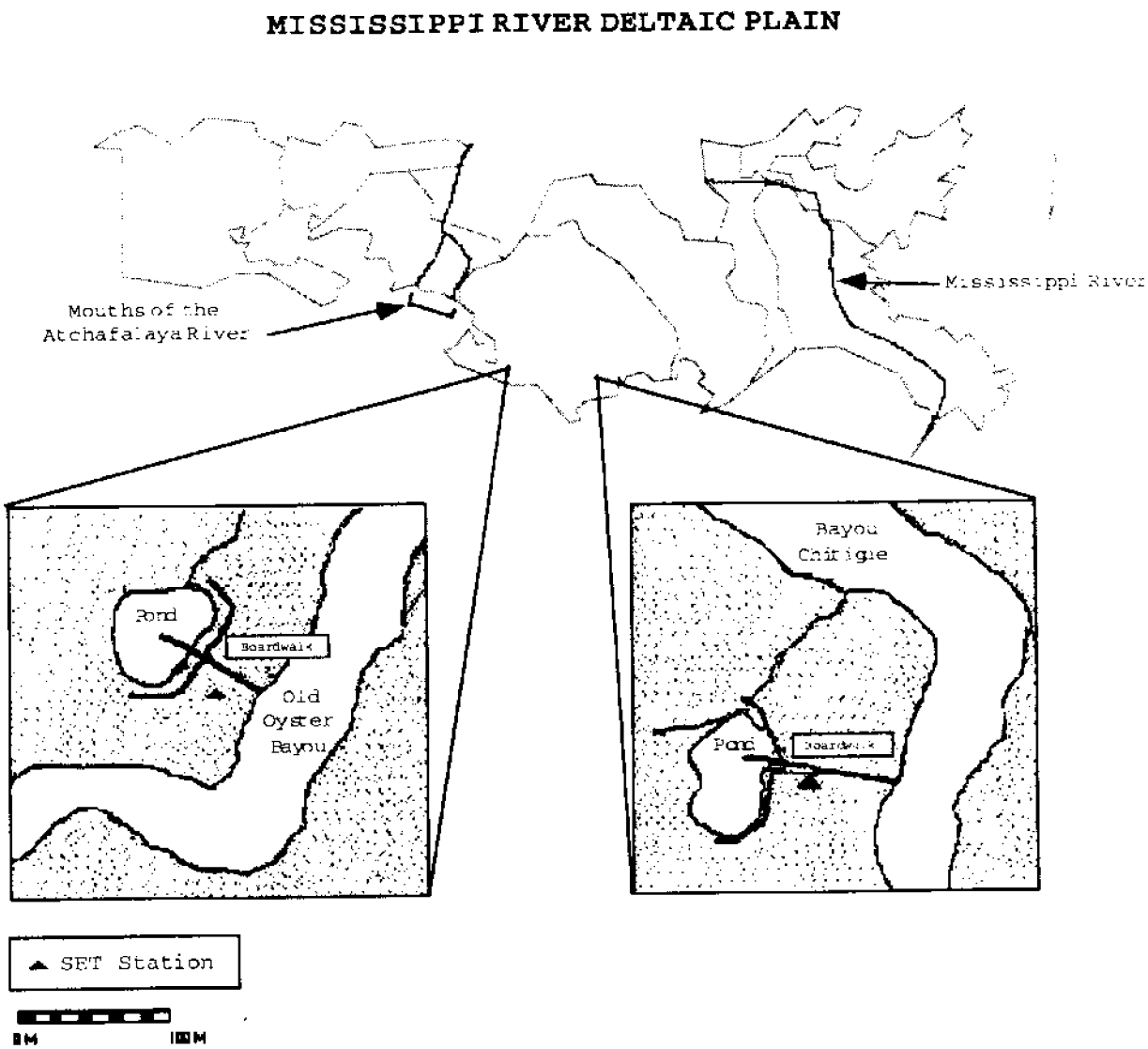

Figure 1. Locations of Bayou Chitigue (BC) and Old Oyster Bayou (OB) salt marsh sites.

Consolidation, from this perspective, is defined by the rate at which the soil loses void space in response to an increase in effective stress. Soils will be affected by all additions to the load that must be supported. A lower soil void volume is generally associated with greater soil strength and the ability to support added loads without settlement (Dunn et al. 1980). Such loads are applied in a natural wetland setting through the addition of sediment at the surface, or as a consequence of drainage. The expected result is a "normal" consolidation profile that exhibits a reduction in void volume and increase in strength with depth (Kosters 1987, Kuecher 1994, Kaye and Barghoorn 1964). The loading resulting from sedimentation and drainage are affected both by proximity to a sediment source (Leonard 1997, Allen 1994, Reed 1988, Stumpf 1983) and by position within the tidal frame (French 1993, Reed and Cahoon 1992, Krone 1987, Pethick 1981, Pestrong 1969).

We extend the analysis of Cahoon et al. (1995a) to quantify processes that affect surface elevation at two Mississippi River deltaic plain sites

proximal to and distant from riverine influences. The first was rapidly submerging and converting to open water, while the latter has remained stable for at least 30 years (Day et al. 1994). We hypothesized that divergence of the elevation trajectories of the two sites, and their marsh loss histories, could be related to differences in (1) sediment supply, (2) in the ability of near surface marsh soils to support added sediment, or (3) to subsidence originating at greater depths.

SITE DESCRIPTION

The study sites were selected about 60 km apart (Fig. 1). Both marshes are of similar age and origin, derived from the Lafourche deltaic complex that was active between 3,000 and 500 BP, though the thickness and sediment characteristics of the underlying Holocene fill sequences are quite different (Frazier 1967, Penland et al. 1987).

The Old Oyster Bayou (OB) marsh is adjacent to a bay that receives inputs of sediments from the Atchafalaya River. The bay system that provides reworked sediments to the Bayou Chitigue (BC) site, in contrast, is currently isolated from direct river inflows (Huh and Rouse 1994). Representative salt marshes dominated by *Spartina alterniflora* were chosen for both sites, each located between a small (< 1.0 ha) pond and a tidal channel (bayou) with similar hydrologic connections to nearby bays (Day et al. 1994).

Boardwalks were constructed at each site along a transect from the bayou to the pond to allow access with a minimum of marsh disturbance. The two sites differed in that the bayou to pond transect at BC was 110 m long, while that at OB was only 50 m (Fig. 1). OB is a firm marsh with a continuous vegetation cover and little exposed mud or channelization of the surface. Wetland loss in this area has been minor (Reed 1995b). At BC, the marsh is deteriorating rapidly and is adjacent to an area that has experienced extensive conversion to open water over the past 40 years (Dunbar et al. 1992, Reed 1995b). The BC marsh surface is difficult to traverse and is interlaced by unvegetated depressions where mud is visible between marsh grass hummocks. Nyman et al. (1993) and Delaune et al. (1994) described a 10 to 15 cm loss of elevation associated with vegetation death in hummocks near BC. They determined that this elevation change was caused by an in-place collapse of the marsh soil matrix, rather than by mechanical removal or erosion.

Relative surveys of the marsh surface at the two sites showed a pronounced natural levee ridge adjacent to the bayou at BC that was not as apparent at OB (Wang 1994). Marsh elevation at BC exhibited 20 cm of relief, cresting 15 m from the bayou, but then decreasing as abruptly on the inland side, and declining more gradually to the pond margin. Elevation at OB was more uniform, except for a 5 cm drop within 10 m of the pond margin (Wang 1994)

The hydrology of these sites has been described by others from concurrent gaging data obtained in 1993 (Wang et al. 1993, 1994; Wang 1994, Wang 1997). Both sites experience the 20 cm annual excursion in sea level that is typical of this part of the northern Gulf of Mexico (Marmer 1954, Reed 1995b). Sea level is highest during the late spring and early fall and is lowest in the winter. When the marsh was flooded by more than 10 cm at the mid-transect gage, flow at both sites was from the bayou toward the pond (Wang 1994, Wang 1997).

The BC marsh experienced more than twice the number of flooding events as that at OB, as well as a greater depth of flooding (Fig. 2). The BC marsh was also inundated for longer periods, and was never dry for more than 10 consecutive days. The OB marsh, in contrast, was not regularly flooded, and was exposed at times for more than 30 days. Both sites occasionally experienced drainage to depths of 10 to 20 cm below marsh surface (Wang 1994). Elevation measurements using a high precision regional differential GPS survey showed that the OB marsh surface midway between the bayou and pond was 10 to 15 cm higher than the BC marsh surface at the same relative position (Alawady and El Taha 1995), thus explaining the offset in flooding characteristics (Fig. 2).

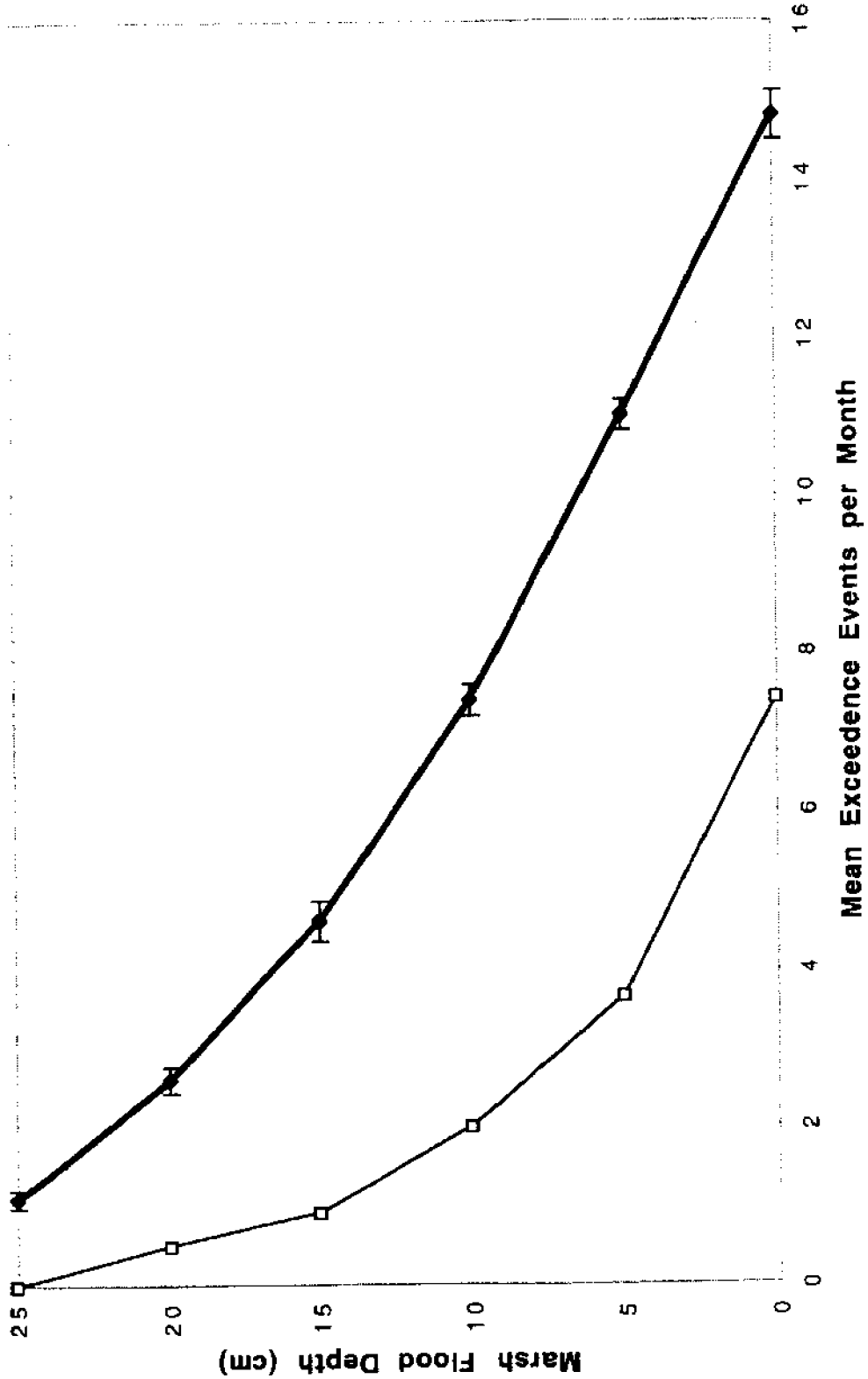

Figure 2. Marsh flooding depth and frequency at Bayou Chitigue (closed) and Old Oyster Bayou (open): January 1993 - September 1993. Mean ± 1 SE.

Data collection was initiated at the BC site in September 1990, and continued at both sites between October 1991 and December 1993. Hurricane Andrew, with sustained winds of about 200 km h^{-1}, made landfall between the two sites on August 25, 1992. An analysis of the hurricane impact at these sites is given in Cahoon et al. (1995b).

MATERIALS AND METHODS

Sediment Deposition, Accretion and Elevation Change

Petri-dish sediment traps consisting of precombusted, preweighed glass fiber filters fixed to the marsh surface were used to measure short-term sediment deposition (DEP) (Reed, 1989, 1992; Leonard 1997). At each site, traps were arrayed at 18 locations between the bayou and pond. They were deployed in groups of three that were collected biweekly over 40 intervals from February 1992 to September 1993. No data was collected at BC over one 4 week interval including Hurricane Andrew (August 19 to September 28, 1992). The filters were washed with deionized water to remove salts and weighed after drying in the oven at 60 °C.

Longer-term rates of sediment accretion (ACC) were determined with feldspar marker horizons using the techniques of Cahoon and Turner (1989). Triplicate horizons were established at nine sites at BC in September 1990 and at OB in August 1991. Each marker was cored in the spring and fall until October 1992 using the cryogenic method (Knaus and Cahoon 1990). About half of the horizons could not be found after the storm when most location indicators were stripped from the sites. Cahoon et al. (1995b) were more successful, however, in locating similar layers at these sites after the storm, and concluded that surface erosion was minor.

Changes in the elevation of the marsh surface (E_{SET}) at the transect midpoints were determined using the SET method (Boumans and Day 1993, Cahoon et al. 1995a). Two SET stations were installed in the marsh at BC in September 1990 and at OB in August 1991, and measurements continued at both sites through December 1993. One SET station was located at the midpoint of each transect, and the second was placed just inland of the pond margin (Fig. 1).

Each SET station provides 36 individual measuring points grouped in quadrants radiating from a common support pipe that serves as a benchmark (Boumans and Day 1993). The SET support pipe was vibrated into the soil to a depth of approximately 4 m to provide a stable benchmark, so that measured changes in soil surface elevation integrate all processes that result in aggradation or degradation within this zone (Cahoon et al. 1995a). The SET collar elevation from the interior station at each site was later tied into the regional differential GPS survey to obtain the absolute differences in marsh surface elevation discussed above (Alawady and El Taha 1995). A paired t-test was used to assess the significance of differences between the sites (SAS Institute Inc. 1991). Shallow subsidence (S) was calculated following Cahoon et al. (1995a) as the difference between ACC and E_{SET}.

Soil Measurements

Cores, 10.2 cm in diameter and 25 cm long, were collected at each site from the marsh between the bayou and the pond along transects that included the marker horizons and SET stations. Two preliminary cores acquired in March 1993 at 60 and 80 m from the bayou at BC, and at 20 and 45 m into the marsh at OB, have been described by Breuil (1994). A more extensive coring program followed later that year in November that produced the data analyzed here. Four cores were collected from the BC site at 10, 20, 60 and 80 m from the bayou, and five cores were extracted at OB at distances of 5, 10, 20, 40 and 45 m (Wang 1994). Thin-walled aluminum tubes were slowly pushed into the soil by hand, while roots were cut along the outside to limit compaction. The top and bottom were capped prior to withdrawal. Cores were frozen and later cut into 5 cm sections that were dried at 80 °C. Core sections were weighed to yield bulk density and then were combusted at 550° C and reweighed to determine percent organic matter (%OM). A paired t-test was used to compare measurements from the two sites (SAS Institute Inc. 1991).

Void Ratio and Moisture Content

Consolidation is determined as a reduction in the dimensionless void ratio, e, defined as:

$$e = \frac{V_v}{V_s} \qquad (1)$$

where V_v is the volume of the voids and V_s is the volume of the solids (Dunn et al. 1980). The void ratio is derived from laboratory analysis of the soil:

$$e = \left[\frac{G_s \gamma_w}{\gamma_d}\right] - 1 \qquad (2)$$

where G_s is the dimensionless specific gravity of the soil solids, g_w is the unit weight of water (1 g cm^{-3} or 9.81 kN m^{-3}), and g_d is the dry unit weight or bulk density of the soil (Dunn et al. 1980, Bowles 1992).

G_s was not determined directly but was estimated from g_d and %OM based upon laboratory measurements made by Breuil (1994) on representative soils from the two sites. These results were consistent with specific gravities of 2.61 and 1.14 for mineral and organic soil constituents, respectively, that were previously determined by Delaune et al. (1983), and have been used by others to characterize Louisiana salt marsh soils (Nyman et al. 1990, Callaway 1994).

Because the marsh soil volume is largely voids, water that enters and is trapped in these voids can result in a moist unit weight that is far greater than the dry unit weight. If the tide level is at or above the marsh surface so that the soil is saturated, then the weight of the water and a portion of that of the soil solids is hydrostatically supported. Only the uncompensated (buoyant) weight of the solids loads the underlying soil (Dunn et al. 1980). When the water level decreases below the marsh surface and some drainage occurs, the soil moisture content drops below saturation. The weight of the water that does not drain, but remains stranded above the water table, must then also be supported by the soil matrix and so contributes to the loading. The degree of saturation can be expressed as:

$$S_r = \frac{V_w}{V_v} \qquad (3)$$

V_w is the that portion of the void volume occupied by water (Dunn et al 1980). In real marsh soils, full saturation is never reached, as roots introduce and trap gases inside living membranes that are not displaced by water (Naidoo et al. 1992).

RESULTS AND DISCUSSION

Sedimentation and organic soil development can lead to aggradation (Nyman et al. 1990). Conversely, loading of the surface and consolidation of underlying soil layers contributes to settlement (Kaye and Barghoorn 1964). If relative position within the tidal frame is one of the primary factors affecting salt marsh survival, then it is important to relate observed changes in surface elevation to short and long-term processes that operate (1) at the surface, (2) within the root zone, (3) in that portion of the shallow subsurface monitored by the SET, and, finally, (4) in the deeper subsurface.

Cahoon et al. (1995a) have shown how SET and accretion data can be combined to estimate shallow subsidence. Here, we examined void ratios to estimate compaction or expansion within the upper 25 cm where wetting and drying, as well as root growth and decomposition might be expected to cause deviation from a normal consolidation profile. It was then possible to estimate how much consolidation must take place in the substratum between the marsh cap and the base of the SET support pipe (4 m).

Finally, we calculated the effects of elevation and normal drainage on loading of the marsh substratum. This provided the basis for a geotechnical model proposed to explain observed differences in marsh elevation dynamics at the two sites.

Sedimentation, Aggradation and Accretion

Short-term sedimentation on the marsh surfaces (DEP) did not vary significantly within sites with distance from the bayou or pond, but was four times greater at BC than at OB (Table 1). Estimated daily rates spanned two orders of magnitude (Fig. 3). High DEP at both sites was associated with Hurricane Andrew in August 1992, but elevated

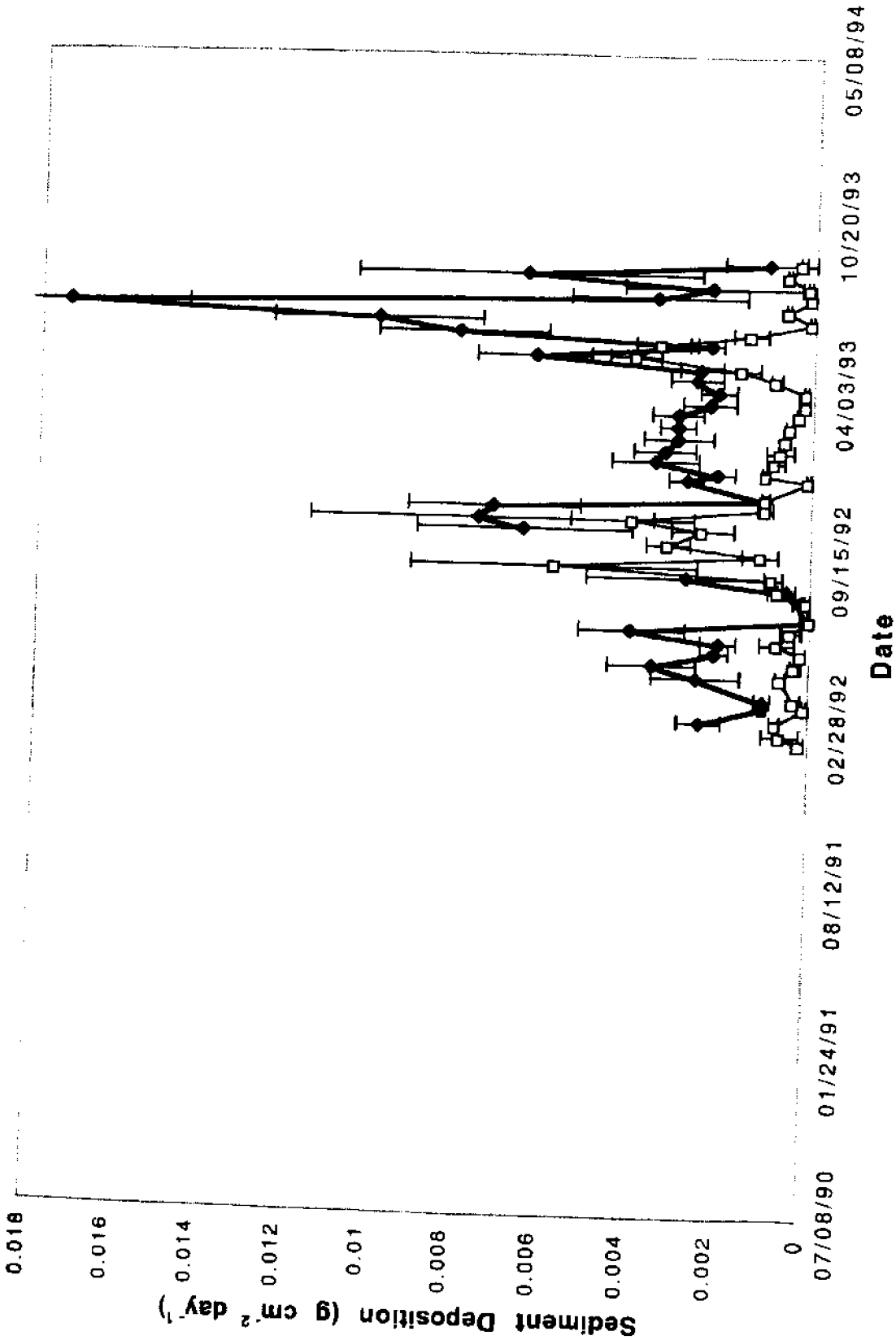

Figure 3. Short-term sedimentation (DEP) at Bayou Chitigue (closed) and Old Oyster Bayou (open): January 1992 - September 1993. Mean ± 1 SE.

Table 1. Marsh elevation at core locations (E), short-term sedimentation (DEP), elevation change at SET station (E_{set}), marker layer accretion (ACC) and shallow subsidence (S) from October 1991 to October 1993; mean ± S.E.. * indicates significant difference between sites ($p < 0.01$).

	Bayou Chitigue	Oyster Bayou
* E cm, MSL	3.5 ± 2.5	12.4 ± 0.4
* DEP g cm^{-2} d^{-1} g cm^{-2} y^{-1}	0.004 ± 0.0005 1.46	0.001 ± 0.0001 0.37
* Eset 2 years cm y^{-1}	-0.29 ± 0.48 -0.15	+0.99 ± 0.13 +0.50
ACC 2 years cm y^{-1}	6.66 ± 0.53 3.33	4.06 ± 0.50 2.03
S = ACC - E_{set} cm y^{-1}	3.48	1.53

rates of deposition persisted at both sites for at least two months following the storm (Cahoon et al. 1995b). Increased deposition also occurred at OB during the period of high river discharge in April and May, 1993, and slightly later at BC in May and June of the same year. Despite its proximity to the unleveed mouth of the Atchafalaya River, sedimentation at OB was limited by the 10 cm difference in marsh elevation and lower flooding frequency (Fig. 2).

The mean rate of sediment accumulation (ACC) measured by marker layer accretion was 40 percent greater at BC than that at OB, but this difference was not statistically significant, primarily because so many stations could not be relocated after the hurricane (Table 1). The OB value is almost twice that reported by Cahoon et al. (1995a), but is only 28 percent higher at BC.

The pond shoreline at OB was stable but retreated at BC into the area monitored by the pond margin SET station. This resulted in an 8 cm decrease in elevation in one quadrant during the winter of 1991-92 (Day et al. 1994). This degradation coincided with vegetation death at this location and is similar to the soil collapse phenomenon described by Nyman et al. (1993) and Delaune et al. (1994). Data from the interior marsh SET stations are compared here, to focus on the more gradual elevation change that takes place prior to vegetation death.

Elevation measured by SET rises through the summer and fall and decreases in the winter (Fig. 4) This cyclic behavior may track the annual Gulf oscillation in sea level (Marmer 1954, Reed 1995b). The annual elevation range for the longer record at BC was 1.7 cm y^{-1}. Similar seasonal variations in marsh elevation have been reported by others (Boumans and Day 1994, Childers et al. 1993).

Elevation trajectories at the two sites diverged in the interval that included Hurricane Andrew. The

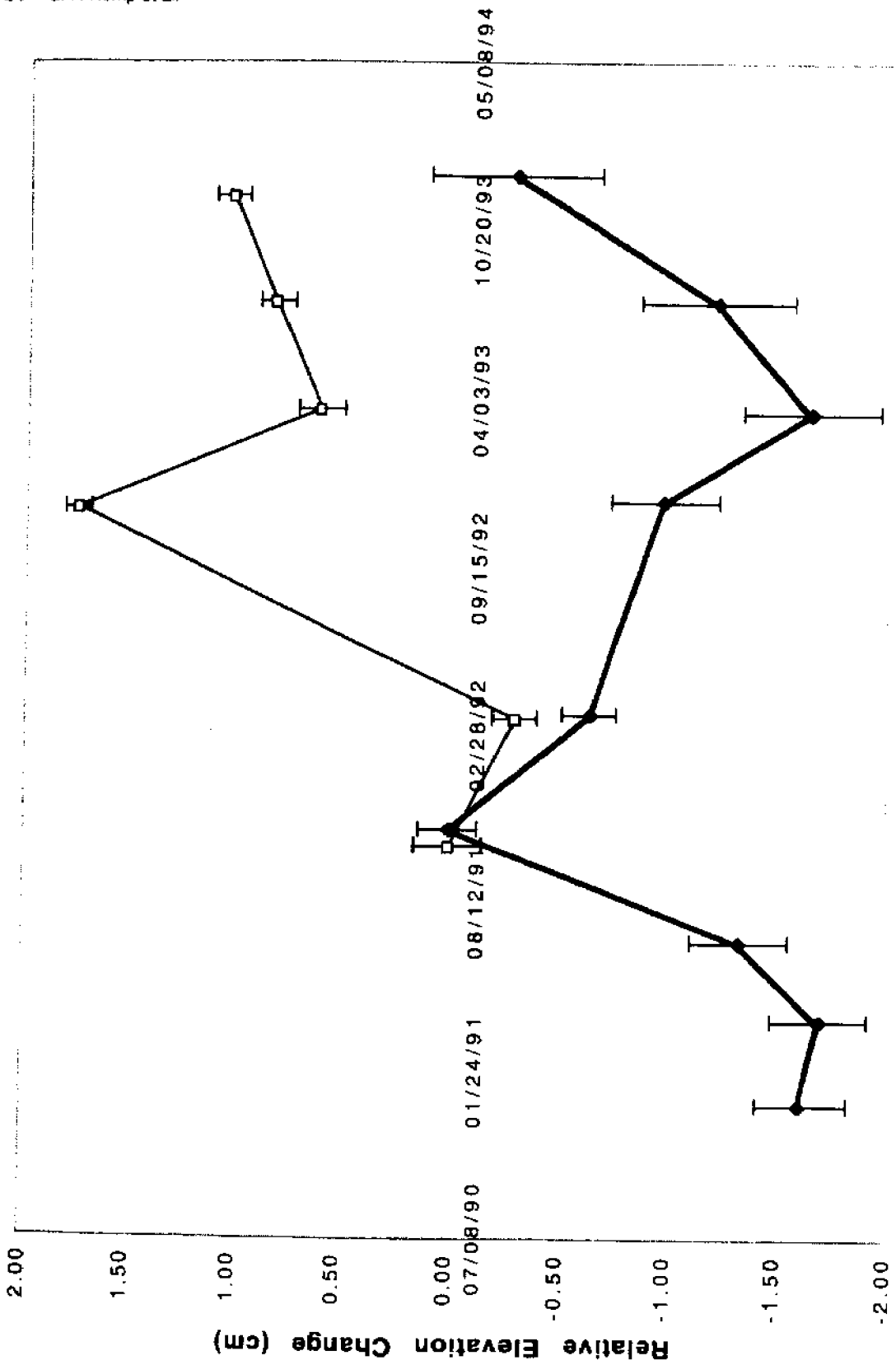

Figure 4. Marsh surface elevation change (E_{cs}) at Bayou Chitigue (closed) and Old Oyster Bayou (open): December 1991 - November 1993. Mean ± 1 SE.

OB marsh experienced more than 2 cm of aggradation, while the BC surface decreased 0.3 cm over the same interval. Elevation dynamics at the BC marsh were little affected by the storm and continued to fluctuate within the range previously established, while the marsh at OB shifted upward into a new elevation range (Fig. 4).

Net elevation changes at the two sites differed significantly (Table 1). Elevation at OB moved upward at 0.5 cm y^{-1}, while the net change at BC was not significantly different from zero. These values compare with annual rates at OB and BC of 0.65 and 0.15 cm- y^{-1}, respectively, measured independently at nearby stations between 1992 and 1994 by Cahoon et al. (1995a).

Shallow Subsidence and Soils of the Root Zone

Shallow subsidence at BC was more than twice that at OB (Table 1). Cahoon et al. (1995a) reported S for the 1992-94 interval that was somewhat lower, 2.45 and 0.40 cm y^{-1} at BC and OB, respectively, but the 2 cm y^{-1} difference between the sites was common to both studies.

The soil core analyses provides more information on processes occurring in the upper 25 cm (Table 2). Mean bulk density, g_d, was significantly higher for the whole 25 cm at OB than at BC ($p = 0.0002$), but only the upper 5 cm differed significantly ($p = 0.003$) when individual layers were compared between sites. %OM was significantly lower for the whole core at OB than at BC ($p = 0.026$), and was also significantly lower for the upper 5 cm ($p = 0.011$). The upper 5 cm at OB was higher in bulk density and lower in organic matter than samples from greater depths at both sites.

The calculated void ratios indicate that voids occupy between 8 and 10 times more soil volume than the soil solids in the samples from the two sites. On average at BC and OB, 90 and 88 percent, respectively, of the upper 25 cm of soil consists of void space (Table 2). The mean void ratio for the entire 25 cm was significantly lower at OB than at BC ($p = 0.001$). The mean void ratio for the upper 5 cm at BC was nearly twice that at OB, but again this was the only layer in which the sites differed significantly ($p = 0.016$). Void ratio increased from the surface to about 15 cm at OB in the zone of active root growth, and then dropped off, while the BC cores showed less variation with depth. The difference in void ratio noted between the sites at the surface disappears with depth (Table 2).

This analysis focuses on differences between the two sites. But these are relatively subtle considering that these two salt marshes exist at opposite ends of a sustainability scale. The summary data masks differences along each of the transects, which both showed a decline in the mineral sediment content of the upper 25 cm of the marsh with distance from the bayou. This trend was not present in the organic matter content, which was uniform along both transects and between sites, as has been noted more generally for the Mississippi deltaic plain by Gosselink et al. (1984). With these caveats, the results suggest that the vegetation dominated soils of the marsh cap remain relatively unchanged as long as the surface is above that threshold in the tidal frame at which waterlogged plants die.

Void ratios determined for the 0-5 cm surface layers should be representative of newly deposited sediments, given the measured accretion rates (Table 1). If the type of sediment deposited does not change greatly from year to year, the variation in mean void ratio with depth below the surface layer at the two sites should indicate whether significant consolidation is occurring in this zone. Void ratios generally increase with depth at both sites, suggesting that the surface is overconsolidated, but end up at the same value for the 20- 25 cm layer (Table 1). This suggests that most consolidation associated with this layer is contemporaneous with deposition and takes place at the surface, presumably as a consequence of desiccation rather than loading. Below the surface, void space is being introduced, possibly as a result of root growth and infauna activity (Frey and Basan 1978). Shallow subsidence, then, as monitored by the SET, must take place primarily below the upper 25 cm for which soil data are available.

Breuil (1994) determined void ratios on a smaller number of samples from cores acquired earlier at the two sites in spring, 1993. At that time

Table 2. Bulk density (γ_d), percent organic (%OM), specific gravity of soil solids (G_s) and void ratio (e) from salt marsh core soil samples; mean values and 1 S.E. Significant differences between sites indicated by ** ($p < 0.01$) or * ($p < 0.05$).

	Bayou Chitigue n = 4	Oyster Bayou n = 5
γ_d, g cm-3		
0.5 cm**	0.23 ± 0.02	0.38 ± 0.04
5-10	0.24 ± 0.03	0.25 ± 0.02
10-15	0.20 ± 0.03	0.22 ± 0.02
15-20	0.21 ± 0.03	0.27 ± 0.02
20-25	0.23 ± 0.02	0.23 ± 0.03
Mean**	0.22 ± 0.01	0.27 ± 0.02
%OM		
0-5 cm*	18.15 ± 1.00	13.68 ± 1.21
5-10	21.40 ± 2.29	24.70 ± 1.72
10-15	23.83 ± 2.83	18.40 ± 1.79
15-20	23.03 ± 2.63	19.62 ± 2.12
20-25	22.03 ± 2.36	21.42 ± 3.24
Mean*	21.69 ± 1.02	19.56 ± 1.14
G_s		
0-5 cm**	2.34 ± 0.01	2.41 ± 0.02
5-10	2.30 ± 0.03	2.25 ± 0.03
10-15	2.26 ± 0.04	2.34 ± 0.03
15-20	2.27 ± 0.04	2.32 ± 0.03
20-25	2.29 ± 0.03	2.30 ± 0.05
Mean*	2.29 ± 0.01	2.32 ± 0.02
e		
0-5 cm*	9.80 ± 1.11	5.64 ± 0.73
5-10	9.23 ± 1.10	8.08 ± 0.45
10-15	10.93 ± 1.20	10.34 ± 1.43
15-20	10.54 ± 1.52	7.75 ± 0.57
20-25	9.40 ± 0.86	9.43 ± 1.08
Mean**	9.98 ± 0.51	8.25 ± 0.50

the SET measurements show that the OB marsh was at the highest elevation recorded during the study (Fig. 4), less than half a year after Hurricane Andrew. Mean void ratios at that time from the upper 30 cm were 30 to 40 percent lower than for the cores described here, which were obtained eight months later (Table 2). Differences in technique cannot be ruled out, but these results suggest that seasonal changes in void ratio may be occurring within the upper 25 cm as a consequence of dewatering, root growth, infauna or decomposition that may explain some of the short-term variations in elevation.

Sediment Capture

Comparing measured accretion, ACC, with that predicted from DEP can provide additional insight into factors that influence the accumulation of sediments at the two sites. A vertical accretion rate can be predicted from annual DEP using the bulk density of the upper 5 cm, as

$$H_{DEP} = \frac{DEP_{annual}}{\gamma_{d\ 0\text{-}5\ cm}} \qquad (4)$$

H_{DEP} was 6.4 cm y^{-1} at BC, twice the observed accretion at this site (Table 1), and 2.5 times that measured by Cahoon et al. (1995a). H_{DEP} at OB was 1.0 cm y^{-1}, half of ACC in this study (Table 1), but comparable to that reported for this site by Cahoon et al. (1995a).

The sediment trap method provides a good standardized measure for comparing sedimentation potential at different sites. But it is still possible that the capture efficiency of the traps may be more or less than that of the marsh surface itself. At BC, it appears that more than half of the sediment mass that is trapped on the artificial surface is not incorporated in the accreting surface layer. Resuspension during the long period required for consolidation of the deposit from a fluid mud at this more continuously flooded site may reduce the effective accretion rate from the potential predicted by the trap data. Accretion predicted from DEP is in closer agreement with observed accretion at the infrequently flooded OB site.

These results suggest that while an elevated position within the tidal frame reduces the potential for sedimentation, it appears to positively affect the capture and incorporation of available sediment into the deposit. It is probable that because of its proximity to the river, the OB marsh would have more sediment available to it than the marsh at BC if it were at a comparable lower position within the tidal frame (Huh and Rouse 1994). But the difference in aggradation observed at the two sites is not simply explained in terms of sediment supply. Further, this difference cannot be attributed to consolidation within the upper 25 cm. The 2 cm y^{-1} difference in shallow subsidence must then be related to settlement occurring within the remainder of the 4 m zone monitored by the SET.

Effective Stress and Soil Strength

No soil data is available below 25 cm where consolidation must be taking place. Collapse and decomposition of the root material that provides much of the soil structure can cause marsh degradation without any increase in loading (Nyman et al. 1993, Delaune et al. 1994). It is also possible that a more gradual divergence in the loading history may affect soils below the marsh cap in a similar way prior to vegetation death. While an increase in the load applied to the marsh substratum below 25 cm may cause settlement, it can also increase the capacity of these soils to withstand greater loads. To test this hypothesis, we combined information about the hydrology, elevation and marsh soil composition to calculate the theoretical loading that would be applied to the substratum at the two sites during normal tidal drainage (Dunn et al. 1980).

Sediments accumulate and add mass at the surface. This can affect surface elevation relative to mean sea level (MSL), as apparently occurred after the hurricane at OB (Fig. 4). As has been discussed, however, much of the effective stress results from drainage during low tides that exposes the marsh substratum to the full weight of the drained surface layer. The load that must be calculated, then, is a function of the thickness and effective weight of the soil column from which supporting water is withdrawn. For any soil column, the effective stress is given by

$$\bar{\sigma} = \sigma - \upsilon \qquad (5)$$

where σ is the total normal stress, and υ is that portion supported by water (Dunn et al. 1980). H is the thickness of the soil layer above the depth for which the effective stress is to be determined, 0.25 m in this case. If drainage occurs, H is divided into an upper drained portion, H_d, and a lower, undrained or saturated section, H_s. The total normal stress is given by

$$\sigma = \gamma_w H\left(\frac{G_s + e}{1 + e}\right) + \gamma_w H_d S_r V_v \qquad (6)$$

where the result is in units of pressure (kPa) (Dunn et al. 1980). The first term inside the brackets is the weight of the soil solids and the second is that of the water that remains within the soil matrix following drainage. Buoyant support provided within the saturated section is

$$\upsilon = \gamma_w H_s \qquad (7)$$

The effective stress, thus, increases with the specific gravity of the soil solids, depth of drainage, and the moisture content of the drained layer.

The effective stress applied to the marsh substratum at the two sites was computed for tide levels from 25 cm above mean sea level to 25 cm below (Fig. 5). Layer-specific values of G_s and e were used directly (Table 2) for marsh surface elevations at BC and OB of 5 and 15 cm, MSL, respectively (Table 1). The drained soils were uniformly assumed to be at 80 percent of saturation ($S_r = 0.8$).

For tide levels greater than the marsh surface, only the buoyant weight (< 0.4 kPa) contributes to the load. The buoyant weight differs by only 0.08 kPa between the sites (Fig. 5). As the tide level drops below the surface at OB, however, the load at this site increases dramatically, when the full weight of the drained soil and retained water begins to apply. At mean sea level, when 15 cm of the soil column is exposed at OB, the effective stress at 25 cm below the surface is an order of magnitude greater than when the marsh is flooded. For this tide level only 5 cm below the BC marsh surface, loadings due to the marsh cap at BC and OB were estimated at 1.16 and 2.92 kPa, respectively. Of this difference, 95 percent is attributable to the greater thickness of the drained layer at OB, a function of its higher position within the tidal frame.

The theoretical difference in loading resulting from normal drainage should be reflected in the bearing strengths of the marsh substrata at the two sites. Given that shallow subsidence at BC is twice that at OB (Table 1), it is apparent that any added loading at OB is not resulting in proportional settlement, but is, instead, contributing to soil strength in the marsh substratum. If increased drainage occurred during a storm, it is likely that settlement at BC could be far greater than that associated with normal tides, and would eventually force the marsh surface below the critical threshold for marsh plant survival.

A GEOTECHNICAL MODEL FOR SALT MARSH LOSS

Recent regional mapping of the depths of radiocarbon dated peats indicates that RSLR at the Bayou Chitigue (BC) site is approximately 1 cm y^{-1}, while that at Old Oyster Bayou (OB) is estimated at 0.5 cm y^{-1} (L. D. Britsch, U. S. Army Corps of Engineers, New Orleans District, pers. com.). If this difference is correct, the BC marsh has always had a greater submergence potential relative to OB. Aggradation at BC would have to be twice that at OB to maintain a sustainable position within the tidal frame. But an apparent local geologic predisposition to submergence is not required to cause the observed divergence in marsh elevation trajectories. A similar outcome could also result from shifts in the sedimentation regime that are a normal part of deltaic evolution (Frazier 1967).

The OB and BC marshes exist at the upper and lower bounds of the tidal frame. Elevation at BC is 10 cm lower on average than at OB and is not increasing at a sufficient rate to offset any RSLR greater than zero, much less the 1 cm y^{-1} estimated for this site. While accretion is greater at BC, shallow subsidence is similarly higher, resulting in no aggradation. As mean sea level rises with respect to the marsh surface, the depth of normal drainage decreases, and the load imposed on soils underlying the root zone is reduced proportionally. As the marsh deposit continues to develop under a reduced loading regime, soils below the marsh cap will experience less consolidation and develop less bearing strength.

A precarious equilibrium develops at the lower end of the tidal frame. The BC marsh essentially "floats" over a weak soil substratum until a drainage or deposition event irreversibly depresses the marsh surface below the point of no return. Consolidation of the substratum is deferred, and then occurs suddenly as a precursor to the collapse phenomenon described by Nyman et al. (1993) and Delaune et al. (1994).

At the upper end of the tidal frame, sedimentation is limited by marsh elevation as existing marsh trajectory models predict (Allen 1994, French 1993, Krone 1987). The OB marsh captures virtually all of the smaller amount of sediment available to it as newly deposited sediments rapidly undergo desiccation and consolidation. This syndepositional consolidation adds strength to the surface layer without increasing shallow subsidence and contributes to 0.5 cm y^{-1} of aggradation, a rate that

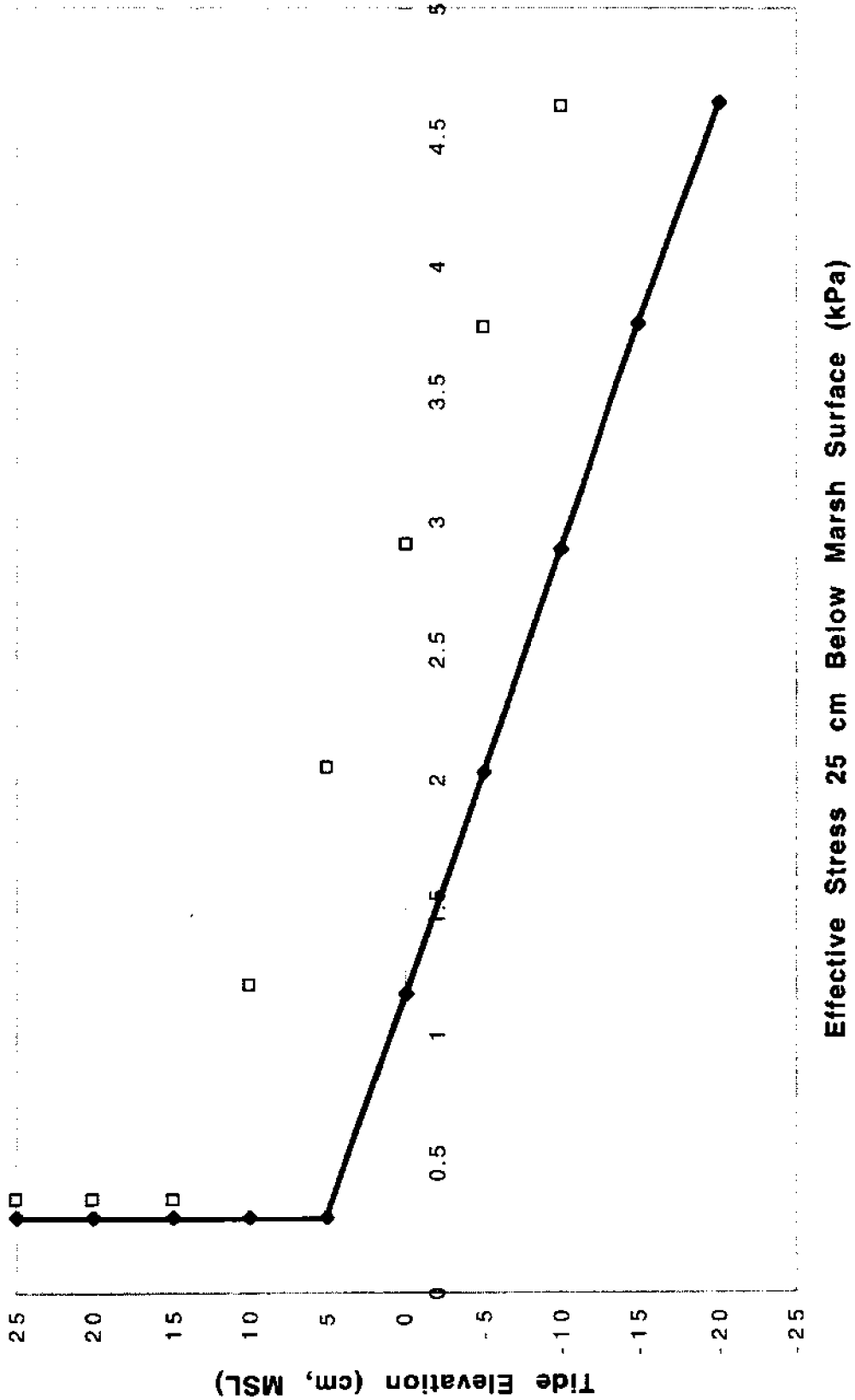

Figure 5. Calculated loading at 25 cm below the marsh surface for Bayou Chitigue (closed) and Old Oyster Bayou (open) for tide elevations from +25 to -25 cm, MSL. Mean ± 1 SE.

equals RSLR for this area. A higher elevation at OB generates greater loading of the marsh substratum, but this does not increase settlement (Table 1). Marsh elevation may not increase at any rate greater than RSLR, but continued sedimentation and drainage during storms (Fig. 4) act over time to consolidate the substratum and increase its strength (Stumpf 1983). The substratum becomes overconsolidated with respect to the effective stress applied during normal tides. Minor depression of the OB marsh surface during prolonged drainage does not imperil the viability of the plants, as they start off so high within the tidal frame, and elevation is quickly restored by increased sedimentation.

Accretion and aggradation is limited at the lower bound of the tidal frame by a low sediment capture efficiency. This factor has not been parameterized in existing marsh trajectory models (Allen 1994, French 1993, Krone 1987). Deposit formation appears to require a consolidation step following sedimentation. Without some consolidation, newly deposited sediments may be repeatedly resuspended if flooding is prolonged. This resuspension differs from erosion in that it occurs above the surface monitored by the SET or by standard accretion techniques. Sediment supply at BC appears to be enhanced as a consequence of the higher frequency and duration of flooding, but may be overestimated if much of the material introduced is repeatedly resuspended and does not permanently deposit on the marsh surface.

We originally hypothesized that the divergent history of the two sites with respect to marsh loss could be related to differences in sediment supply, soil strength or local subsidence. These results indicate that all three factors play complex, interrelated roles. The dominance of any one may shift over time.

At some point, the BC and OB marshes must have started at a comparable elevation within the tidal frame. At OB, the moderate deep subsidence rate has been countered by an increasing regional availability of suspended sediment as the Atchafalaya River distributary has grown in discharge and extended toward the OB site.

In contrast, the BC marsh has experienced a withdrawal of its fluvial sediment source as nearby Mississippi River distributary channels were abandoned (Penland et al. 1987). This reduction caused the marsh to begin a decline through the tidal frame. As marsh elevation at BC decreased toward MSL, shallow subsidence of the marsh substratum became a more significant contributor to degradation. It is likely that the decline at BC has accelerated, until now the marsh teeters at the threshold for survival, waiting for a final precipitating event to push it below the threshold for plant viability and into soil collapse.

Analysis of the end members of a sustainability gradient unfortunately provides little direct indication of the amount of river influence that is required to reverse degradation for marshes that are currently at intermediate points in the decline through the tidal frame. Further, the soils of the marsh cap change only subtly through the decline leading to collapse and, ultimately, submergence. For more direct evidence, we will need in future to look at additional sites between Old Oyster Bayou and Bayou Chitigue, and obtain direct measurements of in situ soil strength and consolidation extending into the marsh substratum.

The results of this work can, however, provide those responsible for designing river diversions and other restoration measures with guidance on how to evaluate the restorability and sediment requirements of receiving tidal marshes. The position of the marsh surface relative to the top of the tidal frame should be the most direct indicator of sustainability. Those that are at the upper bound will require less added sediment, while those near the bottom may be beyond restoration except by direct placement of dredged material. It is difficult, however, to accurately discern elevation in vast estuarine basins without benchmarks. Alternatively, as observed by Pestrong (1969), we expect that soil strength in the upper meter should be positively correlated with position in the tidal frame, and may provide a good field indicator of restoration potential.

ACKNOWLEDGMENTS

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A Preliminary Assessment of Wave, Current, and Sediment Interaction on the Louisiana Shoreface Adjacent to the Isles Dernieres

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ABSTRACT: The Louisiana coast is generally characterized as a low wave-energy environment where sediment transport is dominated by the influence of the Mississippi and Atchafalaya Rivers. Winter cold fronts, however, generate waves and currents that have a significant impact on a variety of Louisiana's coastal environments, although field data regarding their influence on the inner shelf are extremely sparse. During a 12-d period that included the passage of two cold fronts, waves and near-bed currents were measured on the Louisiana inner-shelf (depth ~ 8 m) using a sophisticated bottom-mounted instrumentation system. Bottom boundary layer parameters were then calculated using wave-current interaction models, and sediment transport was predicted by assuming steady state turbulent diffusion within and above the wave boundary layer.

Results indicate that the second front (Front 2) was the more energetic of the two. A maximum significant wave height of 1.33 m and maximum current speed of 0.21 m s^{-1} occurred during this event. Additionally, mean current-induced shear velocity (2.95 cm s^{-1}) and wave-current shear velocity (4.99 cm s^{-1}) were highest during this event's frontal and prefrontal stages, respectively. During the postfrontal stage, currents were strong and well organized, although combined shear velocities were low as a result of reduced wave height. Predicted sediment transport varied considerably in direction and magnitude throughout the deployment, but was highest (12.7-16.2 mg cm^{-1} s^{-1} toward the southeast) during the prefrontal and frontal stages of Front 2. Fair weather transport was low and to the west. Thus, winter cold fronts are likely an important mechanism for sediment movement on the Louisiana inner shelf, although the associated transport direction and magnitude require further quantification.

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Introduction

The Mississippi and Atchafalaya Rivers introduce vast amounts of sediment into the wetland, estuarine, shoreface, and shelf systems along the northern Gulf of Mexico, particularly along the Louisiana coast (Crout and Hamiter 1979). Although much of this material is deposited locally, a considerable amount of fine material is transported with prevailing currents as suspended sediment plumes and deposited offshore. Not surprisingly, therefore, these fluvially-derived sediments serve as important sources of depositional material on the shoreface and continental shelf (Crout and Hamiter 1979; Adams et al. 1987; Roberts et al. 1987; Wright et al. 1997).

In contrast, the importance of entrainment and transport of inner-shelf bottom sediment by waves and currents along the Louisiana coast is poorly documented and quantified. Entrainment of sediment from the bed requires the combined action of waves and currents to generate a shear velocity (u_*) that exceeds a critical threshold determined predominantly by sediment diameter. Since the northern Gulf of Mexico is generally considered a low-energy environment, sediment transport on the Louisiana inner shelf during fair weather is likely minimal (Wright and Nittrouer 1995; Jaffe et al. 1997; Wright et al. 1997).

The passage of cold fronts, however, is a notable exception to these low-energy, fair-weather conditions. Occurring with a frequency of roughly 30 times yr^{-1}, chiefly between November and April (Roberts et al. 1987), the passage of cold fronts generates important hydrodynamic and sedimentary responses in various coastal environments in Louisiana, including deltaic wetlands (Murray et al. 1993), the chenier plain (Roberts et al. 1987), and barrier islands (Dingler and Reiss 1990; Stone and Wang 1999). Data on inner-shelf bottom boundary layer and seabed responses to frontal passages in this region, however, are sparse. With the exception of Jaffe et al. (1997), who modeled sediment transport using representative values of wave and current parameters rather than direct field measurements, no published data for south-central Louisiana are available.

Our objective is to discuss the results of a 12-d instrumented field deployment that included two cold front passages and two intervening low-energy periods. Waves, near-bottom currents, and bottom boundary layer parameters are quantified and used to predict sediment transport magnitude and direction. The results are significant in a practical sense, given that the inner shelf is an important component of the sedimentary system that includes south-central Louisiana's barrier islands and coastal wetlands, which are currently experiencing extremely high rates of erosion.

Materials and Methods

Water level, wave, current, and seabed elevation data were collected from November 20 to December 1, 1997, at an 8-m deep, sandy-bottomed, site on the Louisiana shoreface (Fig. 1) using a bottom-mounted instrumentation system named WADMAS (Fig. 2). The system included a Fluxgate™ compass, a Paroscientific™ pressure transducer, a Digisonics™ sonar altimeter, and a vertical array of three bi-axial Marsh-McBirney™ electromagnetic current meters (at elevations of 20, 67, and 120 cm above the bed). Sensors were programmed for burst-mode sampling; specifically, the pressure sensor and current meters sampled for 8.2 min h^{-1} at a frequency of 4 Hz, while the compass and altimeter recorded one measurement every 30 min.Samples of bottom sediment were obtained using a grab-sampler, and later were dry-sieved to determine mean grain size. Additionally, hourly meteorological data from C-MAN Station GDIL1 (Grand Isle) and daily weather maps from the Southern Regional Climate Center in Baton Rouge, Louisiana were acquired.

Meteorological events were analyzed using a qualitative approach in which each event was subdivided into four stages on the basis of changes in wind velocity. Stages included fair weather, and three frontal stages: pre-frontal, frontal, and post-frontal. The threshold established for wind speed associated with pre- and post-frontal conditions was the mean value for the study period plus one standard deviation. The beginning of the pre-frontal

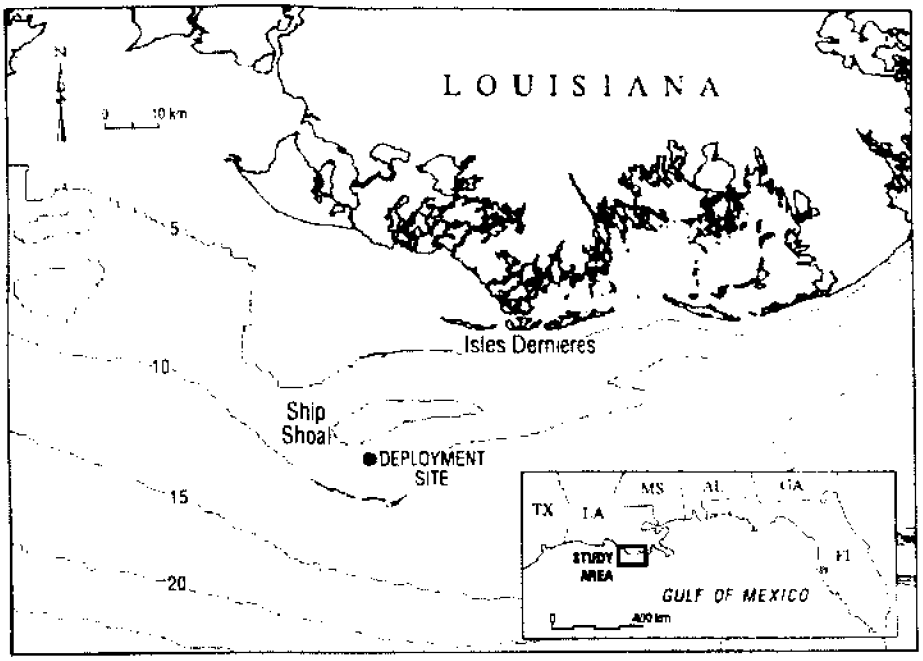

Fig. 1. The Study Area. Instrument was located at 28°50.68′N, 91°07.52′W. Depth contours are in meters.

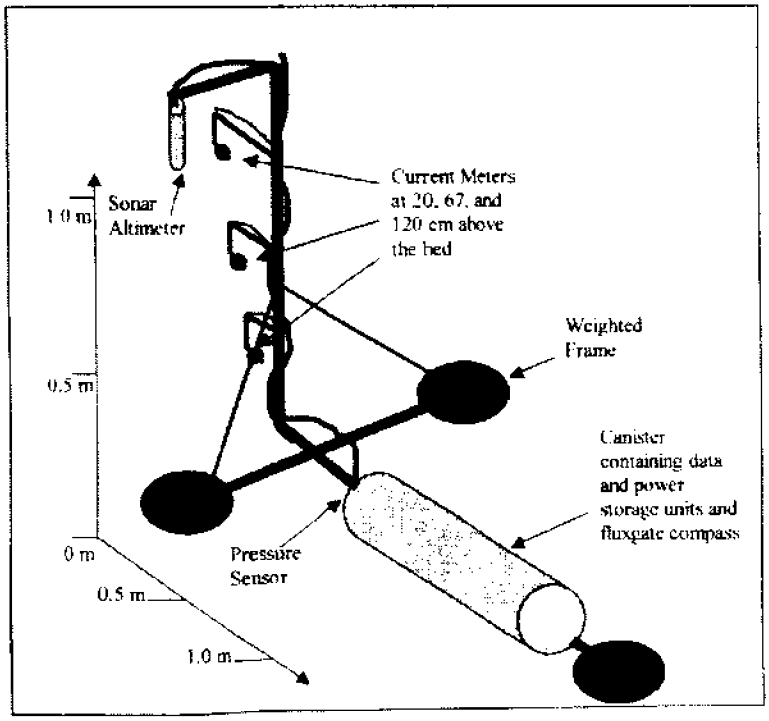

Fig. 2. Schematic diagram of the WADMAS instrumentation system.

phase of the storm was identified as the hour when the wind exceeded this threshold and blew from the south (i.e. between 90° and 270°, measured clockwise). The frontal phase encompassed the period when winds were variable in direction and less than the threshold speed. The post-frontal phase was defined as the interval during which the wind blew from a direction between 270° and 90° at a speed exceeding the threshold. All other wind conditions were considered fair weather.

Significant wave height (H_s), peak wave period (T_p), and mean wave direction (θ_w) were calculated from the pressure and current-meter data using cross-spectral analysis, with horizontal-, and pressure-attenuation correction factors applied to compensate for signal decay with depth (Earle et al. 1995). Current-velocity profiles were generated using the log-profile method, which involves log-linear regression of the burst-averaged current meter velocities (Drake and Cacchione 1992). Two conditions were assumed necessary for a profile to be considered logarithmic in a statistically significant sense: first, a correlation coefficient (r^2) > 0.994 (Drake and Cacchione 1992); and second, a mean directional variation between current meters < 30°. Hourly measurements that were not logarithmic were excluded from the analysis. Current-induced shear velocity (u_{*c}) and apparent bottom-roughness length for all logarithmic profiles were calculated using the von Karman-Prandtl equation:

$$u(z) = u_{*c}/\kappa \ \ln(z/z_{0c}) \qquad (1)$$

where u(z) is the horizontal velocity at height z above the bed, and κ is von Karman's constant (0.4). Once the sediment concentration in the water column had been predicted (discussed later in this section), the shear velocity calculations were iteratively modified to account for the possible effects of suspended sediment induced stratification. To do so, the buoyancy parameter (Z/L) employed in the model introduced by Glenn and Grant (1987) was used.

The Grant-Madsen (1979, 1986) model was used to account for the combined influence of waves and currents. According to the model, a wave boundary layer (wbl) of thickness (δ^*_w) develops during wave activity, and the velocity profile is defined separately within and above this layer as:

$$u_c = \frac{u_{*c}}{\kappa}\left(\frac{u_{*c}}{u_{*cw}}\right)\ln\frac{z}{z_{0l}}, \qquad z \le \delta_w$$

$$u_c = \frac{u_{*c}}{\kappa}\ln\frac{z}{z_{0c}}, \qquad z \ge \delta_w \qquad (2)$$

where u_{*c} and u_{*cw} are the current and combined wave-current-induced shear velocities, respectively, z is the height above the bottom, z_0 is the roughness produced by the sand grains (=D/30, where D is the mean grain diameter), and z_{0c} is the apparent bottom roughness experienced by the current above the wave boundary layer. z_{0c} was used because the current experiences drag due to the combined influences of physical elements (grain roughness and bed forms) as well as non-linear interaction with the wave boundary and mobile bedload layers (Gross et al. 1992).

The assumptions were made that the current-induced shear velocity, u_{*c}, acts in the same direction as the mean current, and that the direction of u_{*cw} oscillates during the course of the wave cycle. As such, when the wave orbital velocity is at a minimum (near zero), the direction of u_{*cw} is the same as that of the current; when it is at its maximum, its direction (φ_{max}) is between the wave and current directions, given by [modified from Cacchione et al. (1994)]:

$$\varphi_{max} = \arctan\left(\frac{\sin\phi}{\cos\phi + \frac{u_b}{u}}\right) \qquad (3)$$

Sediment transport was estimated based on the assumption of steady-state, upward, turbulent diffusion of sediment through the water column. First, an entrainment function was defined, based on the Yalin parameter (Ξ):

$$\Xi = [(\rho_s - \rho)gD^3/\rho\nu^2]^{0.5} \qquad (4)$$

where r_s and r are the respective densities of sediment (2.65 g cm^{-3}) and seawater (1.025 g cm^{-3}), D is the grain diameter, and v is the kinematic fluid viscosity (0.013 $cm^2 s^{-1}$). The critical Shield's criterion (θ_{crit}), and shear stress τ_{crit} were then calculated using:

$$\log\theta_{crit} = 0.041(\log^{\Xi})^2 - 0.356\log^{\Xi} - 0.977 \quad (5)$$

and $$\tau_{crit} = \theta_{crit}(\rho_s - \rho)gD \quad (6)$$

Normalized excess shear stress (S') was then defined as:

$$S = \left(\frac{\tau - \tau_{crit}}{\tau_{crit}}\right) \quad (7)$$

where t is the observed shear stress. This was then used to define the "near-bed" sediment concentration ($C(z_a)$):

$$C(z_a) = C_{bed}\frac{\gamma_0 S}{1+\gamma_0 S} \quad (8)$$

where C_{bed} is the sediment concentration in the bed (0.65) and γ_0 is an empirical constant with a value of 0.002. Suspended sediment concentrations were assumed to take the form of Rouse profiles, defined by:

$$C(z) = C(z_a)\left(\frac{z}{z_a}\right)^{-\alpha}, \text{ where } \alpha = \frac{\gamma w_s}{\kappa u_*} \quad (9)$$

γ is the ratio of the eddy diffusivity of sediment to that of momentum (~1), and w_s is the particle fall velocity.

Finally, burst-averaged sediment transport (Q) was calculated by integrating the velocity and sediment concentration profiles within, and above, the wave boundary layer such that:

$$Q = \frac{1}{t}\int_{z=\delta_w}^{z=\eta}\int_0^t uC\ dzdt \quad \text{for} \quad z > \delta_w$$

$$Q = \frac{1}{t}\int_{z=z_0}^{z=\delta_w}\int_0^t uC\ dzdt \quad \text{for} \quad z < \delta_w \quad (10)$$

where η is the sea surface elevation, and u is the current velocity.

Results and Discussion

Meteorological Conditions

Meteorological data indicated that the study period could be subdivided into two intervals of fair weather and two cold fronts (hereafter, Front 1 and Front 2). The fair weather phases lasted from 19:00 UTM on November 18 (prior to the deployment) to 18:00 UTM on November 21, and from 03:00 UTM on November 25 until 18:00 UTM on November 28. The fair weather phases were characterized by light (1.3-6.6 m s^{-1}) southerly or easterly winds; whereas both frontal passages were characterized by a sequence of strong southerly winds, followed by light and variable winds, and finally by strong northerly winds (Table 1 and Fig.3). Frontal passages differed from each other in several respects. Strongest winds during the pre- and post-frontal phases were from the south and northeast (respectively) in the case of Front 1 and from the southeast and northwest during Front 2. Most notably, however, these fronts differed markedly in intensity. Front 2, which had maximum pre-frontal and post-frontal wind velocities of 11.3 and 13.7 m s^{-1} was much more powerful than Front 1. Thus, the discussion will focus primarily on hydrodynamic and sedimentary responses associated with Front 2.

Hydrodynamic Responses

The influence of the frontal passages, particularly Front 2, on the wave field is shown in Table 2 and Fig. 4. During fair weather and during the first frontal passage, significant wave height was generally below 0.6 m. In contrast, during the pre-frontal stage of the Front 2, a maximum significant wave height of 1.34 m was measured. The trend in wave height was, not surprisingly, accompanied by a very similar trend in near-bed orbital velocity, which reached a maximum of 55 cm s^{-1} during the pre-frontal stage of Front 2. Patterns in wave period accompanying the frontal passages were less clear, although the pre-frontal and frontal phases of Front 2 were notable for the presence of comparatively long period waves, and peak period was observed

Table 1. Characteristics of the cold front passages. Note that "direction" follows the meteorological convention, indicating the direction from which the wind was blowing. Key: U_w = mean wind speed. [a] the first fair weather period began prior to the deployment.

Front	Phase	Arrival Time (m/d/h)	Mean U ($m\ s^{-1}$)	Range of U ($m\ s^{-1}$)	Dominant Direction
1	Fair	11/18/19:00 [a]	4.2	2.7 – 6.2	South
	Pre	11/21/18:00	5.6	4.3 – 6.9	South
	Front	11/22/4:00	3.1	1.5 – 4.3	Variable
	Post	11/22/21:00	6.7	2.9 – 9.0	Northeast
2	Fair	11/25/3:00	4.1	1.3 – 6.6	East
	Pre	11/28/18:00	7.0	3.6 – 11.3	Southeast
	Front	11/29/16:00	2.1	0.3 – 3.4	Variable
	Post	11/30/6:00	8.7	5.4 – 13.7	Northwest
ALL	—	—	5.3	0.3 – 13.7	—

to decline with the onset of the frontal episode. These patterns in wave height and period were likely caused by strong southeasterly winds blowing over a long fetch prior to the second frontal passage, allowing high swell waves to develop. Following the frontal passage, however, strong northerly winds likely generated choppy seas dominated by short, steep waves, whose period gradually increased through non-linear energy transfer as the post-frontal phase progressed.

Water level also appears to have responded to the wind shifts that occurred during the deployment, although with perhaps unexpectedly long lag times (Fig. 5). During both frontal passages, strong southerly winds caused a peak in water level, apparently due to set-up against the adjacent coast. Water level then decreased following the shift to northerly winds that accompanied both post-frontal stages. Unfortunately, the short data record does not permit a detailed discussion of water level responses to frontal passages, which may take place over several days.

Similar to wave height and water level, current velocity also responded noticeably to the prevailing meteorological conditions (Table 3 and Fig. 6). During both frontal passages, current direction was very nearly the same as wind direction for a

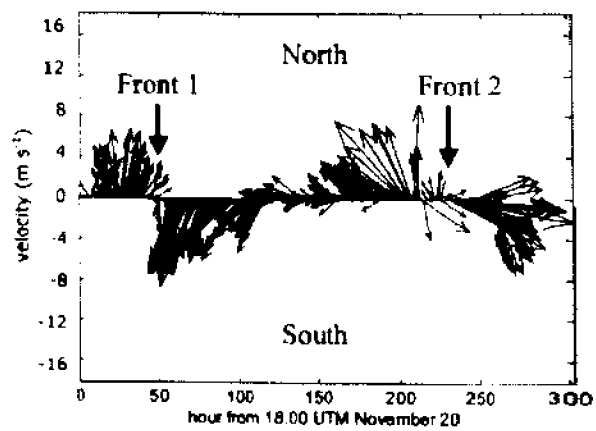

Fig. 3. Hourly wind velocity vectors ($m\ s^{-1}$). Arrows indicate the direction in which the wind was blowing (oceanographic convention).

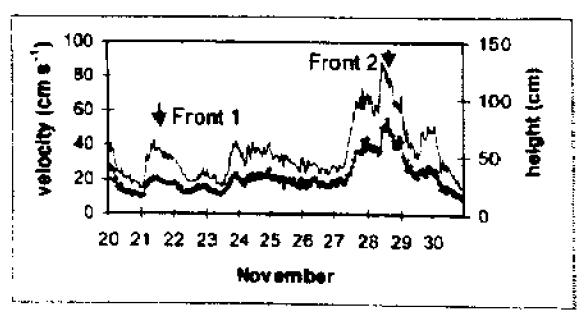

Fig. 4. Significant wave height (smooth line) and wave orbital velocity (marked line) during the study period.

Table 2. Wave characteristics during the deployment. Key: Hs = significant wave height; Tp = peak wave period.

Front	Phase	Hs (m)	Tp (s)	Orbital velocity (cm s^{-1})	Dominant Direction of Propagation
1	Fair	0.36	6.3	15.4	Northeast
	Pre	0.43	5.5	15.9	Northeast
	Front	0.51	5.7	18.4	Northeast
	Post	0.38	5.1	15.8	East
2	Fair	0.54	6.5	21.5	Northeast
	Pre	1.07	7.2	38.2	Northeast
	Front	0.67	7.6	29.5	Northeast
	Post	0.53	5.3	19.9	North
ALL	—	0.52	6.1	21.5	—

significant amount of time and thus rotated clockwise from northward- to southward-flowing as each front passed. The data suggest that currents were driven both by direct wind stress and by "inertial" forces resulting from relaxation of sea level set-up as discussed by Daddio (1977). Currents were strongest during the post-frontal phases of both passages when maximum mean current velocity at 120 cm above the bed reached 22 and 21 cm s^{-1}, respectively. This stands in contrast to orbital velocity, which was at its maximum during pre-frontal stages. The current direction during each of the pre-frontal periods differed between the two frontal passages. In the case of Front 1, post-frontal currents were predominantly southward; whereas during Front 2, currents remained northeasterly for the majority of the post-frontal stage before eventually rotating toward the south. Also of note during this deployment were the strong, steady, southward currents that dominated the second fair weather phase.

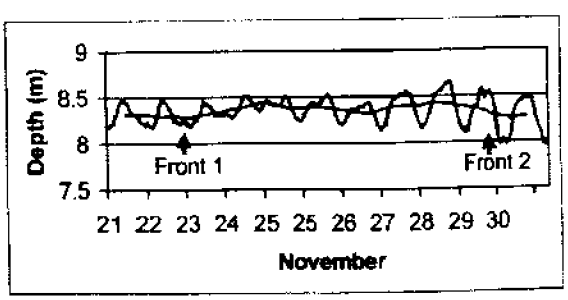

Fig. 5. Hourly water level and water level smoothed using a 24-h moving-average window.

Bottom Boundarty Layer Parameters

Current-induced shear velocity was strong, and logarithmic current profiles were well developed during post-frontal stages (Table 4), which is intuitively consistent with the presence of strong,

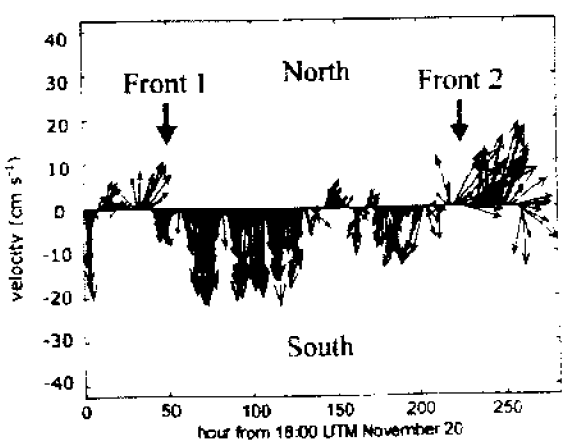

Fig. 6. Mean hourly current velocity vectors (cm s^{-1}) at 120 cm above the bed.

Table 3. Current speed (s) and direction at 20 (bot), 67 (mid), 120 (top) cm above the bed.

Front	Phase	s(top) (cm s^{-1})	s(mid) (cm s^{-1})	s(bot) (cm s^{-1})	Dominant Direction of flow
1	Fair	6.1	5.4	3.2	Variable
	Pre	6.2	5.3	3.4	North
	Front	6.9	5.5	2.3	South
	Post	14.0	12.0	7.3	South
2	Fair	7.5	6.2	3.0	South
	Pre	5.9	5.2	2.6	Variable
	Front	8.8	7.2	3.7	Northeast
	Post	13.4	11.4	6.6	Northeast
ALL	—	9.2	7.8	4.2	—

Table 4. Bottom boundary layer parameters calculated for all hourly bursts with logarithmic profiles. Key: z_{0c} = roughness length; u_{*c} = current shear velocity; u_{*cw} = wave-current shear velocity.

Front	Phase	z_{0c} (cm)	u_{*c} (cm s^{-1})	u_{*} (cm s^{-1})	% Logarithmic Profiles
1	Fair	5.0	1.41	2.05	29
	Pre	1.6	0.69	2.23	18
	Front	9.2	1.08	1.87	69
	Post	3.5	1.59	1.94	67
2	Fair	8.2	1.25	2.18	26
	Pre	7.1	0.97	3.08	11
	Front	8.9	1.90	3.08	23
	Post	7.0	1.75	2.10	47
ALL	—	6.1	1.44	2.12	40

steady currents at these times. It is more difficult to make generalizations regarding the combined wave-current shear velocity, because throughout all stages of Front 1, its value remained fairly constant, and low, relative to Front 2. Front 2, in contrast, illustrates the importance of high waves, rather than strong currents, in generating high combined wave-current shear velocities at the study site. Specifically, the highest shear velocities occurred during the pre-frontal and frontal stages, which were characterized by high waves. Shear velocity was low during the post-frontal stage, despite the presence of strong currents. This alone has unclear implications for the net movement of bed sediment within the water column, which requires a combination of entraining forces (shear velocity) and transporting forces (current flow). The sediment transport model accounts for this, however, and the results derived from this model are discussed in the following section.

Table 5. Predicted sediment transport. Key: Q = predicted transport; z<wbl = within the wave boundary layer; z<wbl = above the wave boundary layer; total = throughout the water column. Units are mg cm^{-1} s^{-1}.

Phase	Q (z<wbl)	Direction	Q (z>wbl)	Direction	Q (total)	Direction
1. Fair	7.26	122	1.48	175	8.25	270
Pre	1.23	313	0.21	324	1.43	348
Front	0.27	148	0.086	9	0.39	357
Post	5.17	126	1.65	175	6.39	269
2. Fair	2.23	119	0.19	150	2.40	276
Pre	11.90	120	1.11	168	12.67	168
Front	14.67	110	1.88	69	16.15	146
Post	7.4	357	2.57	49	9.20	142
Average	3.29	104	0.73	140	3.91	110

Sediment Transport

Fronts 1 and 2 differed considerably in terms of both sediment transport magnitude and direction (Table 5). Perhaps the most notable aspect of these results is the high transport rate associated with the pre-frontal and frontal stages of Front 2. Although the transport direction during these time periods is toward the southeast (essentially offshore), it is interesting to note that the transport during the prefrontal and frontal stages of Front 1 is roughly in the opposite direction. Since wave stresses are essentially bi-directional, it is possible that sediment transport direction during cold front passages is very sensitive to the specific meteorological characteristics of the front and the associated current direction, which serve to shift transport toward one component of the wave orbital flow. Furthermore, the majority of predicted transport occurred within the wave boundary layer during all stages, reflecting both the importance of waves in mobilizing sediment in this low-energy environment and the fact that sediment does not likely diffuse very high into the water column. Finally, our predictions indicate that fair weather periods are characterized by low rates of westward sediment transport.

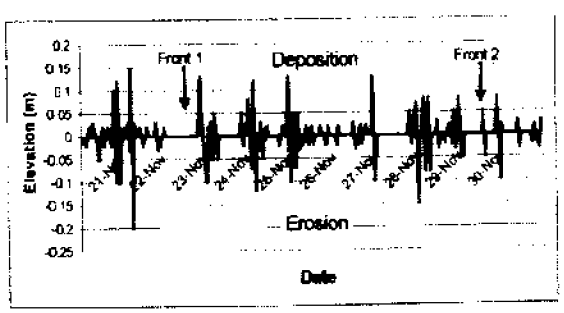

Fig. 7. Bed elevation (m) during the study. The mean bed elevation for the study period has been assigned a value of zero, negative values indicate bed erosion, and positive values indicate bed accretion.

Results from the sonar altimeter (Fig. 7) indicate appreciable movement of bed material throughout the study, including bed height fluctuations of up to 20 cm. These changes occurred over time-scales of hours and included alternating episodes of erosion and deposition, with the result that no net change occurred during the study period. The most logical interpretation for these fluctuations is that bed forms, such as sand ripples, were migrating beneath the altimeter throughout most of the deployment. The time series suggests no particular periodicity to these migrations, which should be expected, given the changing wave and current conditions that occurred. There is also no indication that rates of change were higher during frontal passages than during low energy conditions. There are several possible explanations for this. First, the 30-minute sampling frequency of the

altimeter would not have permitted bed form migration rates greater than one wavelength h^{-1} (as may have occurred during high-energy conditions) to be resolved. Second, rapid changes in the direction of wave and current stresses that accompanied the frontal passages may not have facilitated unidirectional bed form migration, even over a very short time scale. Finally, the increased importance of sediment entrainment and transport high in the water column may have obscured the effects of bed form movement during high-energy conditions. Nevertheless, these measurements are direct field evidence that bed stresses capable of moving sediment were operative during much of the deployment period.

Conclusions

1. High waves and wave-current shear velocities accompanied prefrontal and frontal stages of the cold front passages, facilitating potentially large sediment transport volumes within the wave boundary layer. Transport direction may, however, vary widely.
2. Strong, consistent, currents with highly logarithmic profiles tended to develop following frontal passages. Results do not suggest high sediment transport rates during these periods, however, likely owing to fairly low wave activity.
3. Fair weather periods were characterized by low predicted rates of sediment transport to the west.

Thus, although many researchers have characterized the Louisiana coast as one of low oceanographic energy where transport and deposition of sedimentary material is dominated by fluvial influences, this research indicates that winter cold front passages may generate waves and currents on the Louisiana inner shelf that are powerful enough to resuspend and transport sediment. The direction and magnitude of this transport, however, require further quantification.

ACKNOWLEDGMENTS

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Particle Image Velocimetry (PIV) and Numerical Modeling for Flow Estimation and Analysis

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ABSTRACT: In this paper an image-based measuring technique to collect water surface velocities is described. Unlike point-based measurements that require a substantial investment in equipment and field crews, the Particle Image Velocimetry (PIV) approach uses inexpensive video equipment to make digital images of the surface flow. Measurements can be made quickly and with little site preparation. PIV is well suited for shallow areas, where lack of access limits conventional point measurements. Furthermore, PIV is suitable for wetland and marsh waterways that often have velocities below the limit of detection of conventional devices. Numerical models, guided by the PIV measurements, are used to generate two- and three-dimensional flow-fields. This hydrodynamic information can be used to assess restoration strategies and perform water quality analysis in an economical and effective fashion. It can be also used to investigate the short- and long-term response of natural fluvial systems to human influences. As a first step toward validating the proposed methodology, field measurements were collected at Clear Creek near Oxford, IA. USGS Gage 05454220 is located immediately downstream of the reach under investigation. A three-dimensional flow field is produced for this reach using a kinematic numerical model with the PIV surface-flow measurements used as input. Comparison between profiles extracted from the three-dimensional flow field with point-measurements collected in the field showed good agreement. This study demonstrates that the PIV technology coupled with numerical modeling has a potential of being used as a practical alternative tool for flow estimation and analysis.

Introduction

Flow measurements provide critical information for the study of physical and biological processes in streams, lakes, and marshes. However, often no measurements are available and are difficult or expensive to obtain. Some coastal marshes in Louisiana, for example, are subject to micro-tidal fluctuations and irregular flooding. The tidal ranges and velocities are sometimes small and often below the thresholds of most of the conventional measuring devices. Therefore, it is difficult, if at all possible, to measure these velocities by methods such as electro-mechanical or electro-magnetic flow meters. Moreover, shallow tidal channels and marshes are inaccessible by boats or wading to collect point measurements. Therefore, a non-intrusive technique capable of detecting low velocities is needed.

In recent years, new measurement technologies have played a larger role in hydrologic studies. Advanced electronic sensing equipment, based on acoustic Doppler velocimetry (ADV), has been used to measure two- and three-dimensional flow-fields in natural waterways. Although ADV equipment is relatively expensive to purchase and operate, it has proven to be extremely valuable in flow measurement for complicated flow conditions (Oberg and Schmidt 1994). Remote sensing technologies have also been used in studies of river flooding. Brakenridge et al. (1994) used satellite

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information in the study of the 1993 floods on the Upper Mississippi and Missouri Rivers. For the Amazon River, Mertes (1994) combined estimates of sediment concentrations from Landsat images with flow information from a two-dimensional hydrodynamic model to estimate sediment exchanges between the main channel and floodplain, and rates of floodplain deposition and erosion.

Still, progress on many important scientific problems in hydrology, fluvial geomorphology, and river ecology, are hampered by a lack of available measurements of flow characteristics. Furthermore, there is a growing awareness that two- and three-dimensional components of stream and marsh flows are extremely important to the functioning of fluvial systems. This information on two- and three-dimensional flow characteristics is critical to biological and engineering investigations of wetland restoration strategies, analyses of water quality issues, and evaluation of impact of flow conditions on geomorphology and habitats (Mertes et al. 1995; Miller 1995).

This paper describes a new approach suited for measurement of streams, floodplain, and marsh flows. The approach utilizes Particle Image Velocimetry (PIV), commonly used in fluids research laboratories, for measuring free surface flow velocities.

Background

Particle image velocimetry (PIV) is a well-established technique in fluid mechanics research. In recent years, PIV has become a popular method for two- and three-dimensional flow measurement for laboratory-scale fluids experiments (Adrian, 1991). An appealing aspect of PIV is its inherent simplicity. First the flow is seeded with tracer particles that follow the fluid movement accurately. Then, using appropriate illumination, images of the region(s) of interest are recorded. If the tracer particle density is low, one can determine the displacement of individual particles between successive frames. Velocity is then calculated by dividing particle displacement by the time interval between successive frames. If the particle density is high, then the movement of groups of particles is determined using a statistical approach. The key advantage of PIV is that, whereas methods such as Laser-Doppler Velocimetry (LDV) or Acoustic Doppler Velocimetry (ADV) techniques provide point velocity measurements, PIV produces whole-field velocity measurements with similar accuracy. This attribute is rapidly making PIV the measurement method of choice for laboratory flow measurement in fluid mechanics research.

Surprisingly, PIV is not a well-known method outside of the fluid mechanics community, even though it has obvious applications in other disciplines. The authors believe that the application of PIV to large-scale problems in the environment is an exciting and potentially very efficient method of obtaining one-, two-, and three-dimensional flow field information for scientific and engineering analysis. Possible applications include measurement of flows in streams, tidal estuaries, floodplains, and other waterways. PIV techniques can be very effective in situations where it is unsafe or difficult to obtain field measurements with conventional methods, e.g. in shallow water bodies, near bridges and river training structures, power houses, and spillways. Unlike ADV measurements, which require a substantial investment in equipment and field crews, the PIV-based system uses inexpensive video equipment to make digital images of the flow surface. Furthermore, measurements can be made quickly and with little site preparation. This means that one can respond quickly in urgent situations, such as the measurement of floodplain flows during a flash flood, or make frequent measurements under varying flow conditions.

In recent years, there have been some notable applications of PIV to environmental flow problems. Aya et al. (1995) and Fujita et al. (1997) have applied PIV techniques to measure flows during a flood on the Yodo River (Japan). Mean velocity distributions at several cross-sections, obtained by analyzing sixty consecutive images, measured the changes in velocity distribution between the main channel and floodplain. The authors have recently exploited the PIV system in large-scale physical model studies. The technique was used in model experiments to map ice and water velocities in a study of ice jamming in rivers (Ettema et al. 1997) and to

quantify aeration processes downstream of a spillway. These examples show that PIV techniques have applications beyond fundamental fluids research, and that PIV has enormous potential for measurement of flows in the natural environment.

PIV Field Measurement Techniques

The requirements for PIV measurement are a suitable series of images, flow tracers, a method of digitizing the images, a computer, and PIV software. The images may be comprised of radar images, a series of photographs, or direct video images. In the latter case, the images may be digitized with a personal computer equipped with a frame-grabber. The images must then be processed using a motion estimation technique. An important component for field measurement is the registration of image coordinates in physical space for low oblique camera viewing angles. The details of the proposed PIV field measurement system are discussed in the following subsections.

PIV Field Measurement System

A schematic diagram of the large-scale PIV measurement system is depicted in Fig. 1. It should be noted that a nearby bridge, building, lifting truck, or any other lifting mechanism that the video camera can be mounted on can be used to lift the camera high enough to record images. The PIV components include a high quality video camera, a computer-controllable video recorder, flow tracers, and PIV software to obtain the velocity field from the recorded images. The flow tracers must be large enough so that they map to at least one pixel, but they must be small and light enough to accurately trace the free surface movements. In order to obtain accurate velocity vectors, a large number of tracers are required. In addition, the traces must form sufficient contrast with the water surface. In a laboratory experiment, one can control most of these factors, but in a field experiment, this may present a challenge.

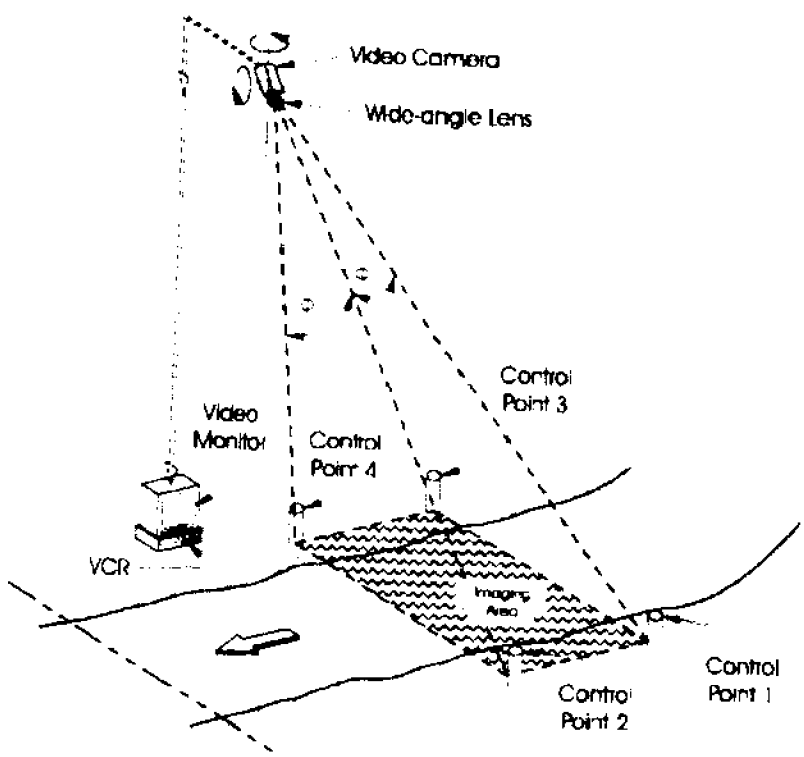

Fig. 1. Proposed field Particle Image Velocimetry setup.

While PIV is conceptually simple, there are many small details that must be resolved in order to get reliable velocity measurements. The image-processing software was developed over many years by a PIV expert, Dr. Ichiro Fujita from Gifu University in Japan, and modified for large-scale applications at Iowa Institute of Hydraulic Research (IIHR). The code is solid and very well-tested in several prior applications (Ettema et al. 1997; Fujita et al. 1997; Aya et al. 1995).

Preliminary testing of components of the PIV system has been carried out along the Iowa River, IA using a Panasonic Palmcorder PV 1Q405 video camera. A PIV analysis was done in video image coordinates using naturally occurring foam, boil and ripple structures, as tracers. A movie of the video images and the estimated velocities is available at the IIHR web site (http://www.iihr.uiowa.edu/rivpiv).

PIV Technique

There are two major approaches to motion estimation from a sequence of images: (a) differential techniques, and (b) correspondence techniques. In differential methods, the image sequence is modeled with a three-dimensional function f (x, y, t). This enables derivation of a set of linear relationships for the spatio-temporal derivatives of the image. In correspondence methods, one searches for a direct correspondence (correlation) between a group of pixels between frames. There are many variants of correspondence methods, called matched-filter or template methods. All of these approaches have been used in many earth science applications (Leese et al. 1971; Ninnis et al. 1986; Emery et al. 1986; Collins et al. 1988). It is not immediately clear which class of methods produces better results. While differential methods are known to be more computationally efficient, Abidi and Gonzalez (1987) have demonstrated that correspondence methods are superior in terms of accuracy.

The PIV method described here is a variation on the classical PIV correspondence method. It is conceptually a little simpler and more accurate than the classical correspondence method. Figure 2 shows two consecutive flow fields overlaid with an

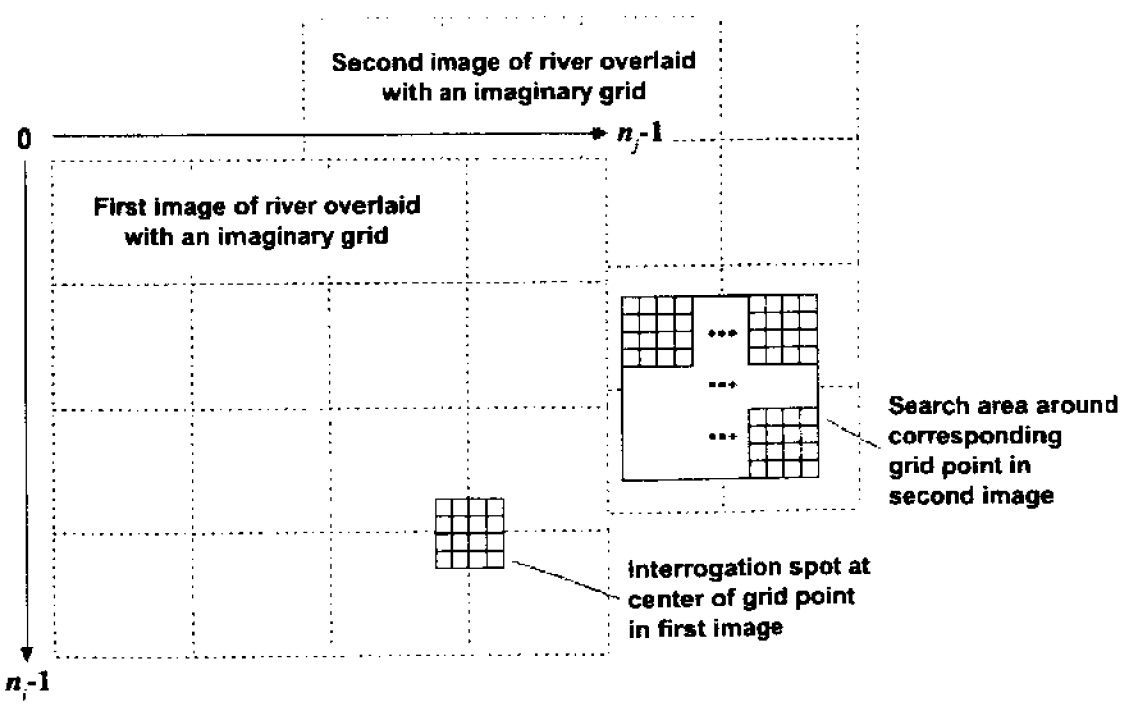

Fig. 2. Two consecutive flow fields overlaid with an imaginary grid.

imaginary grid. An interrogation spot is selected around the center of a grid point in the first field. Next, the most likely location of this group of pixels in the second field is identified as follows. An enclosing search area is selected around the corresponding grid point in the second field. The interrogation spot is placed in the upper left-hand corner of the search area, and the linear correlation coefficient R between the two sets of pixels is computed. The interrogation spot is then moved right one position and R is computed again. This procedure is repeated for all the possible locations of the interrogation spot in the search area. The location where the correlation coefficient takes on its highest value is taken as the most likely location of the pixels from the interrogation spot in the search area. With the displacement and time difference between fields now known, one can compute an estimated velocity at the specific grid point. The whole process is repeated at every grid point and the result is a field of velocity vectors. To analyze 300 images on a 133 MHz Pentium PC requires about 24 hours. With proper seeding, fewer images are required, and a faster computer should drop the processing time to a few hours.

The size of the grid determines the pitch of the velocity field as well as the overall computation time. It is not obvious what the best size for the interrogation spot or the search area is. If the interrogation spot is too small, many sets of pixels in the search area may give high correlation, leading to erroneous vectors. The same is true if the search area is too large. On the other hand, if the interrogation spot is too large, the differences between the computed correlation coefficients become small, also resulting in erroneous vectors.

Image Registration

In order to estimate velocities from an image sequence, pixel displacements/locations must be related to physical displacements/locations. In traditional PIV-based measurements, a grid of known dimensions is placed in the flow facility. Finding a particular pixel's location in physical space is done as follows. First, find the four closest pixels that (a) surround the selected pixel and (b) that correspond to a known grid point location in the flow facility. Then perform a 2-dimensional (bilinear) interpolation to estimate the location of the pixel in physical space. In the laboratory, this important step is straightforward because one can normally ensure that grid points correspond exactly or very closely to integer pixel locations (e.g. by changing the grid size, zooming, etc). Also, if higher accuracy is required, higher-order interpolation functions can be used, or the grid size can be decreased. Furthermore, since the experiments are almost always set up so that the camera is perpendicular to the imaged area, the analysis is inherently two-dimensional. The experiments typically require a small field of view, typically less than five to seven degrees such that the camera lens introduces little spatial distortion. Oftentimes, so-called telecentric lenses, which preserve spatial relationships, are used. In the system used herein, imaging is performed at a low oblique angle. Consequently, the field of view is much larger so that lens distortion may not be negligible. The basic approach of overlaying the imaged area with a grid of known dimensions is no longer feasible, and practically only few selected points on the perimeter of the imaged area are available (e.g., trees, power line poles, building corners, etc. for rectification purposes). The location of these control points will often be dictated by what is available out in the field. A general approach for image registration is the use of a three-dimensional conformal coordinate transformation (Wolf 1983). Fujita et al. (1997) used a simpler, more direct approach, based on an eight-parameter projective transformation to estimate the physical location of points from the image. This fractional linear transformation can empirically model the distortion effects for oblique images (Mikhail and Ackerman 1976). However, the approach assumes a horizontal water surface, and requires control points on the horizontal surface for parameter estimation.

Flow Estimation

PIV measurements may be used to estimate flow movements in shallow water bodies, stream and floodplain discharges, and other variables of interest, including velocity distributions and bed shear stresses. In traditional PIV-based measurements, flow velocities are measured at specific

depths by seeding the flow with particles of carefully selected densities. Velocities at various depths are then integrated to obtain vertical velocity distributions and total discharge. In field applications, the only measurements that can be made are free-surface velocities. To effectively utilize this information to estimate one-, two-, and three-dimensional flow components, a numerical model is needed to assimilate these data under imposed physical constraints.

Kinematic flow analysis is a very robust and computationally inexpensive numerical tool that can be used to analyze complex flow patterns using a discrete set of measurements. With this approach, the discrete PIV field measurements are used to generate a full three-dimensional flow field based solely on kinematic (continuity) principles. The numerical model generates mass-conserving flow fields using a discrete set of measurements, specified boundary conditions, and bathymetry data. The kinematic model initially interpolates the discrete measurements over a Cartesian uniform grid that covers the entire flow domain under consideration. Afterwards, using this interpolated flow field along with the boundary conditions, the continuity equation is solved iteratively until a steady state mass-conserving solution is obtained. The kinematic model serves as a physically-based interpolation scheme for assimilating surface velocities measurements in flow-field estimation.

The main advantage of the kinematic flow analysis is the significant reduction in the scope and detail of field measurements needed to obtain physically reasonable and accurate flow-field estimates. Thus, the utility of the free surface velocity measurements is enhanced because additional measurements of vertical velocity profiles are not required for flow estimation. Reducing the amount of data needed for accurate flow estimation also produces a considerable savings in the field time required for flow measurement, which is critical when flow conditions are rapidly changing (e.g., during floods). The kinematic model has been validated for several case studies with complex flow patterns (Meselhe et al. 1997).

The PIV method has some limitations. This method can not accurately measure flows that are primarily driven by wind, highly stratified flows, flows with strong secondary motions, and flows through heavy vegetation.

Case Study

As a first step toward validating the flow estimation methodology presented earlier, velocity profiles extracted from a three-dimensional flow field generated by the kinematic model (and guided by the PIV measurements) are compared to velocity point measurements made in the field. The selected field site at Clear Creek near Oxford, IA (Fig. 3) has a drainage area of 151.3 km^2 (58.4 mi^2). The USGS Gage 05454220USGS, with permanent discharge recording, is located just downstream of the reach where the measurements were taken. As shown in Fig. 4, five cross sections were surveyed. Contours of the bathymetry are also shown in Fig. 4. Point measurements of velocity were collected at section 4 using conventional current meters. The flow discharge is 0.192 cms, Froude number is 0.11, and cross-sectional average velocity and average depth (at the upstream section) are 0.16 m s^{-1} and 0.224 m, respectively.

The computational grid used had a Δx and Δy of 0.1542 m, and Δz of 0.03048 m. A fully developed flow (for a corresponding Reynolds number = 3.2×10^4) was assumed at both the upstream and downstream sections. The PIV velocity field at the free surface was used as guiding observations. PIV measure ments were taken from one camera location. Flow seeding was made using tree leaves released as uniform as possible over the free surface (Fig.3). It should be noted that the leaves were fairly flat over the water surface and were not affected by wind.

A cross-sectional velocity profile of the depth-averaged velocity at section 4 is shown in Fig. 5. For this early developmental stage, the numerical solution guided by the PIV measurements, the comparison with the field point measurements is satisfactory.

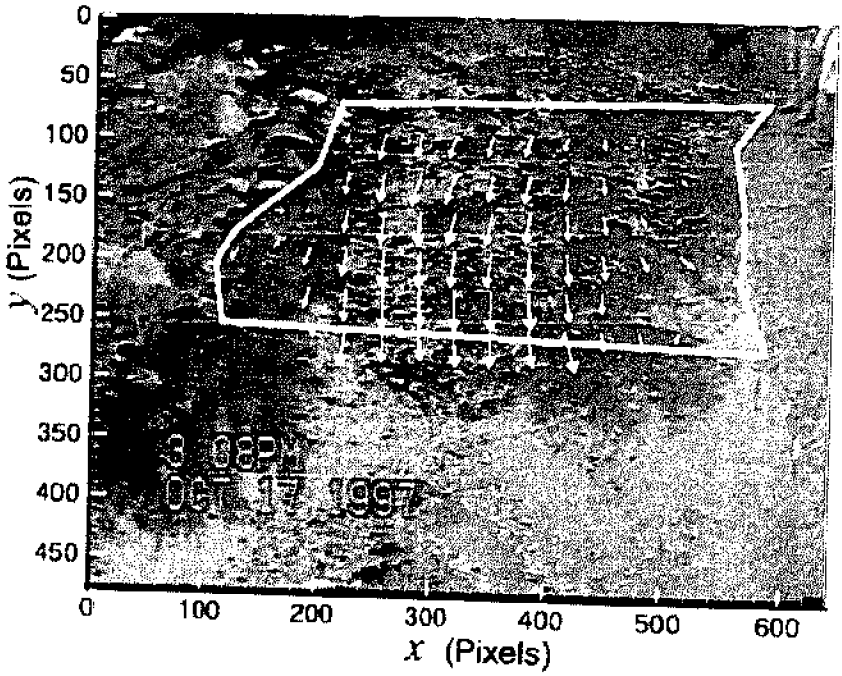

Fig. 3. Image of a section on Clear Creek overlaid with estimates of free surface velocities. Vectors show the average pixel displacement (in pixels/s). The average is based on 60 images taken at 1 second intervals.

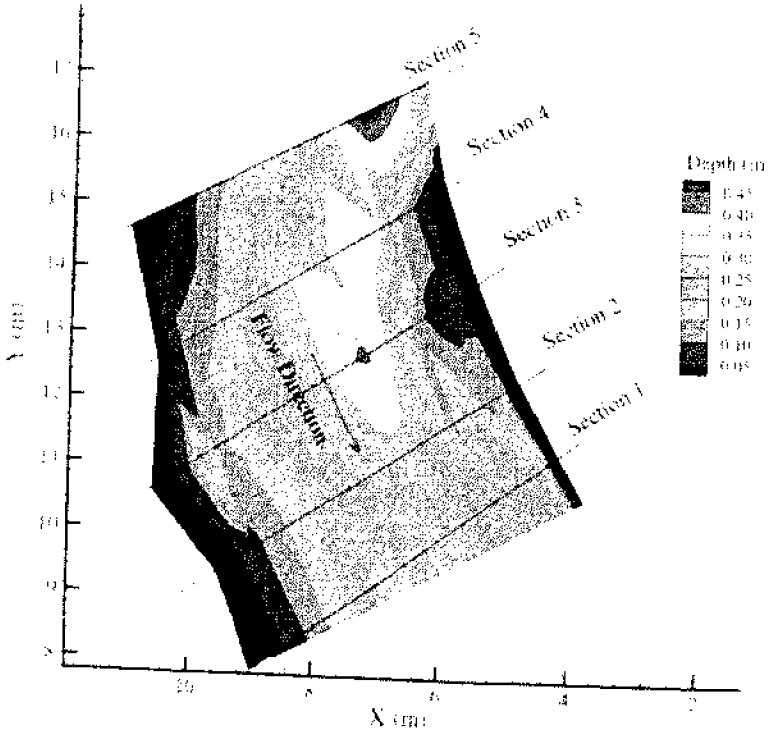

Fig. 4. Bathymetry data for the surveyed reach on Clear Creek.

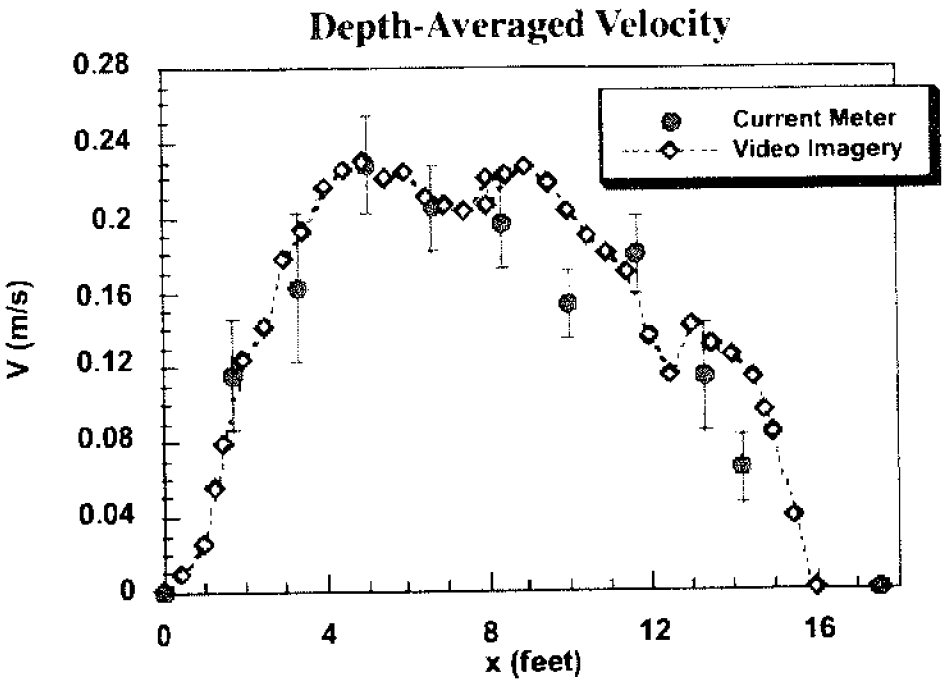

Fig. 5. Depth-averaged velocity estimates based on current meter measurements (points) and the kinematic flow analysis (line). Uncertainties in the current meter measurements (two standard deviations) are indicated by the bars.

Discussion

This case study demonstrates the effectiveness of using PIV supplemented by numerical modeling for flow estimation and analysis. It can be concluded from this preliminary field testing of the methodology, that setting up the PIV system to obtain free-surface velocities is simple and requires little site preparation.

The method is suitable for wide bodies of water, as it captures larger areas per frame compared to point measurements, which would be both expensive and time consuming in this type of application. Using the kinematic model to produce three-dimensional flow fields from the PIV surface measurements allows for performing more rigorous analysis and for computing flow parameters such as flow acceleration and deceleration, and bed shear stresses. However, deriving the whole three-dimensional flow field from only the surface velocities is far from trivial for certain applications. It might not be possible to perform in applications with sub-surface reverse flows or with surface vegetation.

To avoid introducing any environmentally harmful material, flow seeding can be achieved with biodegradable tracers such as rice- or corn cakes, drifting debris, ripple patterns, boil vortices, and patches of foam, wood chips, leaves, or even crushed ice. Testing this technology for a variety of practical applications is underway. These tests will assess the capabilities and limitations of the coupled PIV and numerical modeling technique.

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The Effects of Sediment Addition on Salt Marsh Vegetation and Soil Physico-Chemistry

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ABSTRACT: An imbalance between the rate of marsh elevation increase and the rate of water level rise is one of the major causes of wetland deterioration in coastal Louisiana. This imbalance may be offset by sediment addition to the marsh. We assessed the vegetative and physico-chemical responses of a deteriorating salt marsh to different intensities of sediment addition as a means of mitigating this sediment aggradation deficit. Sediment was hydraulically dredged from the Gulf of Mexico and discharged into a *Spartina alterniflora*-dominated salt marsh, resulting in a gradient in depth of added sediment from trace amounts to as much as 60 cm above natural marsh elevations over a 43 ha area. Replicate samples for vegetative and soil responses were randomly collected within 5 relatively distinct areas, based on depth of sediment burial, in 1993. The addition of sediment created significantly different marsh elevations. The greatest sediment accumulations initially caused aboveground mortality of the vegetation. However, by Fall 1993, almost two years after dredging, both plant cover and height responded positively to sediment addition. Sediment addition increased soil bulk density as a result of the mineral matter added to the marsh surface. The higher mineral matter increased soil fertility and marsh elevation and thereby reduced nutrient, flooding, and interstitial sulfide stresses. Thus, sediment addition generated a more favorable plant growth environment by increasing elevation and reducing potentially toxic sulfides as well as increasing soil fertility.

Introduction

Approximately 80% of all coastal wetland losses in the U. S. occur in Louisiana (Boesch et al. 1994) with an estimated 3,950 km^2 of coastal lands in Louisiana having degraded to open water since the 1930's (Dunbar et al. 1992). Currently, about 65 km^2 of coastal lands are lost in Louisiana each year (Dunbar et al. 1992). The vast majority of wetlands lost in Louisiana have been transformed into open water due to natural processes, such as subsidence and sea level rise, as well as human activities, such as canal dredging and leveeing (Boesch et al. 1994; Turner et al. 1994).

There has been a growing realization that salt marsh restoration and protection will be necessary to at least slow the rate of land loss in Louisiana, because otherwise the future of Louisiana's coastal lands appears bleak. Many management activities have been proposed to help slow the loss of wetlands and to restore them in areas that have already degraded (Mendelssohn et al. 1983; Boesch et al. 1984; Boesch et al. 1994). In addition, regulations have been tightened to restrict human activities which cause wetland loss (Mendelssohn et al. 1983; Boesch et al. 1994). Sediment additions, either by natural or artificial means, represent one of the most promising and long term of the proposed actions to restore and protect these wetlands.

One of the major contributors to salt marsh loss in Louisiana's deltaic plain is a sediment deficit caused by leveeing the Mississippi River (Salinas et al. 1986; Penland et al. 1990). The salt marsh depends on continued sediment additions from the river via overbank flooding and the accumulation of organic matter produced by wetland plants in order to maintain itself above sea level in the rapidly subsiding environment of the deltaic plain (Nyman et al. 1990; DeLaune et al. 1992). Organic matter

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accumulation and sediment addition interact to prevent salt marsh submergence. Reduced sediment inputs cause a decline in the accumulation of mineral matter and the supply of plant nutrients, which in turn reduces the rate of salt marsh accretion through decreased plant growth, organic matter production, and sediment trapping via the network of plant roots (DeLaune et al. 1990; Wilsey et al. 1992). With reduced rates of salt marsh vertical accretion, the salt marsh surface is submerged more frequently and for a longer period of time (Baumann and DeLaune 1981). Increased submergence in the salt marsh can create stressful conditions which kill or reduce the growth of resident plants through processes such as reduced substrate aeration (Mendelssohn et al. 1981) or the accumulation of hydrogen sulfide to toxic levels in the substrate (Havill et al. 1985; Bradley and Dunn 1989; Koch and Mendelssohn 1989). Thus, a reduction in sedimentation associated with reduced organic matter accumulation can cause the eventual conversion of salt marsh to open water.

Spreading dredged material out on the marsh surface is one of the methods that has been proposed for supplying mineral matter to sediment-starved marshes (DeLaune et al. 1990; Wilsey et al. 1992). Sediment dredged from river or canal bottoms is usually dispersed into open water or is used as fill (Kennish 1992), but it can also be used to supply sediment to coastal wetlands (Wilsey et al. 1992). In addition, dredged material is being used in marshes to mitigate for the impacts of canal dredging.

A major obstacle to the successful use of dredged material for sediment supply enhancement in salt marshes is our limited knowledge of how different depths of added sediment affect vegetative growth and wetland soil physico-chemistry. Insufficient sediment addition may have no beneficial effect, while excessive sediment addition may modify growing conditions enough that the site converts to upland habitat. Furthermore, it is difficult to predict the amount of sediment necessary, after compaction and subsidence, to create the elevational conditions required for a particular marsh type.

In January 1992, as part of a pipeline canal mitigation project, sediment was hydraulically dredged from the Gulf of Mexico to fill a pipeline canal. Much of this fill material accidentally overflowed into an adjacent deteriorating salt marsh located 13 km southwest of the city of Venice, Louisiana, near the mouth of the Mississippi River. A gradient in sediment depth was created with levels ranging from trace amounts to as much as 60 cm of sediment above the natural marsh surface over a 43 ha area. The creation of this sediment addition gradient provided a unique opportunity to study the effects of different levels of sediment deposition on vegetative condition and soil physico-chemistry in a deteriorating salt marsh. Although some research has been done on spreading dredged material in the marsh (e.g., Cahoon and Cowan 1988; Ford et al. 1999), to our knowledge this is the first evaluation of the effects of large-scale differential sediment addition on salt marsh plants and soils. If we are to maintain intertidal wetlands as sea level rises, we must learn to understand the mechanisms controlling the successful use of sediment enhancement in salt marshes.

Materials and Methods

The study site, an intertidal saline marsh dominated by *Spartina alterniflora* Loisel, was divided into 5 treatment areas based on an elevation survey taken in 1992: (1) Reference: marsh which received no hydraulically dredged sediment addition, (2) SAR (Sediment Affected Region) I: marsh receiving trace amounts of sediment, but not quantifiable with standard elevation survey techniques, (3) SAR-II: marsh with measurable sediment burial not greater than 15 cm, (4) SAR-III: marsh with 15 to 30 cm of sediment burial, and (5) SAR-IV: marsh with more than 30 cm of sediment burial. These elevation data should be considered a conservative estimate of the actual amount of sediment deposited in these study sites given the potential for subsidence of underlying marsh surfaces and sediment dewatering. Standard survey techniques were also used to generate marsh elevations for 1993, and these survey results are presented here since they directly coincide in time with the vegetation and soil data collected.

Five permanent sampling sites were randomly located within each of the 5 sediment addition areas

for a total of 25 permanent sampling plots. Each sampling plot was a 2 X 2 m area. Data were collected on 12 November 1993 to represent the end of the growing season response. Percent cover of live and dead plant material as well as average plant height were visually estimated in each permanent plot.

In addition, various soil physico-chemical characteristics were measured. Most of these data were also collected in November 1993. However, oxidation-reduction potential (redox; Eh), sulfide and bulk density data were measured on 24 August 1994. Soil Eh was measured within the permanent plots using 3 bright platinum electrodes and a calomel reference electrode with three subsamples taken at the soil surface. An average of the three subsamples was used for statistical analysis. Soil cores for measuring bulk density, salinity and sulfide were collected from the area surrounding the permanent cover plots. The bulk density core (5 cm diameter X 9.5 cm tall) was dried at 65°C for at least 24 hours prior to weighing. The core collected to measure salinity and interstitial sulfide (5 cm diameter X 25-30 cm tall) was immediately sealed in a 500 ml centrifuge bottle, purged with nitrogen gas through an air-tight septum for 2 minutes, and then placed on ice. In the laboratory, these soil samples were centrifuged (5000 rpm at 5°C for 12 minutes) under nitrogen, and interstitial water was immediately removed from them. One subsample of this water was used to measure salinity using an ATAGO S/Mill refractometer. The other subsample was placed in antioxidant buffer and analyzed for total soluble sulfide concentration (Sulfide Electrode, Lazar Research Laboratories, Los Angeles, CA). Another soil sample was collected for extractable NH_4-N, NO_3-N, P, Fe, and Mn. Soil extraction of these elements followed standard techniques (NH_4-N and NO_3-N extraction following Bremner and Keeney [1966]; P extraction following Byrnside and Sturgis [1958]; Fe and Mn extractions following Baker and Amacher [1982]). After extraction NH_4-N and NO_3-N samples were filtered through a 0.45 mm syringe filter and concentrations were measured using the Colorimetric, Automated Phenate Method (US Environmental Protection Agency 1979). The remaining elemental concentrations were measured using a Fisher inductively coupled argon plasma emission spectrometer (ICP). Data from soil extractions are presented on a soil-volume basis since this method is preferable in situations where there is a wide variation in bulk density between study sites (DeLaune et al. 1979).

Data were analyzed using ANOVA. All significant main effects were further investigated using Fisher's protected LSD. In order to improve normality and homogeneity of variance, certain variables had to be transformed. Log transformations were done on sulfide, extractable ammonium-N, iron, and manganese. Arcsine square root transformations were done to the percent cover data. General trends within the data were revealed by plotting the mean and standard errors (SE) of the raw data for each treatment.

Results

Vegetative Parameters

Plants in the sediment affected areas showed a positive response to increasing depths of added sediment (Fig. 1). With increasing levels of sediment addition, there was a significant increase in total cover ($p \leq 0.0001$) with levels in the Reference area only half those for SAR-IV sites (Fig. 1). Similarly, plant height was 30-50% greater with increasing sediment addition (e.g., 101.2±6.0 cm in Reference vs. 175.0±11.9 cm in SAR-IV). Sites that received the most sediment (SAR-III and IV) had plants that were significantly taller ($p \leq 0.001$) than those in sites receiving little or no sediment (Reference, SAR-I and II).

Soil Physico-chemical Status

Most physical parameters also showed some form of response to varying depths of sediment addition. With progressively more sediment added, redox potential rose at the soil surface ($p=0.008$) (Reference: -73.2±45.5; SAR-I: -20.8±25.5; SAR-II: 5.4±29.1; SAR-III: 66.6±8.0; and SAR-IV: 75.6±24.3 mV). SAR-III and -IV sites had higher redox potentials ($p \leq 0.05$) at the surface than areas with little or no sediment addition (Reference and SAR-I).

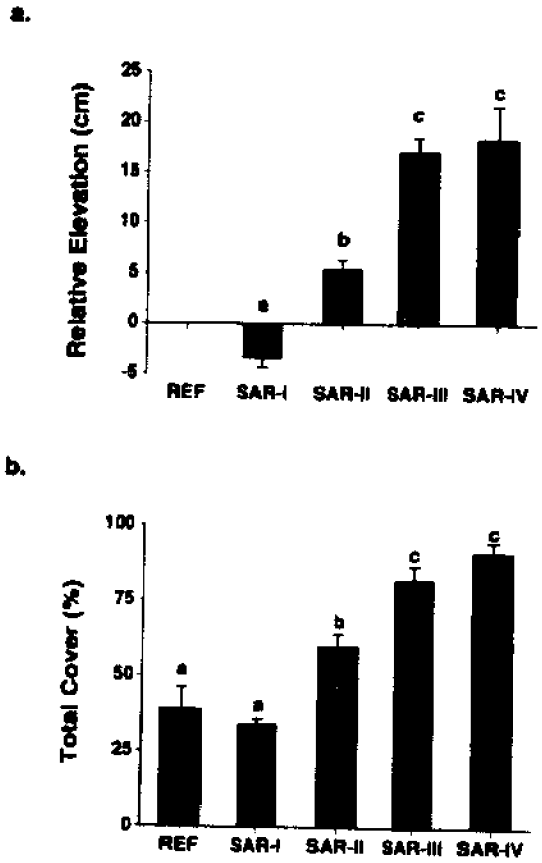

Fig. 1. The effects in 1993 of sediment addition on (a.) relative elevation and (b.) percent vegetative cover in the five treatment areas. All elevation measurements are relative to a reference marsh with an elevation equal to zero. Data are means and standard errors (n=5). The same letters indicate no statistical difference between treatment means (Fishers LSD $p<0.05$).

Bulk density showed a steady and pronounced increase from the Reference to the SAR-IV sites ($p<0.0001$). SAR-IV bulk densities were roughly four times higher than in the Reference and SAR-I areas (Reference: 0.25±0.03; SAR-I: 0.26±0.04; SAR-II: 0.49±0.08; SAR-III: 0.61±0.06; and SAR-IV: 0.97±0.15 g cc^{-1}). Only the Reference and SAR-I sites appeared to have similar bulk densities. Extractable iron and manganese concentrations significantly increased ($p=0.0012$ and $p\leq0.0001$, respectively) with increased sediment deposition (Table 1).

Interstitial sulfide concentrations fell sharply with increased sediment addition ($p=0.004$) (Table 1). Sulfide concentrations in the Reference and SAR-I sites were at least 15X higher than those for SAR-II and roughly two orders of magnitude higher than those in the SAR-III and -IV areas. Average sulfide concentrations for the Reference and SAR-I were not to levels known to be lethal to salt marsh plants (≥2.0 mM), but were approaching concentrations high enough to impair plant growth (≥1.0 mM).

Extractable ammonium-N concentrations decreased as added sediment depth increased, but the differences were not significant ($p=0.4$). Extractable NH_4-N concentrations in the SAR-III, and -IV sites were as much as 2.5X lower than those in the Reference and SAR-I and -II areas. However, extractable NO_3-N concentrations were significantly higher ($p=0.0075$) in SAR-IV. In addition, extractable phosphorus concentrations significantly rose ($p=0.04$) from the Reference to SAR-IV sites (Table 1).

Interstitial soil salinity was higher with more added sediment ($p<0.0001$). Individual comparisons indicate that areas receiving the most sediment (SAR-III and SAR-IV) had significantly higher ($p\leq0.0003$) salinities than those receiving the least (Reference and SAR-I), but in terms of actual differences in concentration, the greatest of these was only about 4 g l^{-1} (e.g., 7.9±0.5 g l^{-1} vs. 4.2±0.1 g l^{-1} for SAR-IV and Reference areas, respectively)

Discussion

Both vegetative and physical parameters responded to sediment additions. Plant height and cover were greater with increasing sediment deposition. The change in marsh elevation associated with sediment addition may help explain this pattern. Marsh surface elevation substantially rose as more sediment was added, and this increase in elevation positively influenced a number of factors affecting plant growth.

Flooding depth was reduced and soil aeration was improved with increasing sediment additions. Redox potential increased with sediment addition indicating that soils were more oxidized in the higher elevation sites than in areas which received less sediment. With a reduction in flooding, the water

Table 1. Extractable element concentrations (mmol cc^{-1}) for the five sediment affected areas for 1993. Interstitial sulfide concentrations (mM) were measured in 1994. Data are means with standard errors in parentheses.[a]

Treatment	NH_4-N	NO_3-N	P	Fe	Mn	Sulfide
Reference	0.26[a] (0.05)	0.002[a] (0.0003)	39[a] (6)	69[a] (24)	3[a] (1)	0.561[a] (0.184)
SAR - I	0.17[a] (0.04)	0.002[a] (0.0006)	47[a] (11)	108[a] (35)	8[a] (3)	0.384[ab] (0.162)
SAR - II	0.17[a] (0.07)	0.002[a] (0.0003)	79[ab] (13)	357[b] (37)	30[b] (3)	0.026[c] (0.020)
SAR - III	0.12[a] (0.08)	0.003[a] (0.0002)	55[a] (9)	287[b] (38)	28[b] (4)	0.003[c] (0.001)
SAR - IV	0.10[a] (0.04)	0.005[b] (0.0010)	102[b] (27)	286[b] (83)	32[b] (6)	0.005[bc] (0.002)

[a] The same letters within a column indicate no statistical difference between means (Fishers Protected LSD $p<0.05$).

table would more often fall below the soil surface allowing for soil drainage and direct exchange of gases between the air and the soil deep into the soil profile. Also, reduced flooding helped aerate the rhizosphere since more aboveground parts of the plant that contain aerenchyma would be exposed to the air for a longer period of time. Plant growth can be reduced in hypoxic soils because of an oxygen deficiency in the rooting zone which forces plants to rely more heavily on anaerobic metabolism for their energy production (Mendelssohn et al. 1981). However, since soils were less flooded and more aerated at the higher elevation sites, the roots of plants in these areas could respire aerobically and reduce their reliance on alcoholic fermentation (Mendelssohn et al. 1981; Mendelssohn and McKee 1988) resulting in more growth.

In addition, the improvement in soil aeration also reduced the concentration of phytotoxins commonly found in more reduced salt marsh soils. Free sulfide is the major plant toxin typically produced in reduced salt marsh soils (Mendelssohn and McKee 1988). In highly reduced salt marsh soils such as those in Reference and SAR-I sites, obligate anaerobic bacteria use sulfate from sea water as their terminal electron acceptor during respiration and convert it to sulfide. However, in soils like SAR-III and -IV which were more oxidized, aerobic and facultative anaerobic bacteria that do not produce sulfide during respiration are usually most active. Results show that sulfide levels in the highest elevation sites (SAR-III and -IV) were far below those that negatively affect plants, but in the lowest elevation sites (Reference and SAR-I) sulfide concentrations did at least approach those which reduce the growth of *S. alterniflora* (~1 mM) (Koch and Mendelssohn 1989). In addition, since sulfide also inhibits plant uptake of nitrogen (NH_4-N) (Koch et al. 1990), which is the primary limiting nutrient to salt marsh plant productivity (Valiela and Teal 1974), production may have been further reduced in the lowest sites by its presence.

Physical characteristics of the sediment may also help explain the increase in production as the amount of added sediment rose. Bulk densities indicate that mineral matter content was much higher in areas receiving the most sediment. Wetland soils with a higher mineral content have a greater ability to take up and sequester nutrients (Mitsch and Gosselink 1993), and they also have been shown to provide more nutrients on a per volume basis when compared to organic salt marsh soils (DeLaune et al. 1979). Sites receiving the most sediment addition also had greater extractable Fe and Mn concentrations. These elements are important in their ability to precipitate sulfides and thereby reduce toxic soluble sulfide concentrations (Gambrell and Patrick 1978).

The effect sediment additions had on soil nutrient pools may also help explain plant responses. Although NO_4-N concentration was lower with greater sediment addition, the high plant biomass at sites receiving the most sediment suggests that the low interstitial N was a result of plant uptake. In fact, extractable NO_4-N in the original dredge material was 40X higher than that for the soil material collected within the study area in 1993 (date not shown). Thus, the sediment initially introduced into this salt marsh was of high N fertility . This is likely the key reason for the much higher cover and plant heights seen in this area, since nitrogen is the major nutrient limiting salt marsh plant growth (Valiela and Teal 1974). In addition, the increase in extractable P concentrations with sediment additions up to SAR-IV also likely had a positive effect.

Conclusion

Sediment additions appear to have successfully rehabilitated this deteriorating salt marsh. With sediment additions salt marsh plant growth improved due to increased soil aeration, nutrient and mineral matter content. Areas receiving intermediate and high amounts of sediment showed increased plant cover. Furthermore, the highest SAR's are likely to be the most long-lived because of the effects of subsidence and sea level rise. Thus, sediment additions could play a positive role in the management of sediment starved marshes, although successful enhancement may be mediated by a number of other considerations.

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WATER QUALITY, EUTROPHICATION, AND SALINITY CHANGES

The Use of Diatom Remains as a Proxy of Historical Salinity Changes in Airplane Lake, Louisiana

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ABSTRACT: We analyzed dated sediment core samples from Airplane Lake, a small, shallow lake west of Barataria Bay to determine if diatom remains preserved a salinity signal in this highly variable environment. We developed a diatom-based index as a proxy for salinity over the 92 years represented by the samples. This diatom-based salinity index indicated an overall increasing trend in salinity at Airplane Lake of approximately 1 ppt per decade. A significant decreasing trend in salinity, however, was evident from 1973 to 1992 on the order of 5 ppt per decade, possibly in response to the documented increased riverine discharges and precipitation in the region. This preliminary study demonstrates that diatom remains in marsh sediment cores may provide a useful proxy for studies of salinity changes in coastal Louisiana.

INTRODUCTION

Coastal Louisiana is experiencing high rates of land loss (0.86% per year between 1955 and 1978; Baumann and Turner 1990). Wetland loss has been linked hypothetically to increasing coastal salinity. Salt water can stress and kill marsh plants, especially fresh water plants, which in turn can result in land loss due to the reduced ability of the marsh plants to hold organic matter in place (McKee and Mendelssohn 1989). The primary evidence supporting a coast-wide increase in salinity includes the movement and shifts of vegetative zones of marsh plants over the past several decades (Chabreck and Linscombe 1982; Fuller et al. 1995) and a northward extension of oyster leases in Barataria Bay (Van Sickle et al. 1976). Conversely, studies of long-term salinity data demonstrate that salinities have increased in some areas but decreased or remained unchanged in others (Wiseman et al. 1990a; Swenson and Swarzenski 1995). There is no strong evidence for a coast-wide increase in salinity. Rather, salinity increases appear to be localized events probably due to the dredging of navigation canals (Wiseman et al. 1990a). While reports of documented salinity changes exist, the available salinity data are too sparse temporally and spatially to properly test for coast-wide salinity increases.

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One approach that can provide historical information on changing marsh salinity is the analysis of diatom remains preserved in the sediment record. The morphology of a diatom valve is species-specific and generally preserves well in the sediment. Many diatom species have specific salinity preferences, which can result in changes in a diatom community if and when salinity changes. Thus, diatom valves are useful indicators of past salinity. Identification and enumeration of diatom valves coupled with information on optimal salinity conditions for growth can be used to "reconstruct" the water conditions at the time the diatoms were living (Battarbee 1986). This study introduces some of the concepts and formulations as applied to one salinity reconstruction that was part of a larger study presented in Parsons et al. (1999).

MATERIALS AND METHODS

Core collection

We collected two sediment cores in August, 1993, from Airplane Lake, a 19 hectare water body west of Barataria Bay (Fig. 1) utilizing 3-in diameter, 1.3-m long plastic tubes. Because of the nature of the core collection process (using ~1-m tubes to collect cores underwater in a ~1-m deep salt marsh pond), it was impossible to accurately measure core compaction during core collection. We measured core compaction during core extrusion in the laboratory. Cores were stored at 4°C, and the longest sediment core was split the following day on a custom-made precision core extruder (a threaded extruder allowing precise measurements for slicing). Visual inspections and length measurements of the core were recorded prior to and during core extrusion. We split the core into 1-cm increments which were then homogenized and subdivided for subsequent analyses.

Core dating

Selected core increments were dated by ^{137}Cs (Milan et al. 1995) and ^{210}Pb (Cutshall et al. 1983) using a Princeton Gamma-Tech 60-mm diameter intrinsic germanium "N" type coaxial detector (40% efficiency) interfaced to an EG&G Ortec 92X spectrum master integrated gamma-spectroscopy system. Samples for ^{210}Pb were held for two weeks before analysis to allow for equilibrium between atmospheric ^{222}Rn and ^{210}Pb. Samples for ^{137}Cs were counted for at least 4 h to yield a counting error of 10% in the vicinity of the 1963/4 peak, corresponding to the peak ^{137}Cs fallout for the southeastern United States (Pennington et al. 1973). Samples for ^{210}Pb were counted briefly with a source of ^{210}Pb to measure the sample self-absorption potential. The samples were then recounted for 24 h to obtain a significant amount of net counts above background. Additional counts were conducted for ^{214}Pb activity so that supported ^{210}Pb could be

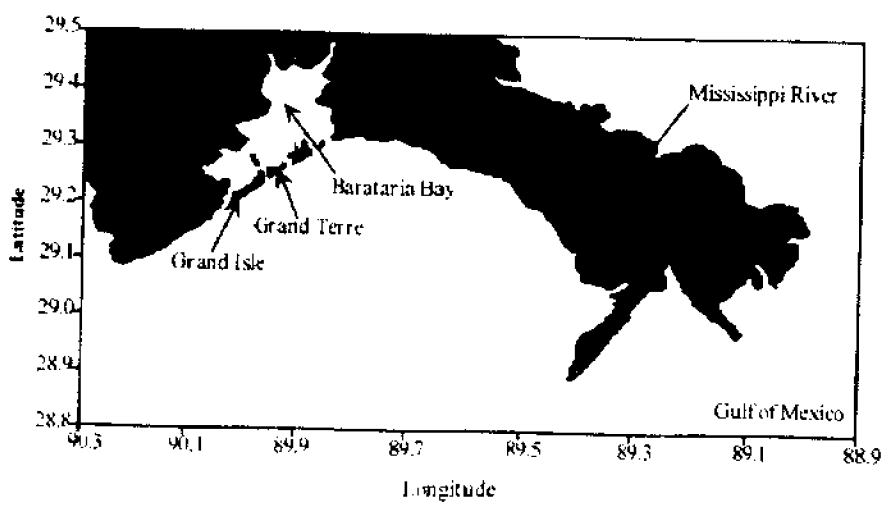

Fig. 1. A map of coastal Louisiana showing the location of Airplane Lake and important geographic features.

determined. Supported ^{210}Pb was subtracted from the total to obtain excess ^{210}Pb, from which sedimentation rates were determined. Results for ^{137}Cs and ^{210}Pb are reported as pCi g^{-1} (dry weight sediment).

The ^{137}Cs-based sedimentation rate was determined by dividing the difference between the date the core was collected and 1963/4, by the depth where the peak in ^{137}Cs activity occurred. A sedimentation rate was determined from the excess ^{210}Pb utilizing the constant rate of supply (CRS) model, assuming that there is negligible migration of ^{210}Pb in the sediment and that the supply of excess ^{210}Pb is constant over time. The sedimentation rate, therefore, was determined through the inverse of the slope of the least-squares regression line that was fitted to the ^{210}Pb profile. The ^{137}Cs and ^{210}Pb estimated sedimentation rates were averaged to give an overall sedimentation rate that would apply for the whole length of the sediment core.

Diatom analysis

Separate portions of selected homogenized samples were prepared and analyzed for diatoms according to Parsons (1998). A 100 µl aliquot containing a known concentration of glass microspheres (20 µm diameter, Unisciences®, Ltd.) was added to an approximate 0.5 cc portion of sediment (to estimate diatom absolute abundance; Battarbee and Kneen 1982) in a 15 ml graduated polypropylene centrifuge tube and rinsed several times with 2% sodium pyrophosphate (NaPP) to remove clay particles (Bates et al. 1978). Samples were cleaned (to remove organic matter) with 2 ml of concentrated nitric acid which was boiled for 30 min in a water bath, followed by six rinses with distilled-deionized water (DDW). Rinses involved centrifuging the sample for 10 min at 2100 rpm, subsequent removal of the supernatant, and addition of fresh NaPP or DDW. Microscopic slides were made by placing one or two drops of cleaned sample (now in solution in 10 ml DDW) onto a #1 25x25 mm coverslip. The coverslip was dried and mounted on a slide with Hyrax™. Slides were examined on a Zeiss™ universal microscope. At least 250 diatom valves were counted per slide, which provides a good estimate of the diatom species ≥1.5% of the population (Parsons 1996).

Diatom data analysis

A diatom-based salinity index (SI) was formulated using a multiple-variable regression technique comparing diatom species salinity groupings with the available salinity data (Parsons et al. 1999). A similar approach was taken by Flower (1986), in his study of pH changes in lakes. The salinity groupings were determined from the literature for as many diatom species as possible that were ≥1.5% of the population. Broad salinity classifications were used to standardize the disparate salinity groupings devised by these references, and to acknowledge the dynamic environment typical of the Louisiana coastal marshes. The salinity groups used in this study, based on Kolbe (1927) and Round (1981), refer to salinity ranges within which a particular species best thrives, or most commonly occurs; oligohalobe (0-5 ppt), mesohalobe (5-20 ppt), and polyhalobe (20+ ppt). The percentage of the diatom population classified in each group was summed for each sample.

There were several cases in which a salinity classification could not be assigned to a diatom species because either the species could not be identified, or no salinity information was found in the literature. This discrepancy was corrected by dividing the percentage of the diatom population within each salinity group by the percentage that could be classified, thereby standardizing the classifications to one and allowing direct comparisons among the samples. The SI was computed using two ratios, the oligohalobe to mesohalobe component (OM) and the mesohalobe to polyhalobe component (MP) in the following manner: $SI = 35.72(OM) - 0.62(MP) + 9.17$. Our preliminary studies have shown that the SI predicts salinity with an $R^2=0.8309$ and a slope of 0.83, indicating that the SI is a reasonable proxy for salinity (Parsons et al. 1999).

Regression analysis

Sedimentation rate estimates acquired through core dating allowed us to assign an approximate date to each sample analyzed for diatoms. The SI model was then applied to the diatom data to estimate salinity trends through the core, and therefore over the time frame represented by the sediment core.

Salinity trends were estimated using regression analysis of the SI versus the core dates. Significant regression results would indicate annual trends in the SI, and therefore, salinity. Regression analyses were computed using the PROC REG procedure of the SAS® statistical program (SAS® 1988).

Salinity data analysis

Salinity data were obtained for Airplane Lake from the Louisiana Department of Wildlife and Fisheries. Salinity averaged 20.5 ppt overall with a range of 2-34 ppt over the course of the record. The data are typical of salinity data gathered around the state, in that the data were collected on a weekly basis from 1972 to 1998. The data were collected sporadically, however, resulting in actual measurements being collected only 42% of the time. We attempted to circumvent potential analytical problems due to the highly variable nature of the data and sporadic sampling by incorporating a seasonal-correction on the data. Salinity fluctuates seasonally in coastal Louisiana in response to varying river flow and precipitation (Swenson and Swarzenski 1995). Therefore, by removing this cyclic variability, we could reduce some variability and correct for sporadic sampling. For example, if salinity were measured solely during the spring flood of the Mississippi River (at a site influenced by river flow), salinity values would be lower than if salinity were measured in the winter when river flow is low. The available salinity data (i.e., not only the data used in this study) contain data in which this very scenario exists; the available data for a given year might only exist for a few spring months (e.g., Airplane Lake). Therefore, we used a seasonal-correction in an attempt to rectify such problems that can arise due to sporadic sampling.

The seasonal-correction was accomplished by calculating monthly means for the salinity data set, subtracting these values from the specific measurements taken during the respective month, then adding this difference back to the overall mean to return the data to the same range as the raw data. While this approach may be flawed (i.e., the spring flood of the Mississippi River does not always occur in April), we believed that little else could be done to correct for sporadic sampling. The salinity data were subjected to a regression analysis to determine if trends were evident.

Freshwater inputs

An index of freshwater inputs was formulated from available precipitation data and discharge data for the Mississippi and Atchafalaya Rivers. Precipitation data were obtained from the Louisiana State Climatology Office Monitoring Station in Houma (LA#4407). The data were summed into annual precipitation from 1930-1994 in preparation for further analysis. River discharge data were obtained from the United States Army Corps of Engineers (courtesy of Dr. W. Wiseman, Coastal Studies Institute, Louisiana State University, Baton Rouge, Louisiana) for Tarbert Landing, Mississippi (Mississippi River) and Simmesport, Louisiana (Atchafalaya River). The river data were converted to average annual discharge from 1900-1994. The three sets of annual data (precipitation and the two river discharge) were standardized about their means by dividing each annual value by the overall mean value. This was done so that these three principal sources of freshwater would be weighed according to their fluctuations over time. Since it would be impossible to quantitatively determine the past proportion of freshwater inputs (riverine versus precipitation, and Atchafalaya versus Mississippi River) at various sites, the three annual standardized values (only the two riverine data for pre-1930 years) were averaged to formulate a relative measure of how total freshwater inputs in the region have changed over time. The major assumption of this standardized technique is that the proportion of freshwater within each source (precipitation or riverine) entering Louisiana coastal waters has not changed over time. The relative measure values were plotted, and a regression line was fitted to the 10-year moving average to test for changing trends in freshwater inputs over this century.

RESULTS

Core dating and site descriptions

Table 1 presents a general site description, field measurements made during the core collection trip, and core dating results. Airplane Lake is a 19 hectare

Table 1. Description and core dating results for Airplane Lake. The area was estimated from scaled aerial maps. Salinity ranges were computed from available salinity data from Airplane Lake. Water depths were recorded from field observations; major plant communities were identified in the field. Compaction data include the core lengths prior to and after extrusion, from which % compaction values were computed. See text and refer to Figs. 2 and 3 for core dating results and discussion.

surface water area (hectares)	19
salinity (ppt)	2-34
water depth (m)	1-2
major plant community	*Spartina alterniflora*
confirmed stability (via maps)	since 1955
date cores were collected	8/26/93
pre-extrusion core length (cm)	110.4
post-extrusion core length (cm)	105.7
% compaction	4.3
^{137}Cs peak (cm)	9.5
^{137}Cs sedimentation rate (cm yr^{-1})	0.33
^{210}Pb slope (ln (^{210}Pb) pCi g^{-1})	-0.0997
^{210}Pb slope standard error	0.011564
^{210}Pb sedimentation rate (cm yr^{-1})	0.31
overall sedimentation rate (cm yr^{-1})	0.32
time frame represented by the core	1664-1993

water body surrounded by salt marsh dominated by *Spartina alterniflora*. The post-extrusion sediment core length was 105.7 cm and compaction was 4.3%.

The peak in ^{137}Cs activity represents the year(s) 1963/4 (Milan et al. 1995), and occurred at a core depth of 9.5 cm (Fig. 2). The ^{210}Pb regression line (Fig. 3) indicates that both dating methods gave similar estimates of sedimentation rates (0.31 vs. 0.33 cm yr^{-1}). We averaged the two rate estimates to give an overall sedimentation rate estimate of 0.32 cm yr^{-1} on the compacted sediment core. The core contained a record of 330 years according to these dating results, although this estimate may be in error as the ^{210}Pb results are only applicable to 150 years BP at best.

Diatom analysis

A total of 27 samples were counted (1-10, 13, 17, 20-22, 24, 26, 28, 30, 40, 50, 60, 70, 80, 90, 100, and 106 cm) in which 7762 diatom valves representing 165 species from 55 genera were identified and enumerated. Diatom data from below 30 cm in the APL core were not used in further analyses because of poor diatom preservation. Therefore, results are only presented for the top 30 cm of the APL core, which was estimated to represent from 1902 to 1992. Of the 165 species, 78 were >1.5% abundant in any one sample (in the upper 30 cm of the core), and 31 of these could be classified in the halobien system (Table 2). The proportion of valves counted that could be classified in the halobien system ranged from a low value of 54% (1 and 4 cm) to a high of 85% (20 cm). Overall, 71% of all valves counted could be classified.

Statistical analyses

Regression analysis of the salinity index demonstrated that there was an overall significant increase in salinity of 0.13 ppt yr^{-1} over the length

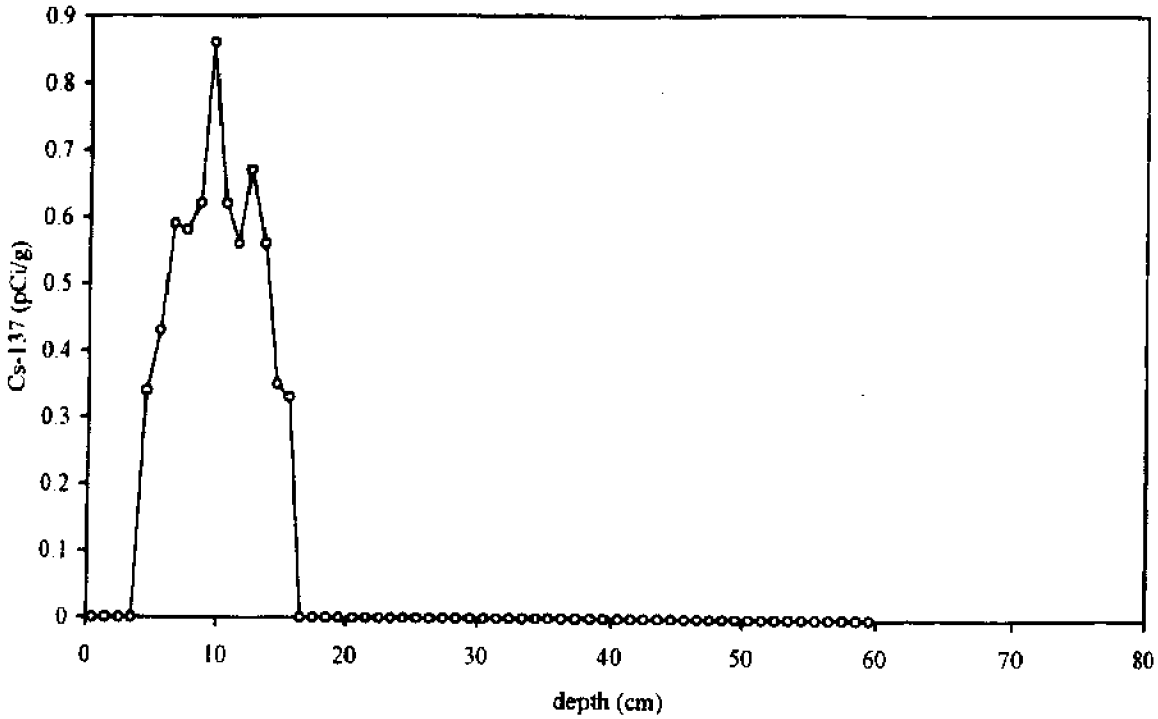

Fig. 2. ^{137}Cs activity versus core depth at Airplane Lake.

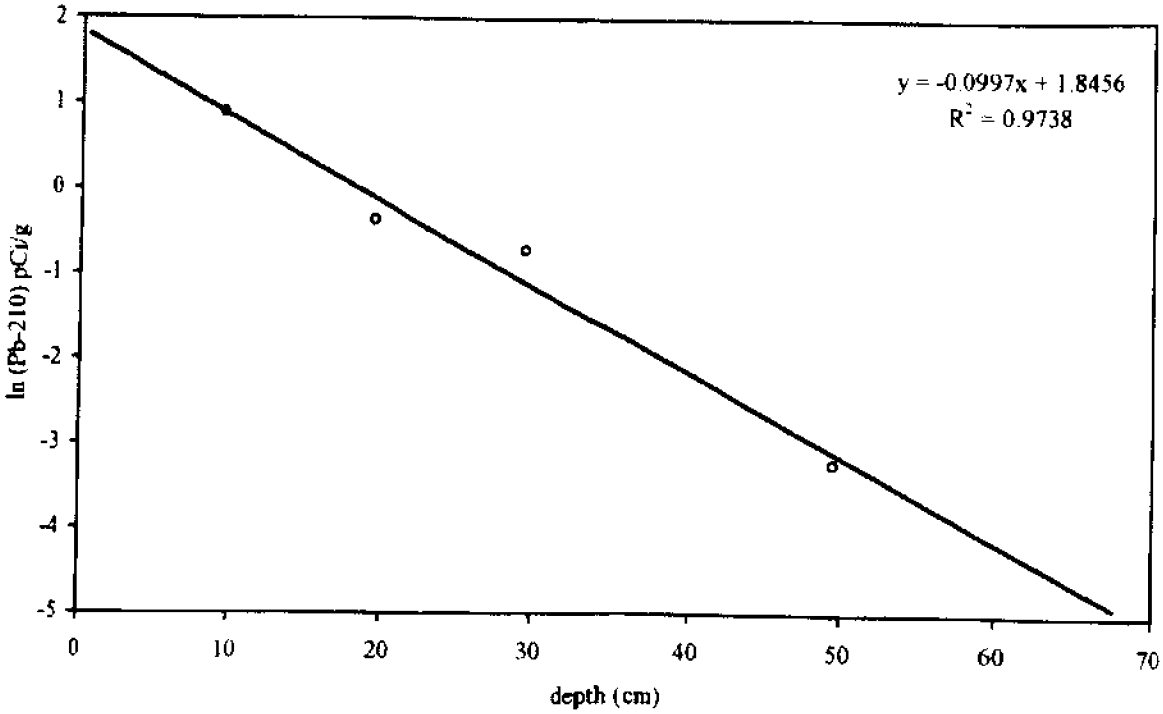

Fig. 3. Regression of ln(^{210}Pb) activity versus core depth at Airplane Lake.

Table 2: A list of all diatom species counted that were ≥1.5% relative abundance in at least one sample that could be classified following the halobien system of Kolbe (1927) and Round (1981) (o=oligohalobe, m=mesohalobe, p=polyhalobe). The references used to obtain these classifications are indicated by the superscript above each classification, which refer to the following works: 1-Foged 1980; 2-Foged 1987; 3-Marshall and Alden 1993; 4-Prasad et al. 1990; 5-Caljon 1983; 6-Vos and deWolf 1993; 7-Foged 1986; 8-Hustedt 1955; 9-Hendey 1964.

Species	authority	halobien classification
Achnanthes brevipes var. *intermedia*	(Kützing) Cleve	$o^{1,6}$
Achnanthes delicatula	(Kützing) Grunow	m^{1}
Actinoptychus senarius	Ehrenberg	$p^{2,6,7}$
Amphora copulata	Giffen	o^{1}
Aulacoseira granulata	(Ehrenberg) Simonsen	$o^{1,6}$
Biremis ambigua	(Cleve) Mann	m^{9}
Caloneis westii	(W. Smith) Hendey	m^{1}
Cocconeis disculoides	Hustedt	o^{6}
Cocconeis placentula var. *euglypta*	Ehrenberg	$o^{1,6,7}$
Craticula cuspidata	(Kützing) Mann	$o^{1,2}$
Cyclotella caspia	Grunow	m^{3}
Cyclotella choctawhatcheeana	Prasad	m^{4}
Cyclotella meneghiniana	Kützing	o^{5}
Diploneis didyma	(Ehrenberg) Ehr.	p^{2}
Ditylum brightwellii	(West) Grunow	m^{2}
Fallacia forcipata	(Greville) Mann	$p^{2,7}$
Fragilaria brevistriata	Grunow	$o^{1,6}$
Fragilaria pinnata	Ehrenberg	$o^{1,6}$
Gyrosigma peisonis	(Grunow) Hustedt	m^{5}
Melosira moniliformis	(O.F. Muller) Agardh	p^{6}
Melosira nummuloides	(Dillwyn) Agardh	m^{1}
Navicula abunda	Hustedt	m^{8}
Navicula pusilla	W. Smith	m^{6}
Navicula salinarum var. *minima*	(Grunow) Colby	m^{2}
Navicula yarrensis	Grunow	p^{2}
Nitzschia compressa	(Bailey) Boyer	m^{5}
Nitzschia lanceola	Grunow	p^{2}
Nitzschia panduriformis	Gregory	p^{2}
Nitzschia scalaris	(Ehrenberg) W. Smith	$m^{5,8}$
Petroneis marina	(Ralfs in Pritchard) Mann	m^{5}
Tryblionella granulata var. *granulata*	(Grunow) Mann	p^{2}

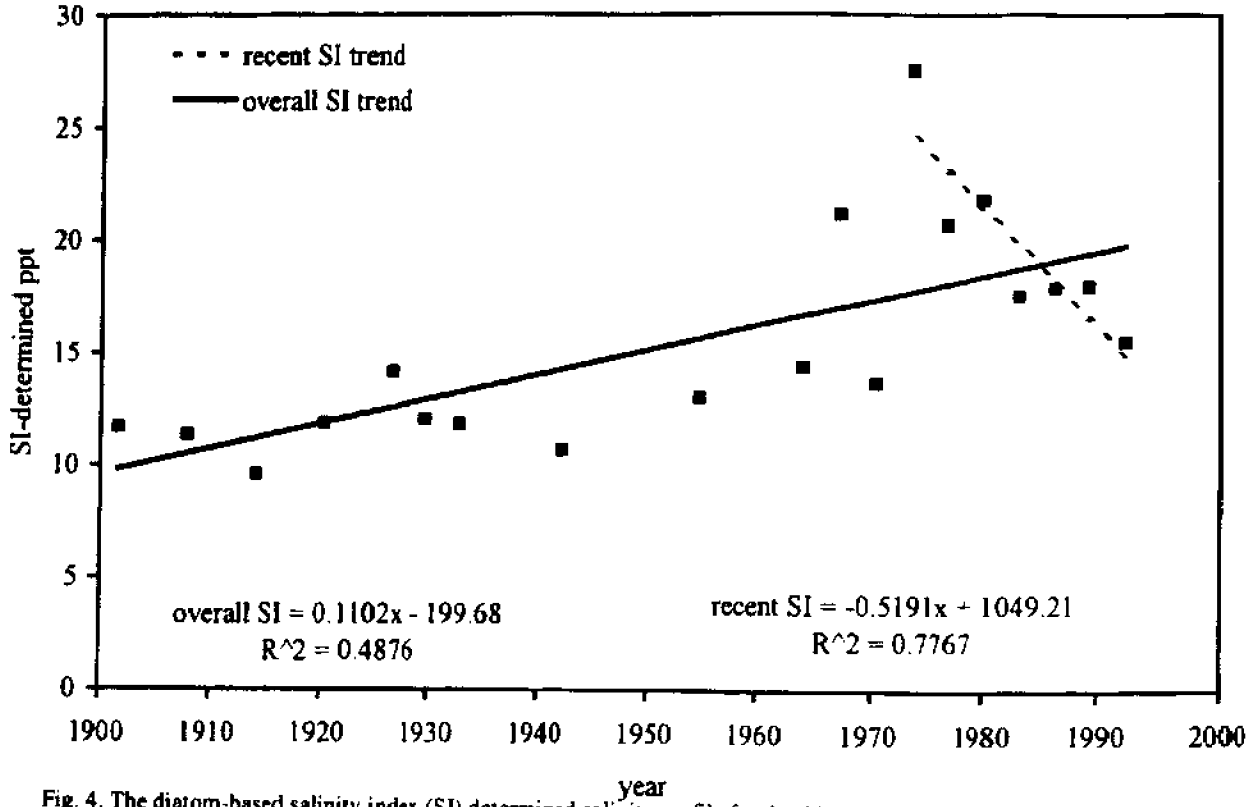

Fig. 4. The diatom-based salinity index (SI) determined salinity profile for the Airplane Lake sediment core. Statistically-significant overall and recent trends in the SI are illustrated.

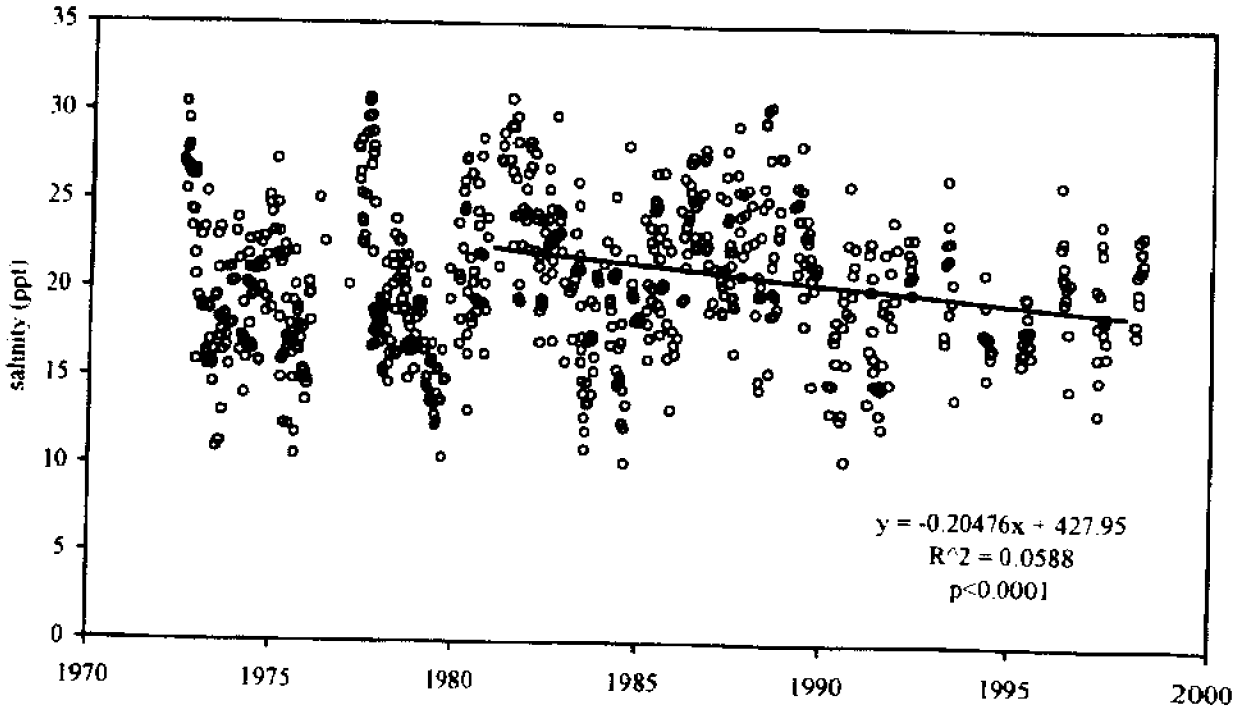

Fig. 5. Salinity data from Airplane Lake: 1972-1998. A statistically-significant regression line was fit to the data from 1981-1998.

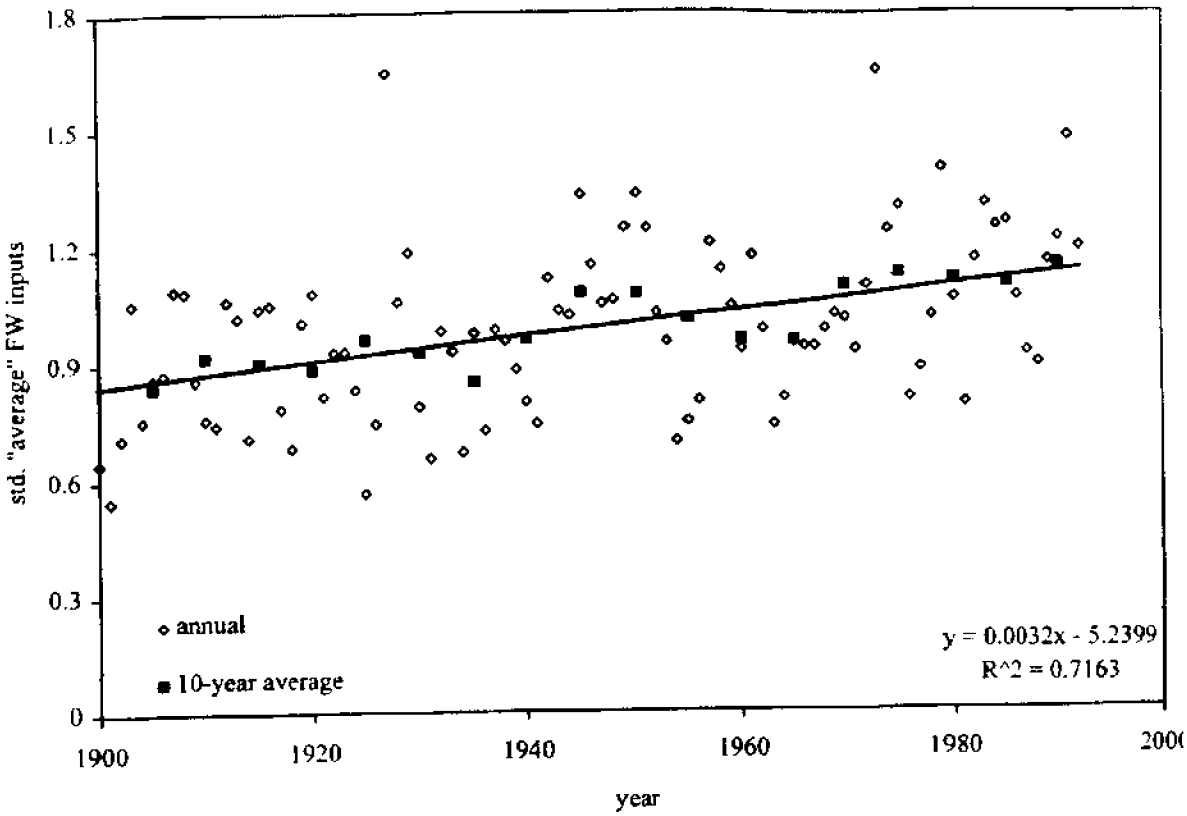

Fig. 6. Annually-averaged, standardized freshwater inputs to the region from 1900-1994. Moving 10-year averages (plotted every five years) are also plotted, and were used to generate the significant regression line ($p<0.01$).

of the core ($p=0.0008$; Fig. 4). The trend has two components. The increase mainly occurs between 1902 and 1973 ($R^2=0.4726$; $p=0.0134$). A significant, recent trend of decreasing salinity is evident in the core between 1973-1992 (-0.52 ppt yr^{-1}; $p=0.0088$). A significant ($p<0.0001$) decreasing trend is also evident in the actual salinity data (Fig. 5) between 1981-1998 (-0.2 ppt yr^{-1}). Freshwater inputs have increased this century at an average rate of 0.3% yr^{-1} (Fig. 6) based on the significant trend seen in the moving average.

DISCUSSION

Salinity changes at Airplane Lake

The SI reconstruction of salinity at Airplane Lake (Fig. 4) displays significant changes in salinity and can be summarized as three characteristics; there is an overall increase in salinity between 1902 and 1992; the increase is most prevalent between 1902 and 1973; salinity levels appear to decrease between 1973 and 1992, possibly reflecting increased freshwater inputs (Fig. 6). This conclusion is supported by work presented by Wiseman et al. (1990a) and Bratkovich et al. (1994), who acknowledged an increase in river discharge since at least the 1950s. Wiseman et al. (1990a) also demonstrated that a negative relationship exists between Mississippi River discharge and salinity at stations near APL (Grand Terre; Fig. 1). Wiseman et al. (1990b) indicated that this Mississippi River discharge influences both lower and upper estuarine salinities at proximal and distal locations.

Other salinity studies

The SI results are corroborated by other studies of coastal Louisiana salinity trends (Wiseman et al. 1990a; Swenson and Swarzenski 1995), providing further evidence that the SI reconstruction was accurate and can be used as a proxy for past salinity.

Wiseman et al. (1990a) analyzed available salinity records utilizing a seasonal Kendall-Tau analysis, which is a non-parametric test for a monotonic trend (not necessarily linear) that can handle the high degree of variability typical of these estuarine salinity measurements. Swenson and Swarzenski (1995) re-assessed salinity trends utilizing additional data collected after the Wiseman et al. (1990a) study was completed, by conducting various linear analyses on year groupings (i.e., 1965-1969 vs. 1969-1972) both of which account for and discount seasonal variation. These studies reveal that salinity data from a different monitoring station near Airplane Lake (Grand Terre, S315) displayed similar trends over the same time periods as the SI did. Wiseman et al. (1990a) showed that there was an overall significant monotonic decrease in salinity at station S315 over the time span analyzed (1962 to 1985). Swenson and Swarzenski (1995) studied the expanded data from S315 in smaller increments and found that there was a significant increase in salinity from 1955-1969, followed by a significant decrease from 1969 to 1994. The APL SI showed similar trends (Fig. 4).

While the salinity increased overall from 1902 to 1992 (mainly between 1902 and 1973), it significantly decreased between 1973 and 1992. Van Sickle et al. (1976) reported a northward extension of oyster leases between 1947 and 1975, possibly due to increasing salinities in northern Barataria Bay, which is in agreement with the APL SI results. It would be interesting to see, however, if oyster leases have retreated southward since 1975, since our results and Swenson and Swarzenski (1995) indicate that salinities appear to have been decreasing in and around Barataria Bay (including APL) since the 1970s.

Comparison between the SI and actual salinity data

The salinity data from Airplane Lake were successfully utilized in the formulation of the SI regression model (Parsons et al. 1999) implying that the SI was in agreement with the salinity data used in this study. This statement is strengthened by trend analyses of the salinity data in which recent (<20 years) decreasing salinity trends were evident at Airplane Lake (-0.2 ppt yr^{-1} between 1981-1998; $p<0.0001$; Fig. 5) just as the SI core reconstructions predict. The trend in the salinity data, however, was less than the SI trend (-0.52 ppt yr^{-1}), either reflecting the sporadic nature of the salinity data, or model error (diatom misclassification or core dating error).

The salinity data used in this study were discrete data, and the sporadic nature of the data collection may have resulted in a poor representation of the salinity regime. This is best exemplified by the observation that data were collected less than 50% of the time, and were often collected solely in the spring (e.g., March through May between 1992 and 1998). Additionally, data were collected with relatively low frequency (weekly), which does not reflect the variability of salinity in coastal Louisiana waters. The hydrology of Louisiana's coastal marshes is very dynamic. Cyclic salinity variability is influenced by diurnal tides (on daily scales), river flow, and precipitation (seasonal to annual scales; Swenson and Swarzenski 1995). Salinity measurements should take these cyclic variations into account, which is generally not the case.

Diatoms and salinity

The responses of diatoms to salinity can vary widely between species and within a species. There is evidence that a diatom species can display different salinity optima in different water bodies, although tolerance ranges generally overlap (Cumming and Smol 1993; Wilson et al. 1996). Some diatom species may have very large tolerance ranges (i.e., euryhaline) and therefore will not be useful to reconstruct estuarine salinities (Juggins 1992). Wilderman (1987) believed that this was especially true at salinities between 8-16 ppt, in which no reliable indicators may exist. Many species will exhibit a salinity optima. Because these responses are not linear, linear models (as was used for this study) might not be the best analytical approach (Juggins 1992). Additionally, a particular diatom species response to salinity changes may differ based on the duration, frequency, and magnitude of change (Carpelan 1978; Admiraal and Peletier 1980). It is important, therefore, to know how salinity changes (frequency, etc.) and how a species will react to these salinity changes.

Core dating

Some of the variability in the SI model could be due to core dating errors (e.g., counting (^{137}Cs) and regression (^{210}Pb) errors) which could cause an inaccurate date to be applied to a core depth, thereby altering trend analysis results. The ^{137}Cs counts were designed to limit counting error to 10% near the 1963/4 peak (Milan et al. 1995), and the ^{137}Cs profile for the Airplane Lake core was defined enough that this peak is clear (Fig. 2). The ^{210}Pb profile (Fig. 3) was clear after the analysis of only four samples, and resulted in a high R^2 (0.9738). The APL sedimentation rate estimates determined by ^{137}Cs and ^{210}Pb were nearly identical (0.33 vs. 0.31 cm yr^{-1}; Table 2), so we are confident that the core dates are accurate for the APL core.

DeLaune et al. (1978), however, analyzed sediment cores from Airplane Lake for ^{137}Cs and estimated sedimentation rates at 1.1 cm yr^{-1}. The differences between our estimates and DeLaune et al. (1978) are not surprising, as there were differences in sample analysis. DeLaune et al. (1978) analyzed 3-cm sections, we analyzed 1-cm sections offering better resolution. DeLaune et al. (1978) used a lithium-drifted germanium detector to count γ particles, an instrument that is less accurate than the germanium "N" type coaxial detector we used some 15 years later. DeLaune et al. (1978) and Milan et al. (1995) both reported a high degree of core-to-core variability both within and around Airplane Lake. DeLaune et al. (1978) reported that one core taken in Airplane Lake displayed a prominent ^{137}Cs peak while a second core taken 350 m away did not. Milan et al. (1995) reported that ^{137}Cs sedimentation rate estimates differed over 50% at different locales, and over 30% within replicate cores at the same location. Lastly, our sedimentation rate is estimated for the compacted sediment core, whereas the DeLaune et al. (1978) value is for uncompacted cores. We gave our values for compacted cores because we were not confident in our estimates of compaction during core collection, and we were primarily interested in dating the core versus estimating actual sedimentation rates. Therefore, the differences in sedimentation rates between our study and others are explainable and expected.

Justification of the salinity index

We believe that the differences that exist in the magnitude of the decrease in the salinity and SI trends for Airplane Lake are primarily due to the sporadic nature of the salinity data and the simplified, linear nature of the model rather than inherent model or core dating errors. Our results demonstrate that the SI can be utilized successfully as a proxy for salinity changes. This conclusion is supported by the similarities presented between the SI trends and salinity data from Airplane Lake (this study) and Grand Terre (Wiseman et al. 1990a; Swenson and Swarzenski 1995). This study demonstrates that salinity signals are evident in a coastal marsh sediment record, regardless of the high variability typical of this environment. Furthermore, a recent decreasing trend in salinity is evident, possibly reflecting increased freshwater inputs into the coastal waters of Louisiana. The decreasing trend exhibited by the SI indicates that salinity levels may be dropping on the order of 5 ppt per decade at Airplane Lake. These results cast some doubt on how freshwater diversions (which will lower salinities further) might affect marsh ecosystems if salinities are already decreasing at some locations. For example, the optimum salinity range for oyster growth and survival for Louisiana has been estimated to be 5-15 ppt (Galtsoff 1964; St. Amant 1964). If salinities are already decreasing in some areas, and/or will be decreasing further with freshwater diversions, oyster communities could be adversely impacted. Hypothetically, other organisms dependent on brackish water conditions could also be impacted by decreasing salinities including brown shrimp (Barrett and Gillespie 1973), spotted seatrout (Bourgeois et al. 1995), and salt marsh plants (e.g., *Spartina alterniflora*). We argue, therefore, that available salinity data are inadequate to properly assess how salinity has changed in coastal Louisiana (over periods >30 years) due to human activities, and that salinity reconstructions of dated sediment cores, utilizing the model presented here or a future, more refined model, can provide a useful tool to make accurate assessments.

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Annual Salinity and Nutrient Budget of Lake Pontchartrain and Impact of the Proposed Bonnet Carré Diversion

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ABSTRACT: A nutrient mass balance identifies the total mass load of the nutrient entering a waterbody, loss across the downstream boundary, and the rate at which the material is synthesized or lost within the waterbody. Nutrient budgets for total phosphorus (TP) and total nitrogen (TN) were developed, along with budgets for lake salinity and volumetric water flows. The analyses reported here were initiated to support the evaluation of a proposal to divert a small fraction of Mississippi River discharge through Lake Pontchartrain. These analyses determine the sensitivity of Lake Pontchartrain to nutrient loading, and provide a basis for development of more complex hydrologic and water quality models. Discharge and nutrient loading data have been analyzed using simplified formulas which predict annual average nutrient concentrations within the Lake. For other aquatic ecosystems, this simplified analytical approach has often proven to be a valuable management tool in support of environmental decision making. Total freshwater inflow, Q_T, is estimated to be 13.2 km^3 yr^{-1}, or an annual average inflow of 419 m^3 s^{-1} (14,800 cfs). The proposed diversion would increase freshwater inflow by 6.6 km^3 yr^{-1}. Average residence time is projected to drop from 102 d to 76 d following implementation of the diversion. In Lake Pontchartrain, projected annual average TP and TN concentrations without the proposed river diversion project are 0.060 mg-P l^{-1} and 0.65 mg-N l^{-1}. With the proposed diversion these concentrations are projected to rise to 0.071 mg-P l^{-1} and 0.86 mg-N l^{-1}.

Introduction

There is a need for simple quantitative assessment tools for evaluating the impact of nutrient additions on lakes and estuaries. In the U.S., the Clean Water Act requires the determination of the total maximum daily load (TMDL) of nutrients and other potential pollutants, and the allocation of these loads among point and nonpoint sources within a watershed. To this end, EPA has developed wasteload allocation guidance for TMDL determinations for control of eutrophication in lakes and impoundments (Mancini et al. 1983). This EPA guidance reviewed and was based on the extensive earlier work of Vollenweider (1976) and others. Here, as far as possible, we have followed this guidance. This study provides a case study for the application of these techniques to an estuary, and illustrates which factors must be included to extend the procedures of Mancini to coastal waters.

A proposed Mississippi River diversion at the site of the existing Bonnet Carré Floodway led to public concerns about the potential eutrophication of Lake Pontchartrain. This river diversion was designed to reduce salinities in oyster beds near the Mississippi State border, beyond the downstream

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boundary of this study. Although alternative diversion discharge schedules have been considered during project re-evaluation, only the monthly diversions proposed in the general design memorandum (GDM) are analyzed in this paper (United States Army Corps of Engineers 1990). Monthly GDM diversion discharges from January through December are 0, 0, 306, 850, 473, 413, 91, 74, 57, 156, 91, and 0 $m^3 s^{-1}$. The GDM discharges are higher than those proposed in other plans which have been considered since the publication of the GDM. Thus, the GDM provides a maximum diversion scenario, and diversions of smaller size should have proportionately smaller impacts.

Ideally, a nutrient budget analysis considers total nutrient concentration, incorporating all biologically available forms of the nutrients, including nutrient which has been sequestered or incorporated into planktonic biomass. Total phosphorus (TP) analysis (American Public Health Association 1992) provides such an estimate, but may in some instances also include P which is unavailable. Total nitrogen (TN) is not directly measured, and must be calculated as the sum of measured components. Total Kjeldahl nitrogen (TKN) measures the concentration of organic nitrogen and ammonia nitrogen (Metcalf and Eddy Inc. 1991; American Public Health Association 1992). Therefore, TN is estimated as the sum of TKN, and nitrate plus nitrite N. In rivers and estuaries with short displacement time, both TP and TN may often be adequately modeled as conservative materials. In lakes and estuaries with long displacement time, it is unlikely that such a conservative model would be adequate because both TP and TN loss within the waterbody are significant relative to hydraulic displacement.

Methods

Site Description

Lake Pontchartrain is located in southeastern Louisiana. Lake Maurepas, a smaller coastal lake, lies upstream of Lake Pontchartrain. These lakes are hydrologically connected by a short waterway named Pass Manchac. At its downstream boundary, Lake Pontchartrain connects to Lake Borgne and the Mississippi Sound through Chef Menteur Pass and Pass Rigolets. Circulation is driven primarily by wind, rather than by river discharge or tidal exchange (Stone et al. 1972; Gael 1980). The long retention time of Lake Pontchartrain compared to other Gulf Coast estuaries (Solis and Powell 1999) and limited tidal exchange (Swenson 1980) reduce the spatial variation of salinity and nutrients within Lake Pontchartrain. These characteristics make Lake Pontchartrain well suited for the analytical methodologies developed for lake nutrient budget analysis. The Lake Pontchartrain watershed (14,490 km^2) drains a large area of southeastern Louisiana and a smaller area of Mississippi. The potential sources of freshwater input, including drainage basins, open water areas, pumped stormwater from the New Orleans area, and the proposed diversion were numbered (Figure 1). Sub-basin drainage areas were reported by Sloss (1971) and Earl (1992).

Freshwater Inflows

The U.S.G.S. maintains continuous discharge records on the major streams in the Lake Pontchartrain Basin (Arcement et al. 1993). Monthly mean gaged discharges for the period-of-record were used as the basis for monthly discharge estimates. The period-of-record for these discharge statistics ended with water year 1992. Stream gaging stations used in this study are located on the Amite River near Denham Springs, the Tickfaw River at Holden, the Tangipahoa River at Robert, and the Pearl River near Bogalusa. Drainage areas associated with these gaging sites are respectively 3,315 km^2, 640 km^2, 1,673 km^2, and 17,024 km^2. Conventional discharge gage sites must be located where a stable and sensitive stage-discharge relationship exists, often well upstream of the mouth. Additionally, small drainage sub-basins have no appropriate gaging site. Thus, gaged discharge from the Lake Pontchartrain Basin (excluding the Pearl River Basin) represents only 39% of the total Basin area. Runoff, stream discharge per unit of watershed drainage area, is commonly used to extrapolate discharge to ungaged areas. Average monthly runoff calculated for each gaging station provides a basis for estimating average monthly discharge from each Lake Pontchartrain sub-basin by multiplying the most appropriate gaged runoff by the sub-basin drainage area.

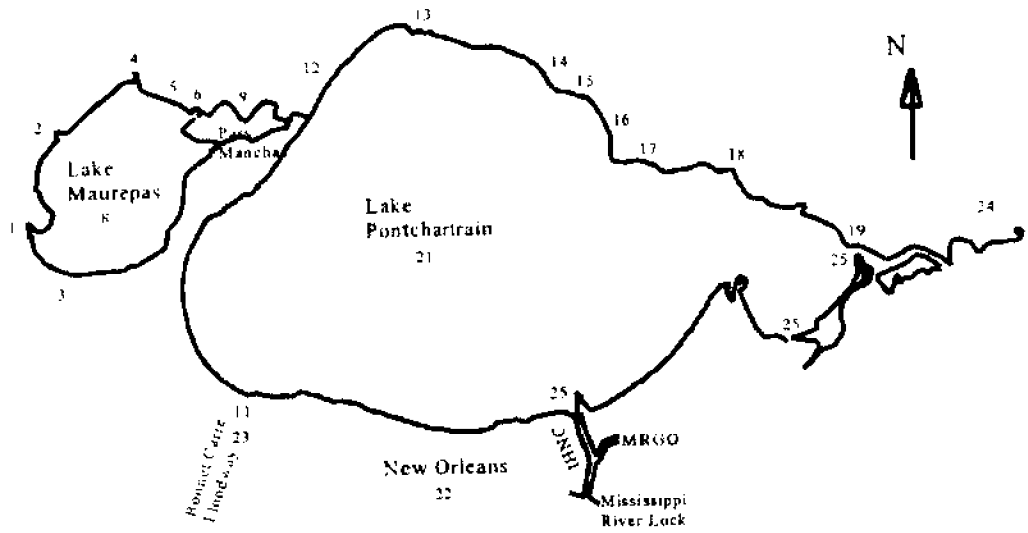

Figure 1. Map of Lake Pontchartrain system with loading sources indicated by numbers.

Freshwater inflow to Lake Pontchartrain must include not only all tributaries, but also some fraction of the Pearl River flow (Sikora and Kjerfve 1985). The mouth of the Pearl River is located just outside of Lake Pontchartrain at Pass Rigolets, and therefore contributes a part of its flow to the Lake during each tidal cycle. Assuming a symmetrical tide cycle, this fraction must be less than half, and for the purposes of this analysis one third of the Pearl River flow was assumed to contribute to the freshwater inflow and nutrient loading of Lake Pontchartrain.

Discharge from leakage through the existing Bonnet Carré spillway structure was estimated to be 15% of the flow and load of the proposed diversion. Nutrient loading from the proposed structure has been adjusted to include the anticipated load reduction that will result from operation of the sedimentation basin (Benndorf and Klaus 1987; Putz and Benndorf 1998). Based on an estimate that 30% of the sediments will be removed within the sedimentation basin, it is assumed here that this will result in a 20% reduction in TP, and a 10% reduction in TN loading from the proposed Mississippi River diversion.

An alternative method was used to estimate pumped urban stormwater inflow. The runoff coefficient is the ratio of runoff to rainfall, and can provide an estimate of runoff in ungaged watersheds (Mancini et al. 1983). Runoff coefficients typically vary between 0.1 and 0.6, with increasing values being associated with increased impervious ground cover in the drainage basin (Bowie et al. 1985). Total annual runoff values indicate that considerably less than half of the average annual rainfall of 156 cm (Louisiana Office of State Climatology) is discharged as runoff from the watershed. East Bank Jefferson Parish stormwater pumping stations records for 1988 indicate an annual runoff coefficient of 0.5 (Earl 1992, Volume 2, page 5-22). Total average monthly discharge from the New Orleans area pump stations were therefore estimated as 50% of the average monthly rainfall falling over the pumped drainage areas (Earl 1992, Volume 2, page A7). Estimated drainage areas (Jefferson Parish 131 km^2, Orleans Parish 134 km^2) and water quality of pumped runoff were also obtained from Earl (1992, Tables 5-6 and 5-7).

Nutrient Concentration and Loading

The Office of Water Resources of the Louisiana Department of Environmental Quality (LDEQ) maintains a statewide water quality monitoring network. This fixed station, long-term surface water quality monitoring network currently

provides data from 186 monitoring sites. Sampling and laboratory procedures follow extensive quality assurance plans available from LDEQ. Field measurements and samples are collected at one meter or, in shallow streams, at half depth. Laboratory nutrient analysis is performed on whole water samples and generally follows Standard Methods (American Public Health Association 1992). Data from 28 of these sites were used in this assessment. Most sites have more than 10 years of data, and many have been monitored for more than 20 years. Most sites are monitored on a monthly basis, however some newer sites are monitored on a bimonthly basis. The period-of-record utilized here ends in August 1993.

An environmental organization voiced a concern during comments on the early plans for the reevaluation of the Bonnet Carré diversion project. It was suggested that pollutant levels may have increased over past decades, and load calculations in this study should be representative of present conditions. Bahr (1983) also concludes that P loading has increased over past decades. Therefore, in this study, monthly mean concentrations, used here in load calculations, were calculated from observations more recent than January, 1985. This should both provide an adequate number of monthly observations, and also reduce the influence of any long-term trends in concentration which may exist because of land use changes or other historical factors.

The term load or loading rate is the rate at which mass of substance is entering a waterbody through point sources, nonpoint sources, or tributary inflow. Regardless of temporal, spatial, or dynamic complexity, all models of nutrients and eutrophication require the development of nutrient load estimates. The load (mass/time) of a substance entering a system through water inflow is calculated from concentration of the substance (mass/volume) multiplied by the stream discharge (volume/time). Concentration and volumetric inflow or stream discharge for each source of water entering the waterbody are necessary components of a load estimate. It must also be noted that average values for concentration and discharge must be used with some care in calculating loads because, in general, the average of the product of two variables does not equal the product of their averages. Consequently, the product of average annual discharge and average annual concentration may provide inaccurate estimates of average annual load. In order to reduce this undesired effect, monthly average loads were here calculated and summed to develop annual average load estimates.

Simplified Model Analysis

Following Mancini et al. (1983) we represent Lake Pontchartrain as a single well-mixed compartment. Assuming a first order loss term corresponding to net sedimentation:

$$\frac{d(c_t V_t)}{dt} = W_t - (Q_t + K_s V_t)c_t \qquad (1)$$

where t = time and subscript t denotes time dependent variables;
c = concentration within the waterbody;
V = volume;
W = sum of all mass loads including all internal waterbody sources;
Q = total of all volumetric flows into the waterbody;
and K_s = net sedimentation rate.

The long-term average value of the left hand side of Eq. 1 should approach zero. This yields a result analogous to the steady-state solution of Eq. 1:

$$c = \frac{W}{Q + K_s V} = \frac{W/V}{\rho + K_s} \qquad (2)$$

where $\rho = Q/V$, the hydraulic displacement rate; and all variables in Eq 2 are averaged over a period which is long relative to $1/(\rho + K_s)$.

The hydraulic displacement time, $\tau = 1/\rho$. Loading rate comparisons among waterbodies is facilitated by normalizing Eq. 2 for water surface area:

$$c = \frac{w'/z}{\rho + K_s} \qquad (3)$$

where w' = load divided by surface area;
and z = mean waterbody depth.

The occurrence of salinity within Lake Pontchartrain demonstrates that in addition to the freshwater inflows, there is a net saltwater inflow, Q_B, entering from the downstream (seaward) boundaries. The method applied here to estimate Q_B is a variation of the "fraction of freshwater method" reviewed in Bowie, et al. (1985, page 43-44) and by Solis and Powell (1999), and utilized by Swenson (1980) in estimating Lake Pontchartrain retention time. Average salinity, S, represents a special case of Eqs. 1-3, with K_s=0, and the salinity load $W = Q_B S_B$, where S_B is the boundary water salinity. This assumes that the salinity of the freshwater inflow is negligible. Average total inflow, Q, equals the sum of the average total freshwater inflow, Q_T, and Q_B. From Eq. 2, average salinity is:

$$S = \frac{Q_B S_B}{Q_B + Q_T} \qquad (4)$$

Results

For Lake Pontchartrain, area (A) is 1637 km^2 (Sloss 1971), mean depth (z) is 3.4 m (Stone et al. 1972), resulting in volume (V) of 5.56 km^3. Including inflows as described earlier, Q=19.653 $km^3\ yr^{-1}$ under current conditions, and Q=26.252 $km^3\ yr^{-1}$ after the proposed diversion. Thus, τ= 0.28 yr (102 d) without the proposed diversion, and τ= 0.21 yr (76 d) after the proposed diversion.

Boundary salinity and nutrient concentrations can be estimated from the highest salinity values observed at the Lake Pontchartrain outlet monitoring sites, Chef Menteur Pass and Pass Rigolets. Over a period-of-record beginning in 1978, maximum salinity observed at these sites was 12.9 ppt, and the 95th percentile salinity was 9.8 ppt. It is assumed that the characteristics of the boundary waters entering the lake are similar to those which are present at the outlet sites under these conditions of elevated salinity. An estimate of the boundary salinity and concentrations was performed by averaging observations where salinity met or exceeded 9.8 ppt. Average salinity under this constraint was 10.8 ppt, and average nutrient concentrations were: TP=0.082 mg-P l^{-1}, nitrate plus nitrite=0.037 mg-N l^{-1}, and TKN=0.669 mg-N l^{-1}. Since 1985, average salinity was 3.6 ppt at the Lake Pontchartrain monitoring stations.

Average annual seawater inflow, Q_B, is estimated using Eq. 4 to be 6.44 $km^3\ yr^{-1}$, or 33% of total inflow. Annual average total discharge is estimated to be 19.65 $km^3\ yr^{-1}$ (Table 2). Tributaries flowing from the watershed provide 36%, and the Pearl River is estimated to provide 20% of total inflow. The remaining 11% of flow is divided between urban pumped stormwater discharge, direct net precipitation, and leakage from the Mississippi River. With these results, the average salinity in Lake Pontchartrain after diversion can be estimated by including the diversion discharge in a recalculation of Eq. 4. Under the GDM diversion scenario average Lake Pontchartrain salinity is projected to decrease from 3.55 to 2.66 ppt.

Annual Loading and Areal Loading

Average annual TP and TN loads total 3,300 and 35,700 metric tons yr^{-1}, respectively (Table 2). Loading from the diversion is estimated to add 1,050 and 17,030 metric tons yr^{-1} of TP and TN. The TN:TP mass ratio of total nutrient source loadings is thus estimated to be 10.8 without the diversion, and 12.1 following the proposed diversion. A characteristic inflow concentration may be calculated by dividing total load by total discharge. For TP and TN this is 0.168 and 1.82 mg l^{-1} without diversion, and 0.166 and 2.01 with diversion. Areal loading is loading divided by waterbody surface area. Areal loading of TP rises from 2.02 to 2.66 g $m^{-2}\ yr^{-1}$ after diversion, and TN rises from 21.82 to 32.22 g $m^{-2}\ yr^{-1}$.

Annual average LDEQ TP and TN observations (Table 1) in Lake Pontchartrain were 0.060 and 0.65 mg l^{-1}, respectively. A TP net sedimentation rate, K_s, of 6.4 yr^{-1} was determined by model calibration using Eq. 2 and substituting values of volume, discharge, TP load, and average Lake Pontchartrain TP concentration. Equation 2 then projects TP concentration following diversion implementation to be 0.071 mg l^{-1}. Simplified

Table 1. Average parameters at DEQ monitoring sites (1985-1993) by class (L. PON.=Lake Pontchartrain, M. RIV.=Mississippi River, OUTLET=Lake passes, TRIB.=tributaries).

Parameter	Units	CLASS			
		L. PON.	M. RIV.	OUTLET	TRIB.
NO_x	mg-N l^{-1}	0.054	1.363	0.051	0.226
TKN	mg-N l^{-1}	0.60	0.94	0.77	0.94
TN	mg-N l^{-1}	0.65	2.30	0.82	1.17
TP	mg-P l^{-1}	0.060	0.241	0.091	0.187
TN:TP (mass)		10.9	9.6	9.0	6.2
TOC	mg-C l^{-1}	5.84	5.78	6.44	8.97
Temperature	deg C	21.0	18.8	21.3	20.8
DO	mg-O l^{-1}	8.51	8.48	8.34	6.17
Conductivity	µmho	5478	369	7965	1589
Secchi disk	cm	94.4	23.8	64.0	55.6
Salinity	ppt	3.40	0.15	5.00	0.92

Table 2. Annual discharge and nutrient loading values.

SOURCE	Q $km^3\ yr^{-1}$	TP $kg\ yr^{-1}$	TN $kg\ yr^{-1}$
Tributaries	7.178	1,366,926	8,377,933
Pearl River	3.944	345,037	3,84,112
Stormwater	0.104	292,562	322,863
Precipitation	1.000	65,484	0
Saltwater Inflow	6.436	1,036,235	6,256,029
Mississippi River	0.990	196,362	2,837,906
N-fixation			14,100,000
Total without diversion	19.651	3,302,606	35,708,843
Diversion	6.598	1,047,263	17,027,437
Total with diversion	26.249	4,349,869	52,736,280

modeling approaches for TN have not been as extensively studied and tested as those for TP. Mancini et al. (1983) suggest that an approach similar to that used here for TP should also be adequate for other nutrients. Calibration of Eq. 2 for TN results in an estimate of $K_s = 2.5\ yr^{-1}$, and a projected average TN lake concentration of 0.972 mg l^{-1} after diversion. However, this calibration neglects an estimate of load from N fixation (as well as precipitation load of TN to the lakes surfaces). Calibration for TN is therefore deemed to be inappropriate here. Mancini et al. (1983) notes that both TN and TP are removed from the system by permanent burial following settling of particulate organic materials, and conjecture that because this mechanism of loss is the same for TN and TP, it is reasonable to assume that the K_s value for TN is equal to that for TP. Assuming $K_s = 6.4\ yr^{-1}$ for TN as well as TP, a N fixation term may be added to the TN load such that Eq. 2 is calibrated. This gives a N fixation load estimate of 14,100 metric tons yr^{-1}. This estimated load is greater than any other single load source listed in Table 2, and 65% of the total load from all other existing sources combined.

Discussion

Isphording et al. (1989), and Flowers and Isphording (1990) report freshwater discharge to Lake Pontchartrain to be 6.8 $km^3\ yr^{-1}$ (7600 $ft^3\ s^{-1}$) and volume of Lake Pontchartrain to be 5.77 km^3 (2.038 10^{11} ft^3). This compares favorably with the estimates presented here (7.18 $km^3\ yr^{-1}$ and 5.49 km^3). Argyrou et al (1997) estimate a similar total volume, 6.58 km^3, but estimate annual average discharge of rivers into the Lake Pontchartrain Estuary to be only 4.48 $km^3\ yr^{-1}$ (142 $m^3 s^{-1}$). Swenson (1980) concluded that gaged discharge must be scaled by an average factor of 2.4 to provide an appropriate freshwater discharge. This compares closely with the value of 2.6 used here. Swenson also estimated a comparable displacement time in Lake Pontchartrain, 105 d, using the "fraction of freshwater method" which reduces the waterbody volume to the equivalent freshwater volume. The displacement time of Lake Pontchartrain is estimated here to be 102 d. Sikora and Kjerfve (1985) also recognized the need to scale gaged discharges to estimate total tributary discharge. They applied scaling factors ranging from 1.06 to 2.4 to their tributary inflows. Solis and Powell (1999) present displacement times of Gulf Coast Estuaries in graphical format, with Lake Pontchartrain residence time near 140 d. Argyrou et al. (1997) estimate hydraulic residence time to be 537 d. This anomalously high estimate results primarily from an underestimate of freshwater inflow and also from failure to consider saltwater inflow or volume.

Bianchi and Argyrou (1997) estimate watershed nutrient loading of phosphate, ammonium, and nitrate plus nitrite. Because those nutrients are components of the TP and TN loads, we anticipate that their associated loads should be less than the total nutrient loads estimated here. Although the total tributary inflow estimate of 142 $m^3 s^{-1}$ used by Bianchi and Argyrou is somewhat lower than the value used here, their published load estimates are anomalously high and appear to be in error. Loads can be calculated from nutrient concentrations presented by Bianchi and Argyrou, and these loads are consistent with loads calculated here.

Areal nutrient loading rate (loading per unit of lake surface) in Lake Pontchartrain is estimated to be 2.02 and 21.82 g $m^{-2}\ yr^{-1}$ for TP and TN. These values generally fall within the range of values which have been reported for lakes (Reckhow 1979; Mancini et al. 1983). Bahr (1983) projected a P load of 2 g $m^{-2}\ yr^{-1}$ near the year 2000.

Ryding and Rast (1989) state that it is rare that available nutrient sampling data will produce nutrient load estimates within ±25%. In the case of Lake Pontchartrain, uncertainty involving the Pearl River, N fixation, and the saltwater load contributions add uncertainty to the load estimates. However, for the purposes of this trophic comparison and comparison of projections with and without the implementation of the proposed diversion, these load estimates should be adequate.

Calculations presented here illustrate the importance of incorporating the seaward boundary contributions in estuarine budgets. Even in the case of Lake Pontchartrain, with a relatively limited seaward exchange, the magnitude of net saltwater inflow and total nutrient loading was nearly as large

as freshwater runoff and nutrient loading (Table 2). Failure to incorporate this flow and load source can result in over estimation of estuarine displacement time and sensitivity to loading for nutrients and other substances.

The Lake Pontchartrain seaward boundary is complicated by the location of the Pearl River at one seaward boundary channel. For the purposes of the comparison of diversion project alternatives, it is not critical to exactly identify the fraction of the Pearl discharge and load that should be incorporated with other tributary inflow. However, this is one source of uncertainty, and determination of the amount of the Pearl River flow and load entering Lake Pontchartrain is worthy of further study. Both the Pearl River contribution and the more general seaward boundary exchange should be quantified in future hydrodynamic computer modeling studies.

In this model, over 39% of present N loading results from fixation (or other unaccounted inputs such as dry deposition or precipitation). There is great uncertainty in this estimate because this loading was estimated through an indirect calculation, and is based on Mancini's conjecture that the same K_s value is adequate to model net sedimentation loss of TP, TN, and other total nutrient concentrations, such as total organic carbon (TOC). Future research in the Lake Pontchartrain Estuary, as well as other lakes and estuaries, should be directed toward obtaining a better estimate of K_s for total nutrient concentrations, and testing Mancini's conjecture. For nutrients with allochthonous sources and sinks, this will necessitate more direct estimation.

Although the stormwater discharge and load estimates are uncertain, they are small relative to the total estuary discharge and load (0.5% of discharge, 9% TP load, and 0.8% of TN load). This suggests that the urban stormwater pumps have little impact on estuary trophic state. This conclusion is likely true, but it does not follow that these nutrient sources are environmentally benign and need not be considered in a Lake Pontchartrain pollutant management plan. The pumped stormwater load likely plays an important role in the reduction of near-shore water quality, and the pulsed nature of the input may promote local algal blooms uncharacteristic of lake-wide events. Both of these potential consequences illustrate limitations of the average annual nutrient load and steady-state modeling approach presented here. More study is needed to determine the impact of these urban sources and other sources near their points of discharge. This should include application of dynamic modeling with spatial resolution adequate to identify local impacts.

The annual steady-state nutrient modeling approach presented here has clear value in support of environmental management planning. This includes plans for evaluation of overall nutrient control from point and nonpoint sources and TMDL determinations, and comparison of some impacts of alternative designs. The approach also provides a background for comparison in studies utilizing more temporally or spatially complex modeling. The modeling approach requires limited effort and computer resources relative to more complex methodologies, and provides simple straightforward predictions which support comparison of alternatives. It is essential, however, that the projections of this simple approach not be applied inappropriately. For example, this approach can not predict impacts resulting from changes in seasonal nutrient patterns, or local impacts of discharges. Applied with care, the approach is a valuable tool for environmental analysis, management, and decision support.

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Water Chemistry Dynamics in Lake Pontchartrain, Louisiana, During the 1997 Opening of the Bonnet Carré Spillway

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ABSTRACT: In 1997, the Bonnet Carré Spillway was opened for about one month as a result of the fourth largest flood of the century on the Mississippi River. Peak Mississippi flow reached 48600 m^3s^{-1} and up to 16% of the total flow was diverted through the spillway and into Lake Pontchartrain. Several water quality parameters (NO_3+NO_2, NH_4, total phosphorus, total Kjeldahl nitrogen, and total suspended sediments) were measured in the river and at six stations in Lake Pontchartrain from the beginning of the spillway opening (March 17) until late September. Just after the opening, stations in the southwestern portion of the lake became fresh and NO_3+NO_2 (>1.0 mg l^{-1}), total phosphorus (>0.1 mg l^{-1}), and total suspended sediments (>50 mg l^{-1}) were unchanged from those in the river. Concentrations of these constituents increased at the other stations, but to a lesser degree. Concentrations of NO_3+NO_2 and total phosphorus were highest from early April to mid May and returned to pre-opening levels by mid-June. The strong reductions in these two parameters was most likely due to denitrification, algal uptake - there was a large blue-green algal bloom in the lake - and P sorption to sediments.

INTRODUCTION

Estuaries in southeastern Louisiana are strongly influenced by freshwater inflow and its associated sediments and nutrients, mainly due to discharges of the Mississippi River. The high nutrient concentrations in the river discharge have been shown to have significant water quality impacts on shelf waters near the river mouth, as they lead to intense phytoplankton blooms and cause low dissolved oxygen in bottom waters when the phytoplankton sink and decompose (Turner and Rabalais 1991). A hypoxic zone of low oxygen bottom water is now widespread each year and has been linked to fish kills and other deleterious effects (Turner and Rabalais 1991).

During the spring of 1997, the lower Mississippi River experienced the fourth largest flood of this century, resulting in the opening of the Bonnet Carré Spillway from March 17 to April 17, 1997. Apparently, few studies have been conducted of massive freshwater discharges and the resultant water chemistry changes in estuaries (Nixon 1981; Kemp and Boynton 1984; Madden et al. 1988; Paerl et al. 1998; Stern et al. 1986). High concentrations of nutrients and pollutants in Mississippi River water are of concern because of the potential impacts of Bonnet Carré openings on water quality and fisheries in Lake Pontchartrain (Louisiana Department of Environmental Quality 1991). Lane et al. (1999) recently showed that there was a strong uptake of some inorganic nutrients and suspended

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sediments at the Caernarvon freshwater diversion site. Research has shown that diverting nutrient-rich water through wetlands can lead to substantial nutrient removal and to enhanced accretion (Richardson & Nichols 1985; Breaux and Day 1994). We therefore undertook an analysis of water chemistry dynamics during the 1997 opening of the spillway.

STUDY AREA

The Bonnet Carré Spillway. The Bonnet Carré Spillway was designed to carry flood waters from the Mississippi to Lake Pontchartrain when New Orleans is threatened by high water levels. It was constructed in 1931 after the devastating flood of 1927 (Barry 1997) and has been opened eight times during high flow events (Sikora and Kjerfve 1985). The Bonnet Carré Spillway is located 25 km upriver from New Orleans, Louisiana (Fig. 1). The 3.4 km wide spillway is confined by two 8.6 km levees and connects the Mississippi River to Lake Pontchartrain. There are 1300 ha of forested wetlands in the spillway, or approximately 50% of the total spillway area. A water flow regulation structure, which has 350 flood gates (each consisting of twenty 20x30 cm wooden beams), is located at the juncture to the Mississippi River. The structure is opened and closed by removing or replacing the beams one at a time. Thus, it can take several days to open or close the structure. The spillway is located where one of the many crevasses breached the Mississippi River levee in the 1800s and introduced up to 4,000 m^3s^{-1} of water into Lake Pontchartrain (Davis 1993). The present spillway was designed to divert up to 7,000 m^3s^{-1} from the river during floods.

Lake Pontchartrain. Lake Pontchartrain is a large (1630 km^2) oligohaline lake located north of New Orleans, Louisiana, with a mean depth of about 3.7 m and a volume of $1.66 \times 10^9 m^3$ (Fig. 1). Tides in the lake are diurnal with a mean range of 12 cm. Lake Pontchartrain is well mixed and is generally not stratified. In the natural state, the lake was surrounded by extensive wetlands, but large areas have been reclaimed or impounded on the south shore due to growth in the New Orleans metro-

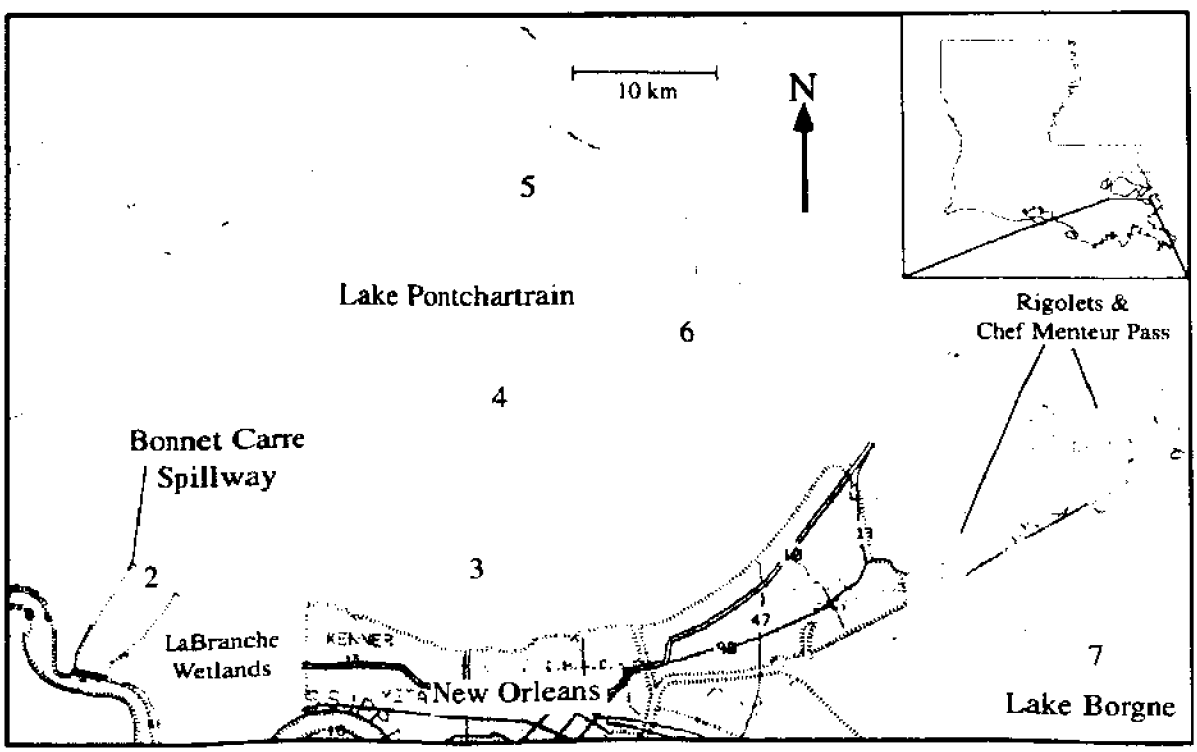

Fig. 1. Map of Lake Pontchartrain showing the locations of the Bonnet Carré Spillway and the sampling stations.

politan area. The lake receives freshwater input from several rivers as well as periodic openings of the Bonnet Carré Spillway. Without the spillway, mean freshwater input to the lake is about 370 m^3s^{-1}, resulting in a replacement time (volume of the lake divided by freshwater input) of about 51 days. Three large inlets connect the lake to the larger estuarine system. Two natural inlets, The Rigolets and Chef Menteur Pass, communicate with Lake Borgne and Mississippi Sound; while a dredged canal, the Inner Harbor Navigation Canal (IHNC) is connected to Breton Sound. The rivers supply 5% of the tidal prism in the lake, whereas the remainder enters through the tidal passes (Swenson 1981).

The 1997 Opening

During the Spillway opening, Mississippi River discharge was 39,400 m^3s^{-1} (1.39×10^6 cfs) on March 17, 48,600 m^3s^{-1} (1.72×10^6 cfs) on March 25-26, and about 30,500 m^3s^{-1} (1.08×10^6 cfs) on April 20. The Bonnet Carré Spillway was opened on March 17 as a precaution against flooding. Water flow through the spillway gradually increased to a maximum of about 6800 m^3s^{-1} (2.40×10^5 cfs) on March 25-26, or about 16.4% of the total flow of the Mississippi River at the time (Fig. 2). As river stage decreased, lessening the threat of flooding, the structure was gradually closed and flow through the spillway declined. The structure was fully closed on April 17. Approximately 1.0×10^{10} of river water was discharged from the Mississippi into Lake Pontchartrain, and the replacement time of the lake was reduced to five or six days.

METHODS

Individual water samples were collected at approximately weekly intervals from March 17 to September 22 at seven stations; the Mississippi River, 5 stations in Lake Pontchartrain, and one station in Lake Borgne near the outlet of Chef Menteur Pass. Water samples were collected 10-20 cm from the surface in acid washed glass or plastic containers with teflon coated lids. The water samples were cooled to 4°C for preservation and transported to the Corps of Engineers laboratory in New Orleans where both filtered and unfiltered samples were frozen until analysis. The samples were analyzed for nitrite+nitrate (NO_2+NO_3), ammonium (NH_4-N), total Kjeldahl nitrogen (TKN), Total Phosphorus (TP, Wershaw et al. 1987), total suspended solids (TSS, Banse et al. 1963), and salinity (Greenburg et al. 1985). At least 10 to 15% of all analyzed samples were duplicates or spikes to verify analytical performance. Total nitrogen (TN) was calculated by adding NO_2+NO_3 and TKN values. Organic nitrogen (ON) was calculated by subtracting NH_4 from TKN. For each station, the measured parameters were plotted with respect to time.

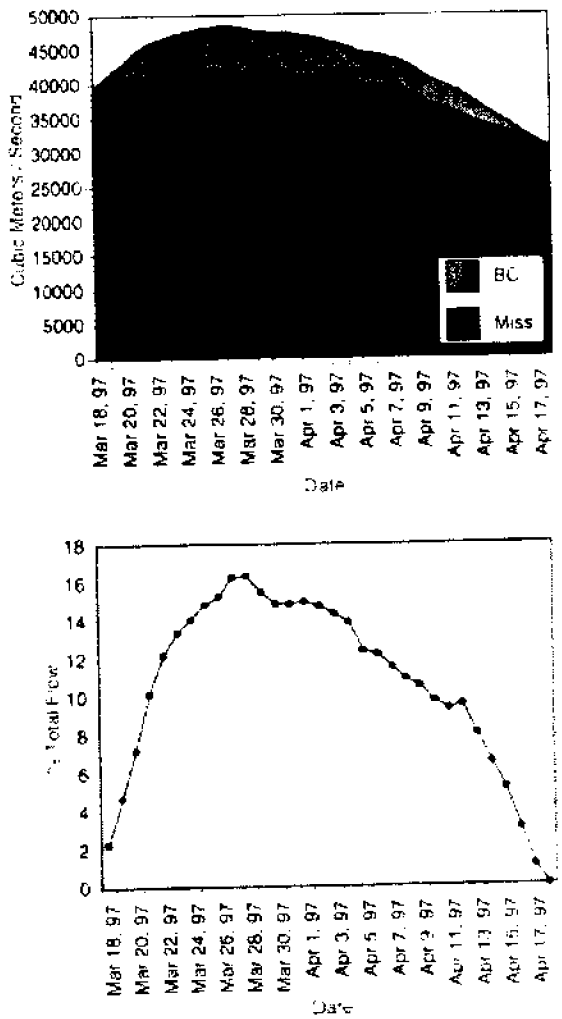

Fig. 2. Top. Total flow of the Mississippi River during the spillway opening showing the amount of water which was diverted through the spillway (BC, gray area). Bottom. Percent of total flow of the Mississippi River diverted through the spillway.

Statistics

Statistical analyses were conducted to identify changes over time and space. Temporal analysis was carried out by comparing mean concentrations from April 1 through May 5 to concentrations from July 28 through September 22 at each station; a total of five samples were used from each of these periods. These periods were chosen to represent conditions during and after the opening of the spillway, with a two week lag time in the first period to allow water to reach distant stations in the lake. Spatial analysis was carried out using the same temporal grouping (five samples during and after the spillway opening), but compared differences between stations. Due to the small sample size, nonparametric analyses using the Wilcoxon Rank Sum test was used to test for differences between means (Sall and Lehman 1996). A probability level of <0.05 was used to define a significant difference.

RESULTS

Concentrations of the different parameters in Mississippi River water varied during the diversion; TKN ranged from 0.34-0.93 (mg l^{-1}), TP 0.17-0.33 (mg l^{-1}), NH_4 0.08-1.26 (mg l^{-1}), NO_2+NO_3 1.08-1.26 (mg l^{-1}), and TSS 34-110 (mg l^{-1}). These values are somewhat low for average Mississippi River water, but perhaps the high water volume led to dilution of these constituents.

Spatial and Temporal Trends

The introduction of river water reduced salinity and increased nutrient levels in the lake (Figs. 3-8). Statistical analysis indicated that there were higher NO_2+NO_3, TN, and TP, as well as lower salinity at most stations during the period the spillway was open compared to later in the summer (Table 1). This condition was most pronounced at stations 2 and 3 where the system went completely fresh within two weeks of the opening and nutrient levels were in the same range as those in the river (Figs. 3 and 4). At these two stations, high concentrations of NO_2+NO_3 and TP persisted for about a month after the closure of the structure but decreased by early to mid June. In contrast to the rapid declines in nutrient concentration, salinity gradually increased through the study period. Total nitrogen increased at all stations, except 6, during and directly after the opening of the spillway, and though not statistically significant, there were increased ON concentrations at stations 4-7 several months after the spillway was closed.

At station 4 in the mid lake (Fig. 5), NO_2+NO_3 and TP concentration reached the levels in the river but a week or two later than at station 2. Salinity declined more slowly and never reached completely fresh conditions. As with stations 2 and 3, nutrient concentrations returned to preopening levels by mid June while salinity did not return to preopening

Table 1. Results of statistical analysis comparing mean concentrations from April 1 through May 5 to concentrations from July 28 through September 22. Arrows denote if concentrations increased (↑) or decreased (↓) during the spillway opening. NS-no significant difference; * a<0.05; ** a<0.01; * a<0.001; **** a<0.0001.**

Station	NO_2+NO_3	NH_4	TKN	ON	TN	TP	TSS	Salinity
2	** ↑	NS	NS	* ↑	** ↑	** ↑	* ↑	** ↓
3	** ↑	NS	NS	NS	** ↑	** ↑	NS	** ↓
4	** ↑	NS	NS	NS	** ↑	** ↑	* ↑	** ↓
5	** ↑	NS	NS	NS	* ↑	NS	NS	NS
6	NS	NS	NS	NS	NS	NS	NS	NS
7	** ↑	NS	NS	NS	** ↑	** ↑	* ↑	** ↓

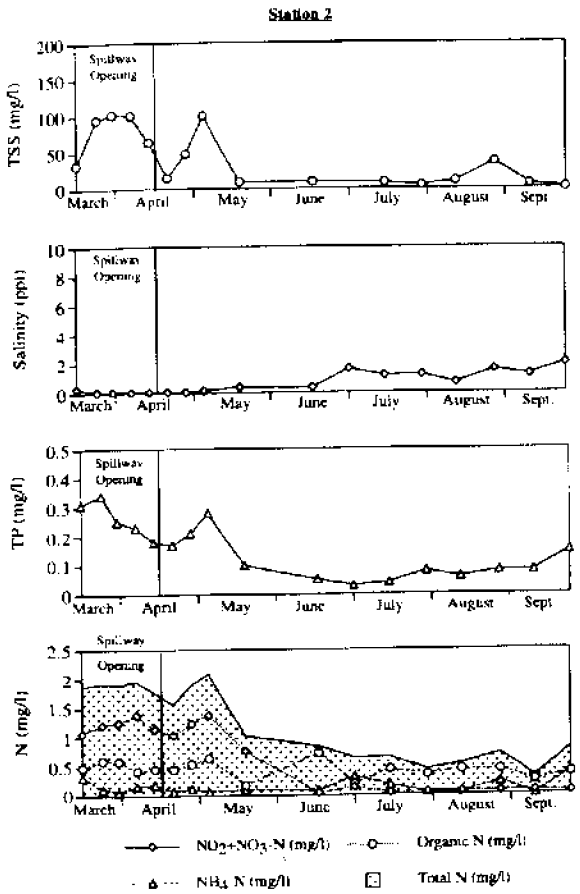

Fig. 3. Water quality data at Station 2 during and after the Bonnet Carré Spillway opening.

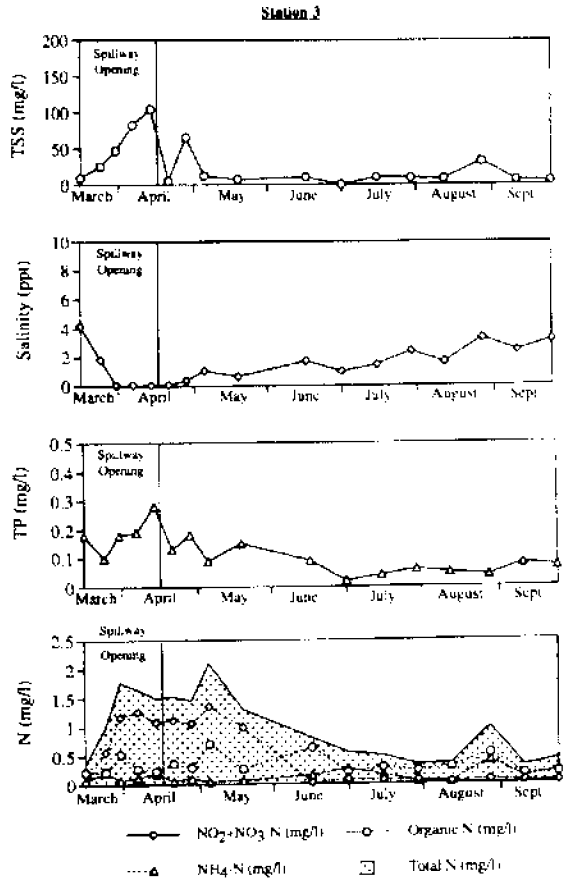

Fig. 4. Water quality data at Station 3 during and after the Bonnet Carré Spillway opening.

levels until September. Station 5, near the north shore, was similar to station 4, but nutrient concentrations were lower and salinity was not significantly changed due to the diversion (Figs. 5-6, Table 1). NO_2+NO_3 levels were significantly higher during the spillway opening, but never reached the concentrations found in the river while TP did so only for a short period in late April and early May. Salinity was near fresh for an extended period from May to early July and had not returned to preopening levels by late September. This is likely due to discharge from north shore rivers.

There was less influence of river water at station 6 in the northeastern portion of the lake (Fig. 7) than at station 7 in Lake Borgne (Fig. 8). Station 6 appeared less affected by the spillway opening and there was no significant temporal differences between nutrient, TSS or salinity concentrations. NO_2+NO_3 and TP concentrations at station 6 were generally less than half of that in the river, while concentrations at station 7 were close to river water for much of April and early May. NO_2+NO_3 and TP concentrations returned to preopening levels by mid June at both stations. Salinity was near fresh at station 7 for about two weeks in mid April while salinity was higher at station 6 during the same period. The salinity and nutrient data suggest that river water moved preferentially along the south shore of the lake and most of the time flowed out of the lake through Chef Menteur Pass.

Results of the spatial statistical analysis indicate significant differences between stations for NO_2+NO_3, ON, TN and salinity during the spillway opening, but during late summer there were no significant differences between stations, except for salinity (Tables 2 and 3). During the spillway

opening, stations 5 and 6 had lower NO_2+NO_3 and TN concentrations than all of the other stations (Tables 2a and 2c). Organic nitrogen was higher at station 2 than stations 4, 5, and 6, station 3 had higher ON than station 5, and station 7 had higher ON than station 5 (Table 2b). There were no significant differences in salinity between stations, except station 6 had higher salinity than station 2 (Table 2d). In the late summer period, there were higher salinities at station 7 than any other station (Table 3).

DISCUSSION

Lake Circulation

The circulation in the lake is affected by freshwater input, winds, Coriolis force, and the several tidal inlets. Mean riverine input to the Lake Pontchartrain system is about 370 m^3s^{-1} (Bianchi and Argyrou 1997) while the maximum transport at the tidal inlets is about 6400 m^3s^{-1} (Swenson and Chuang 1983). Peak spillway discharge was about 6800 m^3s^{-1}, thus making it equivalent to maximum tidal transport in the inlets. Thus, while the spillway was open, freshwater input to the lake was dominated by Mississippi River water (approximately 95%). During the opening, freshwater flowed preferentially along the south shore of the lake and out Chef Menteur Pass as suggested by the higher salinities at stations 5 and 6. Transport through the three tidal inlets is not balanced. The Rigolets is the largest inlet and carries about 60% of the discharge of the three inlets compared to 30% for Chef Menteur Pass and 10% for IHNC. The Rigolets is flood dominated with

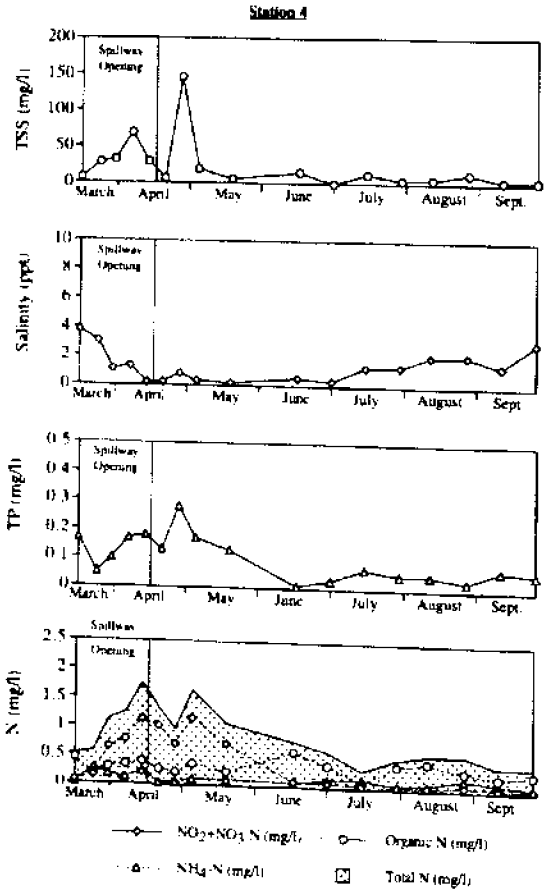

Fig. 5. Water quality data at Station 4 during and after the Bonnet Carré Spillway opening.

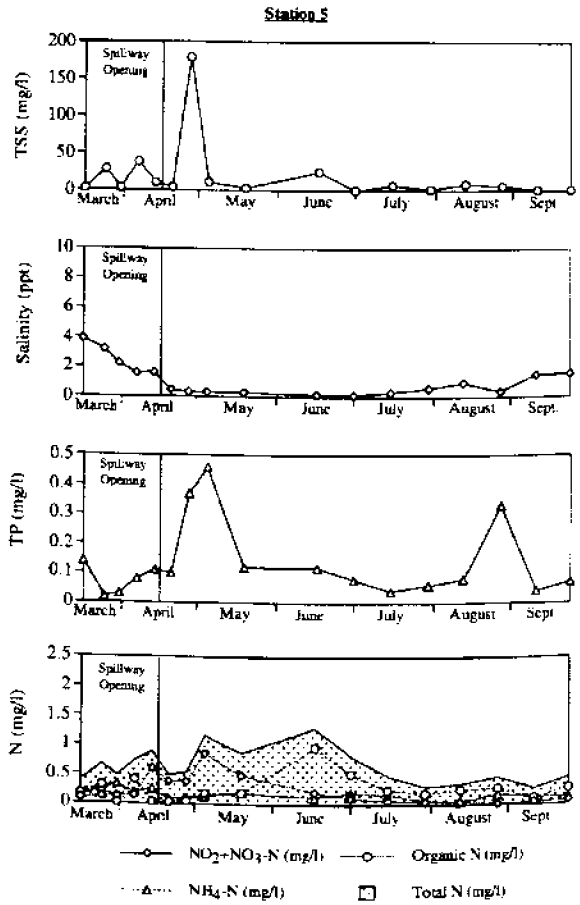

Fig. 6. Water quality data at Station 5 during and after the Bonnet Carré Spillway opening.

flood transport 27% greater than ebb transport, while the other two inlets are ebb dominated (ebb transport is 37% greater for Chef Menteur and 28% greater for IHNC, Swenson and Chuang 1983). These circulation characteristics help to explain the water quality patterns in the lake.

Algal bloom

Following the closure of the spillway there was an extensive blue-green algal bloom, predominantly *Anabaena circinalis* and *Microcystis aeruginosa*, in Lake Pontchartrain from late May which persisted through July (Dortch et al. 1998; Poirrier and King 1998). Both of these genera are known to be stimulated by excess nutrients and are capable of positive buoyancy, which allows them to avoid light limitation in turbid waters (Dortch et al. 1998). Fish kills attributed to the bloom were reported during June and July, during which time algal cell counts were as high as 10^{10} cells per liter (Poirrier and King 1998). There is evidence that the algae were taking up ON early in the bloom and fixing nitrogen later during the bloom (Dortch et al. 1998). There were increased ON levels in the central and northern portions of the lake when the algal bloom was observed. Paerl et al. (1998) found algal blooms in the Neuse River Estuary when high freshwater discharge events were followed by relatively low discharge periods, presumably due to increased retention time which allowed phytoplankton biomass to accumulate rather than to be flushed out of the system.

The introduction of freshwater into estuaries has broad affects on phytoplankton productivity.

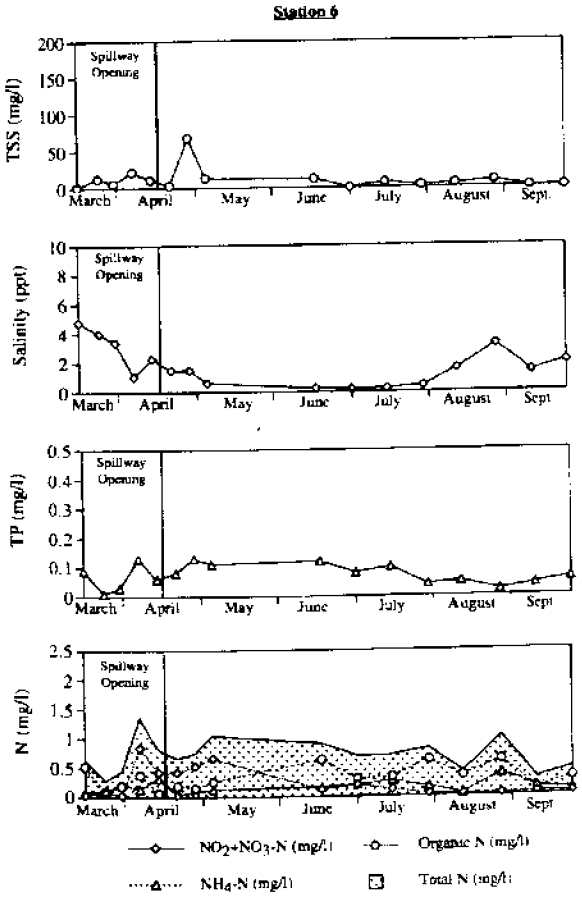

Fig. 7. Water quality data at Station 6 during and after the Bonnet Carré Spillway opening.

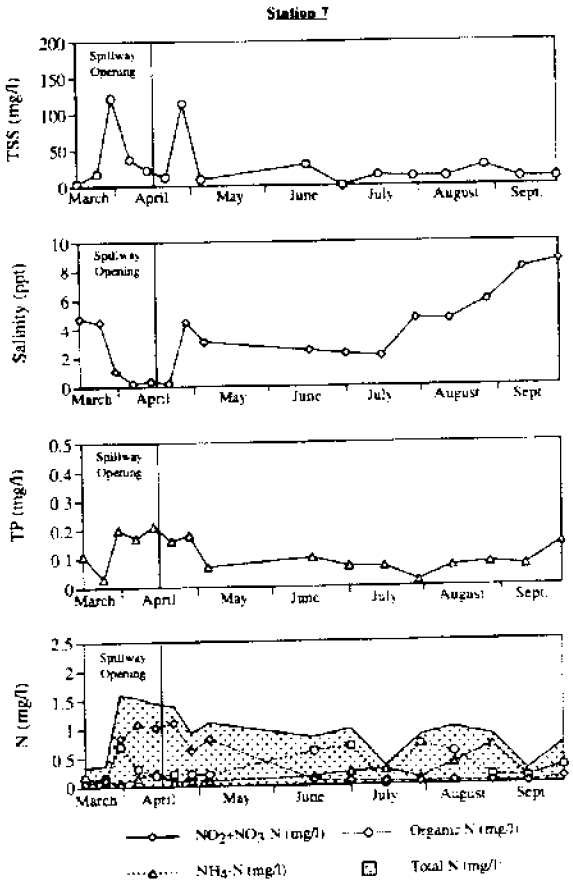

Fig. 8. Water quality data at Station 7 during and after the Bonnet Carré Spillway opening.

Table 2. Post-ANOVA results of statistical analysis comparing mean concentrations from April 1 through May 5 between stations. Greater than (>) or less than (<) symbols indicate significant differences between stations listed vertically compared to stations listed horizontally. A. NO_2+NO_3; B. Organic Nitrogen; C. Total Nitrogen; D. Salinity.

Table 2A

Station	2	3	4	5	6
3	NS				
4	NS	NS			
5	<	<	<		
6	<	<	<	NS	
7	NS	NS	NS	>	>

Table 2B

Station	2	3	4	5	6
3	NS				
4	<	NS			
5	<	<	NS		
6	<	NS	NS	NS	
7	NS	NS	NS	>	NS

Table 2C

Station	2	3	4	5	6
3	NS				
4	<	NS			
5	<	<	<		
6	<	<	<	NS	
7	NS	NS	NS	>	>

Table 2D

Station	2	3	4	5	6
3	NS				
4	NS	NS			
5	NS	NS	NS		
6	>	NS	NS	NS	
7	NS	NS	NS	NS	NS

Table 3. Post-ANOVA results of statistical analysis comparing mean salinity concentrations from July 28 through September 22 between stations. Greater than (>) or less than (<) symbols indicate significant differences between stations listed vertically compared to stations listed horizontally.

Station	2	3	4	5	6
3	NS				
4	NS	NS			
5	NS	NS	NS		
6	NS	NS	NS	NS	
7	>	>	>	>	>

High primary productivity in estuaries receiving freshwater has been related to the introduction of nutrients (Nixon 1981), but production is also limited by light availability that is attenuated by high suspended sediment concentrations usually associated with freshwater inputs (Cole and Cloern 1984). In river-dominated estuaries, environments with turbidity often exceeding 50 mg l^{-1}, light is attenuated rapidly in the water column and phytoplankton photosynthesis is confined to a shallow photic zone. For this reason, phytoplankton productivity in turbid estuaries is often higher in the coastal ocean, adjacent to estuaries, where suspended sediment has dropped out of the water column yet high nutrient concentrations are still available (Cloern 1987). The high chlorophyll *a* concentrations after the spillway was closed was probably due to this effect, with lower sediment concentrations in the lake due to settling, yet high residual nutrient availability.

Spatial and Temporal Patterns of Water Quality Parameters

The results of this analysis showed that the lake was a sink for nitrate because nitrate decreased much more rapidly than salinity increased. Various studies have reported similar reductions in estuarine environments with a significant part of the reduction due to denitrification (Khalid and Patrick 1988; Lindau and DeLaune 1991; Nowicki et al. 1997). Jenkins and Kemp (1984) reported that up to 50% of NO_2+NO_3 introduced into the Patuxent River estuary underwent denitrification.

Another transformation of NO_2+NO_3 is assimilation into particulate organic matter by phytoplankton. Both denitrification and phytoplankton uptake were likely significant processes in the lake. Vascular plant uptake can also remove NO_2+NO_3 but this was probably not a significant process because the vegetation:open water ratio of this system is low. NO_3 reduction to NH_4 has also been found to occur (Smith et al. 1982), with as much as 50% of NO_3 applied to marine sediments being reduced to NH_4 (Sorenson 1978).

Ammonium levels increased in Lake Pontchartrain. Lane et al. (1999) reported similar results for the Caernarvon freshwater diversion site located further south on the Mississippi. This was most likely caused by the regeneration of NH_4 by the decomposition of organic matter (Kemp and Boynton 1984), as well as reduction of NO_2+NO_3 to NH_4 (Sorenson 1978). Numerous studies have shown the net mobilization of NH_4 by benthic sediments (Koike and Hattori 1978; Callender and Hammond 1982; Teague et al. 1988). The relatively shallow water depths, rapid settling rates and rapid bacterial utilization result in fairly short residence times for organic material in estuarine waters (Moran and Hodson 1989). Therefore, much of the regeneration of nutrients probably takes place on or in the sediments, which is where NH_4 regeneration is highest (Blackburn 1979).

Total nitrogen concentrations decreased in the lake after the spillway was closed, suggesting that Lake Pontchartrain acted as a sink for inorganic nitrogen. TN is a combination of all the forms of nitrogen discussed above, and its behavior is governed by processes controlling those constituents. Dilution, however, could have been responsible for some TN reduction. Denitrification plays a major role in the loss of nitrate from estuarine waters, but another permanent loss of nitrogen is through burial of organic material such as detritus and senescent phytoplankton cells. ON concentrations increased in the lake several weeks after the closure of the spillway, largely as a result of the algal bloom, a portion of which undoubtedly settled to the bottom of the lake.

The results indicate that most stations in Lake Pontchartrain had increased levels of TP during and directly after the diversion, but levels decreased during the summer months. The highly charged phosphate anion, PO_4, is readily sorbed onto the surfaces of charged clay (and detrital organic) particles at high concentrations, while at lower concentrations PO_4 is released into the water, thus maintaining moderate ambient concentrations (Jitts 1959). Cyclic aerobic and anaerobic conditions in the sediments also effect the sorption and release of phosphate (Patrick and Khalid 1974). Sharp et al. (1982) found these sorption-desorption processes provide a buffering mechanism for phosphorus in the Delaware estuary, and Madden et al. (1988) showed that TP behaved similarly in Fourleague Bay, Louisiana, with little change in concentration throughout the year. The same processes, as well as algal uptake, most likely affects TP concentrations in the lake.

Suspended sediments were rapidly trapped in the estuary. This was likely due to decreasing water velocity when entering the estuary, allowing suspended sediment to drop out of the water column. Lane et al. (1999) reported similar findings for the Caernarvon diversion and Villarrubia (1998) reported 164 ha of new marsh has formed in Breton Sound estuary since the opening of the diversion, presumably due to sediment accumulation.

Flint and Rabalais (1981) found the periodic additions of freshwater into Corpus Cristi Bay had a beneficial affect on ecosystem functioning. They suggest that higher phytoplankton productivity associated with increased nutrient input is assimilated into benthic biomass. Increased benthic production represents additional food supply for important fisheries such as shrimp. Increased production of oysters, finfishes and penaid shrimp have been attributed to previous openings of the Bonnet Carré Spillway (Chew and Cali 1981). Caddy (1993) proposed a model for the effect of increasing nutrient inputs on various trophic levels in large enclosed water bodies. Additions of nutrients initially increases overall production for all trophic levels, with peak rates occurring when the water body is slightly eutrophic. As the system becomes more eutrophic the benthic community declines rapidly in response to bottom water anoxia. This trend is followed by other trophic levels and leads to eventual system collapse. Where Lake

Pontchartrain is on this scale is uncertain, but future studies should address this question so the health and trophic functioning of the lake can be properly managed.

CONCLUSIONS

The diversion of Mississippi River water into Lake Pontchartrain during the spring of 1997 had a profound effect on Lake Pontchartrain water quality during, and several months after, the diversion ended. There were increases in nitrogen and phosphorus concentrations in the lake associated with the diversion, which triggered a blue/green algal bloom that persisted for several weeks. The lake system did, however, assimilate a significant portion of the nutrient load either through algal uptake, microbial processes, or burial. Though the effects of the diversion on fisheries and wildlife populations in Lake Pontchartrain are not yet clear, the results from this study indicate only a short-term impact on water quality in the lake. Although the 1997 Spillway opening was for flood control purposes, the results of this study have implications for freshwater diversions. The lake rapidly assimilated much of the introduced suspended sediments and nutrients, but there was an intense algal bloom. Results by Lane et al. (1999) and Villarrubia(1998) for the Caernaervon diversion indicate that when the river is diverted into a marsh dominated estuary, there is wetland enhancement and rapid reduction in nutrient levels without intense algal blooms.

ACKNOWLEDGMENTS

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Sediment-Water Material Fluxes in Benthic Microalgal-Dominated Sediments from Beds of Submersed Vegetation: Impacts of Light Reduction and Nutrient Enrichment

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ABSTRACT

Benthic microalgal communities, including those within beds of submersed vegetation, are important contributors to primary production, water quality, and oxygenation of the water column. The impact of eutrophication on these communities has received little study. We investigated the effects of light reduction and nutrient enrichment with ammonium (NH_4), phosphate (PO_4), and nitrate+nitrite (NO_x) on the exchanges of oxygen (DO), dissolved inorganic carbon (DIC), NH_4, PO_4, and NO_x. Cores were collected in July, 1995 from Lake Pontchartrain, LA, from bare sediments within a bed of *Vallisneria americana*. After acclimation, one set of cores was used for determination of initial ambient flux rates. The remaining cores were placed in 15 cm-diameter pots in greenhouse tanks and grown under three light levels (100%, 60% and 40% of ambient) and two nutrient levels (ambient, and enriched at 3-6X ambient).

In pre-treatment July samples, mean net production and respiration were high, 89 and 71 mg O_2 m^{-2} h^{-1}, respectively. Ammonium was released in the light and dark but light fluxes were only 47% of dark rates, a significant difference (68 and 146 Fmol m^{-2} h^{-1}, respectively). Phosphate and NO_x were released from the sediments at low and variable rates under both light and dark conditions, means < 40 μmol m^{-2} h^{-1}.

In September, cores grown under nutrient enriched conditions had significantly higher net production (1.7-2.8 x ambient) and chl. *a* concentrations (1.3 x ambient) but were unaffected by shading. Respiration was also significantly higher (1-3-1.4 x ambient) in enriched treatments, but was also higher in the high light treatment. No significant effects of treatments were found for any other material flux. Nutrient fluxes were extremely variable, with both uptake and release occurring among replicate cores within each experimental treatment. In general, sediments took up NH_4 in the dark and released it in the light, while PO_4 and DIC were taken up in the light and released in the dark. Nitrate was taken up in enriched cores, and in the dark, but released under ambient nutrient levels in the light. Benthic autotrophy was clearly stimulated by nutrient enrichment, with a coupled increase in sediment oxygen consumption. Light treatment effects may have been obscured through canopy formation by *Vallisneria* at the end of this experiment. The lack of impact on nutrient fluxes indicates that exchanges of these constituents are not as closely tied to either photosynthesis or aerobic respiration as in other systems.

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INTRODUCTION

Benthic microalgae are significant sources of organic matter and oxygen in estuaries (Sundback et al. 1991; Rizzo et al. 1992;). They can also reduce nutrient release by sediments or remove nutrients from the water column (Sundback and Graneli 1988; Rizzo 1990; Rizzo et al. 1992; Reay et al. 1995). Most estuaries have come under stress from eutrophication within recent decades. Eutrophication has had well documented effects on the autotrophic communities of phytoplankton and submersed macrophytes (Kemp et al. 1983; Lapointe and Clark 1993; Valiela et al. 1993; Doering et al. 1995), but there have been few studies of the benthic microalgal community. Inferences from the few existing field and microcosm studies have shown substantial and variable impacts on this community. In the Neuse River Estuary benthic microalgal production is lower in the oligohaline zone, near the major source of nutrient loading, than in the mesohaline zone, probably from greater light attenuation due to both high particulate concentrations and greater phytoplankton biomass (Rizzo et al. 1992). In coastal waters with intensive raft culture of filter-feeding bivalves, the resultant organic and nutrient loading increased microalgal biomass, probably by increasing water column transparency; however, organic loading ultimately promoted net system heterotrophy and an oxygen deficit for the sediments (Barranguet et al. 1994). Fertilizers applied to the sediments of intertidal saltmarshes increased benthic microalgal biomass in one study (Sullivan and Daiber 1975), but had no direct effect in a second study (Estrada et al. 1974). Experimental nutrient enrichment of microcosms has increased both productivity and biomass (Nilsson and Sundback 1991; Nilsson et al. 1991), and increased meiofauna biomass and grazing on bacteria (Nilsson et al. 1991). In another microcosm study, light reduction decreased microalgal productivity, but not biomass, while nutrient additions had no effect (Dailey 1995). Neither light nor nutrient treatments affected NH_4 fluxes (Dailey 1995). In conjunction with a mesocosm study of the effects of light reduction and nutrient enrichment (increased loading) on the growth of *Vallisneria americana*, we undertook this study to evaluate those effects on the productivity of the benthic microalgal component of this community, and to evaluate the effects on sediment-water nutrient exchange.

MATERIALS AND METHODS

Intact sediment plugs, 15 cm in diameter, were taken from the north shore of Lake Pontchartrain about 5 km east of the macrophyte collection site, on July 10, 1995. Cores were taken from bare sediments between clumps of submersed vegetation and placed in flower pots for transport to the greenhouse at ambient temperature. At the greenhouse, the pots were placed in the same mesocosm tanks containing pots with submersed macrophytes, which had been previously collected. The tanks were filled with a micronutrient solution (Smart and Barko 1985). Water depth was adjusted to achieve light levels typical of the depth at the collection site (1 m). Nutrients (NH_4, PO_4, and NO_3) were added daily in sufficient amounts to achieve typical ambient nutrient concentrations for this area of Lake Pontchartrain. Each week one-third of the water was replaced and the tanks were scraped and the water filtered with a diatomaceous earth pool-filtering system to limit periphyton and phytoplankton growth.

After a week of acclimation 18 randomly selected pots were used to make estimates of pre-treatment flux rates. Pots were subcored using 10 cm plexiglass cores (70 cm^2 surface area, 2.5 L volume). The cores were then filled with the micronutrient solution used in the mesocosm tanks and placed in aquaria maintained at ambient temperature (25-30 °C) and receiving saturating light (600-800 mmol m^{-2} s^{-1}). The cores were stirred throughout the experiment. Flux calculations were made from differences in initial and final concentrations of DO, DIC, NH_4, PO_4, and NOx for sequential 3 h light and 2 h dark incubations.

Dissolved oxygen was measured polarographically (Yellow Springs Instruments Model 55) and DIC was measured by infra-red gas analysis (Capni Con 4 Total Carbon Dioxide Analyzer). Automated analysis (Alpkem Flow Solution III autoanalyzer) was used for nutrient analyses. Sediment chlorophyll a was determined using a Turner 10AU fluorometer using the method of

Welschmeyer (1994) and the calibration procedures of Wetzel and Likens (1991).

On July 25, we began the experimental treatments and continued them for 2 months. An increase in nutrient loading was achieved by making 3-fold additions of NH_4 and PO_4 and a 6-fold addition of NO_3 above ambient concentrations. In the absence of historical data for Lake Pontchartrain, we based the nutrient additions on the documented level of nutrient enrichment which occurred in Galveston Bay, where there have been extensive losses of SAV (Stanley 1992). Nutrient concentrations were determined prior to making nutrient additions, and within 20 min following additions. Added inorganic nitrogen was removed between additions except in the lowest light treatment, where NO_3 concentrations increased during the experiment, reaching concentrations near 40 µM. Mean ambient concentrations ± standard deviation of NH_4, PO_4, and NOx were 0.92 ± 0.14 µM, 0.41 ± 0.05 µM, and 1.41 ± 0.45 µM, respectively. Mean treatment concentrations were 3.97 ± 1.52 µM, 1.50 ± 0.61 µM, and 11.60 ± 12.13 µM, respectively. Therefore enrichments averaged 4.3X for NH_4, 3.7X for PO_4 and 8.2X for NOx.

The light reduction treatments were based on Secchi disk monitoring data (Cormier et al. 1994), which we used to calculate the degree of light reduction in areas where SAV was formerly present in Lake Pontchartrain. These calculations suggested that impacts would occur at reductions of 40-60% of ambient.

Final flux rate and chl. a determinations were done from September 19-22, 1995. Incubation procedures were the same as for the intial flux measurements except that dark incubations were carried out overnight, i.e. 16 h. Final sample sizes were 9 cores for each treatment block. Two-way analyses of variance were carried out on the flux and chl. *a* data. Effects were considered significant at $F_{prob} \leq 0.05$.

RESULTS

Flux rates as a function of the experimental treatments are shown in Figs. 1-5 for oxygen, DIC, NH_4, PO_4, and NO_x, respectively, in addition to pre-treatment flux values. Results of the chl. a determinations are shown in Fig. 6. The results of the analyses of variance are presented in Table 1. Oxygen metabolism increased significantly as a result of nutrient addition, while light reduction had no effect. Net productivity in ambient nutrient treatments declined from pre-treatment values under all light levels, while nutrient-enriched treatments increased at all light levels. Oxygen uptake under enriched conditions was similar to pre-treatment uptake rates, while rates under ambient nutrient

Table 1. Results (F_{prob}) of two-way analysis of variance on flux rates and chlorophyll a.

Variable		Incubation Condition Light	Dark
Oxygen			
	Light	0.1692	0.0202
	Nutrients	0.0002	0.0001
	Interaction	0.2338	0.5430
DIC			
	Light	0.4152	0.5319
	Nutrients	0.0529	0.1408
	Interaction	0.3315	0.4338
Ammonium			
	Light	0.1222	0.7967
	Nutrients	0.8994	0.1566
	Interaction	0.3853	0.5889
Phosphate			
	Light	0.2823	0.5253
	Nutrients	0.8801	0.3773
	Interaction	0.2985	0.3780
Nitrate+nitrite			
	Light	0.1099	0.7666
	Nutrients	0.1900	0.0655
	Interaction	0.4426	0.2896
Chl. a			
	Light	0.2953	
	Nutrients	0.0102	
	Interaction	0.6848	

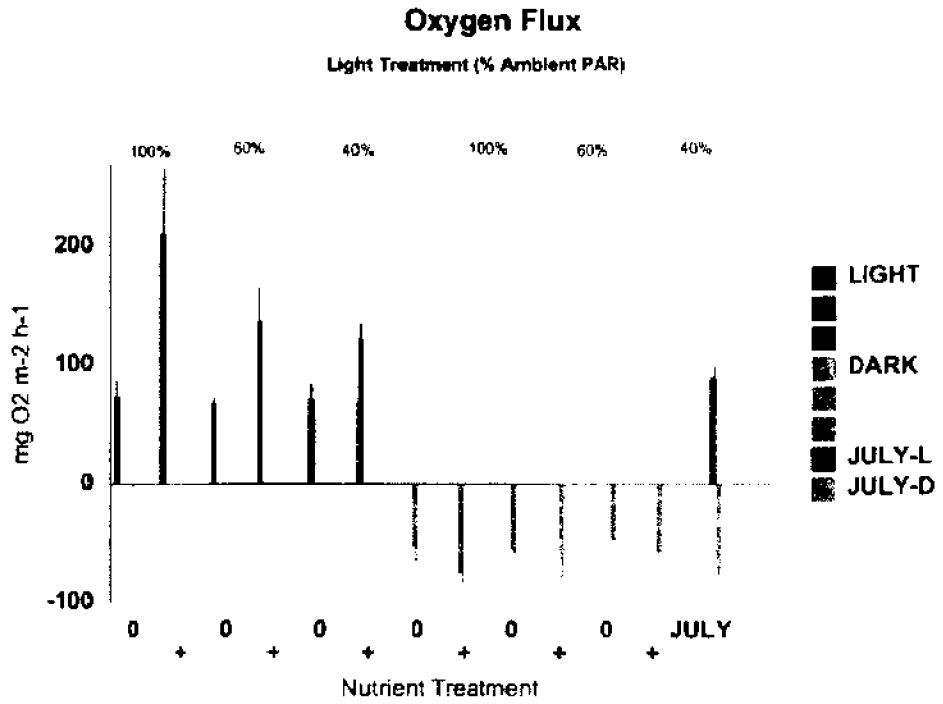

Fig. 1. Mean oxygen fluxes (mg O_2 m^{-2} h^{-1}) ± standard error of sediments for light and dark incubations. Labels on the x-axis designate tank treatments for non-enriched (0) and enriched (+) conditions, and for PAR (1 = 100%, 2 = 60%, 3 = 40% of ambient light). Fluxes into sediments are negative and fluxes out of sediments are positive. July denotes initial pre-treatment measurements.

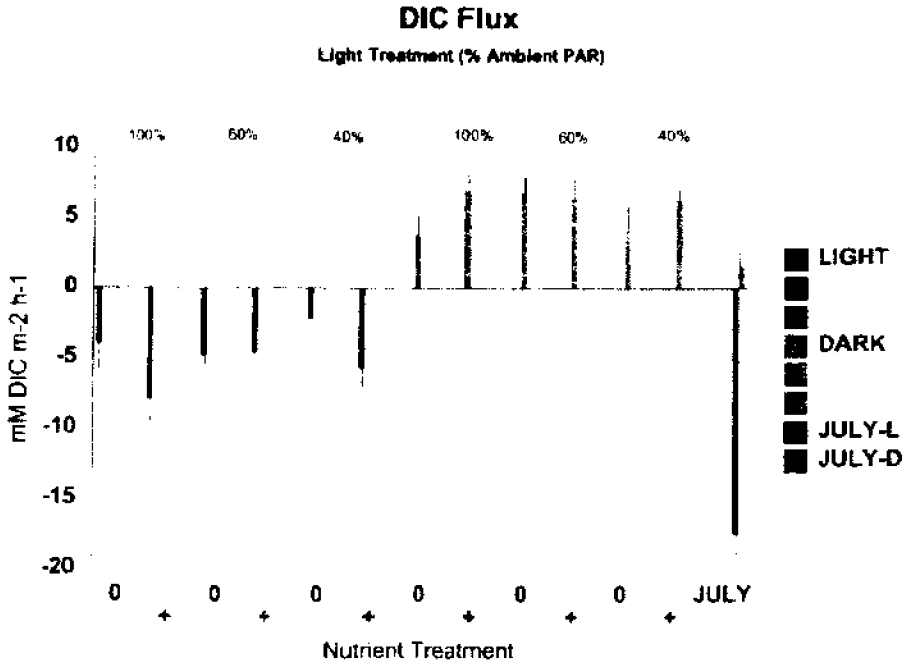

Fig. 2. Mean DIC fluxes (mmol m^{-2} h^{-1}) ± standard error of sediments for light and dark incubations. Labels on the x-axis designate tank treatments for non-enriched (0) and enriched (+) conditions, and for PAR (1 = 100%, 2 = 60%, 3 = 40% of ambient light). Fluxes into sediments are negative and fluxes out of sediments are positive. July denotes initial pre-treatment measurements.

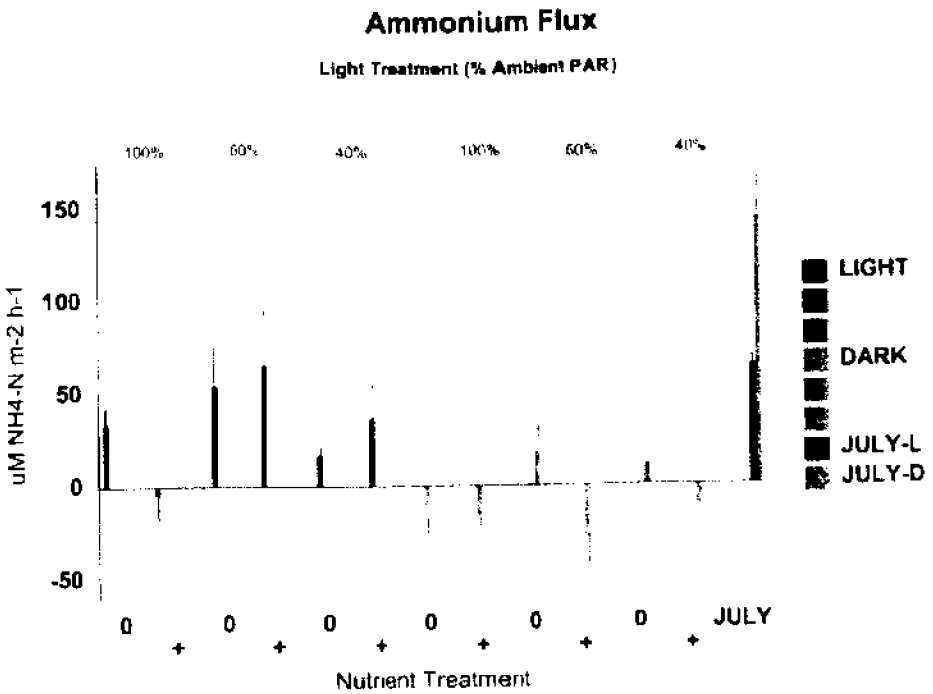

Fig. 3. Mean ammonium fluxes (μmol m^{-2} h^{-1}) ± standard error of sediments for light and dark incubations. Labels on the x-axis designate tank treatments for non-enriched (0) and enriched (+) conditions, and for PAR (1 = 100%, 2 = 60%, 3 = 40% of ambient light). Fluxes into sediments are negative and fluxes out of sediments are positive. July denotes initial pre-treatment measurements.

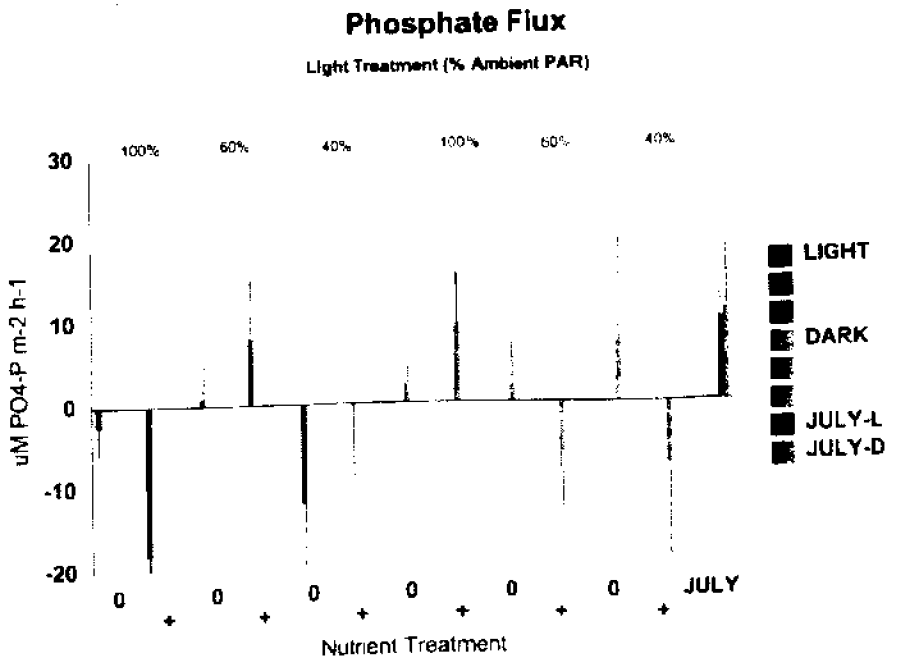

Fig. 4. Mean phosphate fluxes (μmol m^{-2} h^{-1}) ± standard error of sediments for light and dark incubations. Labels on the x-axis designate tank treatments for non-enriched (0) and enriched (+) conditions, and for PAR (1 = 100%, 2 = 60%, 3 = 40% of ambient light). Fluxes into sediments are negative and fluxes out of sediments are positive. July denotes initial pre-treatment measurements.

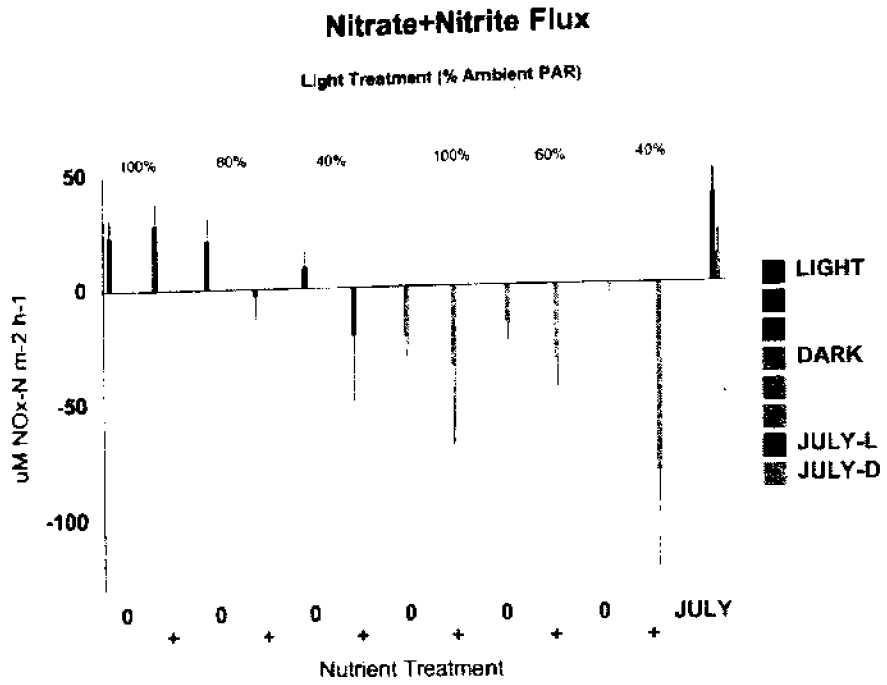

Fig. 5. Mean nitrate + nitrite fluxes (μmol m^{-2} h^{-1}) " standard error of sediments for light and dark incubations. Labels on the x-axis designate tank treatments for non-enriched (0) and enriched (+) conditions, and for PAR (1 = 100%, 2 = 60%, 3 = 40% of ambient light). Fluxes into sediments are negative and fluxes out of sediments are positive. July denotes initial pre-treatment measurements.

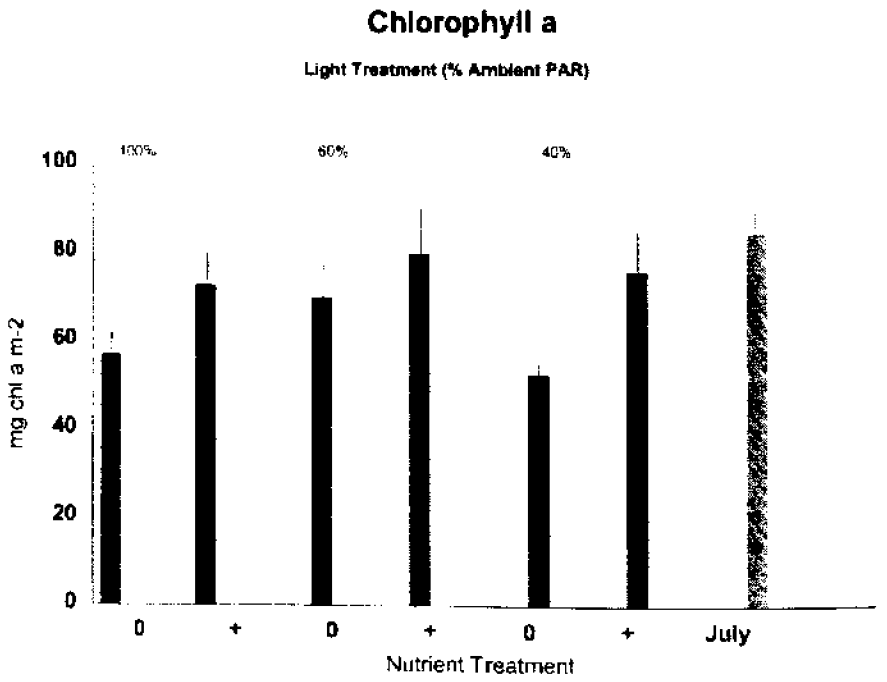

Fig. 6. Mean sediment chlorophyll a (mg m^{-2}) " standard error of sediments for light and dark incubations. Labels on the x-axis designate tank treatments for non-enriched (0) and enriched (+) conditions, and for PAR (1 = 100%, 2 = 60%, 3 = 40% of ambient light). July denotes initial pre-treatment measurements.

additions declined. Similarly, nutrient addition increased uptake of DIC in the light, but dark DIC release was not significantly affected even though releases were almost two-fold higher in the high and low light nutrient addition treatments. Pre-treatment uptake in the light was nearly 3-fold higher than any post-treatment value, while dark DIC release was less than half the value of any post-treatment flux.

There were no significant differences in any nutrient fluxes due to experimental treatments for either incubation condition. After treatments NH_4 was generally released in the light and taken up in the dark at fairly low rates. In contrast, pre-treatment fluxes showed high rates of NH_4 release in the dark, while release in the light was about half that in the dark. Fluxes of PO_4 and NO_x were low and erratic except for a high rate of NO_x uptake in the dark for the enriched low light treatment. Pre-treatment fluxes showed release of PO_4 in both the light and dark, but flux rates were still low. Under light incubation, NO_x was released at higher rates before treatments began, while dark incubations also showed release of NO_x in contrast to uptake under all post-treatment conditions.

Agreeing with the net production data, nutrient additions significantly increased chl. a concentrations but were unaffected by light reduction. Control concentrations ranged from 55 - 70 mg m^{-2}, while enriched treatment concentrations ranged from 72 - 80 mg m^{-2}, about 30% higher than mean control group concentrations, and similar to pre-treatment values.

DISCUSSION

Community Productivity

Our results clearly demonstrate an increase in benthic microalgal net production, chl. a biomass, and sediment oxygen uptake, the latter probably coupled to the increase in system autotrophy. The nutrient effect was not surprising considering the low ambient nutrient concentrations at this site (typically < 2 μM for all nutrients), and agree with Nilsson and Sundback (1991) and Nilsson et al. (1991). Dailey (1995) found a significant decrease in productivity with light reduction but no effect of nutrient enrichment. However, Dailey (1995) points out that ambient nutrient concentrations were probably sufficiently high in her study to alleviate nutrient limitation without enrichment. Our light reduction treatment for the benthic microalgal pots may have been negated by the extensive canopy formation achieved by the *Vallisneria* which was reaching peak biomass in all the mesocosm tanks. Since benthic microalgae respond rapidly to light reduction (Dailey 1995), even a week under full plant canopy may have nullified the effects of our light treatments. Unfortunately light levels were only determined for the water surface, so we cannot determine what actual light levels were reached at the sediment surface.

Nutrient enrichment significantly increased DIC uptake in the light, similar to the results of the oxygen fluxes, but the results for the dark incubations were not significant though the same trends were present as in the oxygen data. Stimulation of nitrification in dark-incubated cores could have acted antagonistically in reducing the increase in dark DIC efflux expected based on stimulation of oxygen uptake. Nitrification could have been increased by the direct addition of NH_4 substrate and by increased sediment oxygenation from the increase in benthic microalgal productivity.

Nutrient Fluxes

Pre-treatment NH_4 fluxes were similar to others reported for the literature, especially for heterotrophic (dark) conditions (Rizzo et al. 1996). Benthic microalgal production in the pre-treatment cores reduced release of NH_4 substantially (50%) over dark-incubated cores, but rates of release were still high compared to other studies which showed much greater reduction of NH_4 release, or reversal of flux direction (Nowicki and Nixon 1985; Sundback and Graneli 1988; Rizzo 1990; Rizzo et al. 1992; Reay et al. 1995). The high NH_4 release in the light also differs from the results of an ongoing study of spatial flux patterns on both the north and south shores of Lake Pontchartrain which have consistently shown removal of NH_4 with positive net production (Rizzo, unpublished data). Microbial response to an organic loading event may have

stimulated a heterotrophic response in excess of that supported by autotrophic metabolism, similar to the effects of bivalve culture (Barranguet et al. 1994). Such an event may have been caused by loading from houseboats and septic tanks within a few hundred meters of the collection site; or possibly as a response to senescence of a recent cyanobacterial bloom (John Burns, pers. comm.).

The post-treatment fluxes of NH_4 are much lower than those reported from other systems, especially for the prevailing temperature (ca. 30°C) and also differ from most flux studies in showing consistent NH_4 uptake in the dark (Kamp-Nielsen 1992; Vidal et al. 1992; Yoon and Benner 1992; Rizzo and Christian 1996). Since sediment oxygen consumption, and hence NH_4 regeneration was as high as in other studies showing NH_4 release (Kamp-Nielsen 1992; Vidal et al. 1992; Yoon and Benner 1992), uptake must have been due to an NH_4 demanding process which became dominant during the study. Nitrification is the most likely process to explain the dark uptake, since it has been shown to be very important in nitrogen fluxes in other estuaries (Risgaard-Petersen et al. 1994). The observed releases in the light could have been induced by sudden inhibition of nitrification by the high incubation light levels.

Fluxes of PO_4 were low (< 20 μmol m^{-2} h^{-1}) and erratic under all treatments and incubation conditions. Such findings are typical of both autotrophically- and heterotrophically-dominated sediments under oxic conditions (Rizzo and Christian 1996, and discussion therein). Ratios of N:P based on NH_4, are typically < 5 (Rizzo, unpublished data), indicating little liklihood of phosphate limitation, and little reason to suspect an autotrophic impact on PO_4 fluxes.

Similarly, NO_x fluxes were also generally low (< 20 μmol m^{-2} h^{-1}) and erratic as reported in other studies (Rizzo and Christian 1996, and discussion therein), but there were consistent incubation differences in flux direction. In the dark NO_x was taken up under all treatment conditions, while it was generally released in the light. The NO_x fluxes could be explained by coupling of denitrification to nitrification. An active denitrifying community would also be poised to remove NO_x from the overlying water-column. Light-inhibition of nitrification, discussed above, may have limited denitrification and resulted in release of NO_x in the light as well as NH_4. The high dark uptake and light uptake in the enriched low light treatment may have resulted from build-up of added NO_x in the water column. Concentrations were 17 μM in this treatment, compared to 1-4 μM in the other treatments, and may have afforded additional substrate for denitrification. Removal of NO_x by sediments is widely reported when water column concentrations are high (Boynton and Kemp 1985; Van Raaphorst et al. 1992; Rizzo and Christian 1996).

In summary, both microalgal net productivity and biomass, and coupled aerobic respiration were increased by nutrient enrichment in Lake Pontchartrain sediments. Further research would be desirable to determine if such a response would occur in the presence of competition from phytoplankton blooms. It is also interesting to note that despite substantial microalgal productivity and an apparently nitrogen-limited system, there was no significant effect on DIN fluxes for these sediments, or those supporting submersed macrophytes (Boustany and Rizzo, this volume). This points out the substantial contribution of other microbial processes to nutrient cycling in this system and underscores the need for studies of eutrophication effects on other microbial processes.

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Unpublished Materials

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Whole System Material Fluxes Within Meadows of *Vallisneria americana* from Lake Pontchartrain, LA: Effects of Light and Nutrient Manipulations

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ABSTRACT: The importance of SAV (submersed aquatic vegetation) communities to coastal and inland aquatic ecosystems is widely recognized, and dramatic loses of SAV habitat have been well documented throughout the entire Nothern Gulf of Mexico. Losses of SAV habitat have been primarily attributed to the adverse ecological effects of nutrient enrichment and light reduction induced by increased development of the coastal zone. Successful management and restoration will largely depend on understanding the response of SAV communities to widespread environmental changes. The objectives of this study were to determine the whole system (macrophytes, epiphytes, sediments) response of an SAV community to nutrient enrichment and light reduction on fluxes of dissolved oxygen, dissolved inorganic carbon, and nutrients. Intact sods of *Vallisneria americana* were collected from Lake Pontchartrain, LA, and grown in a greenhouse during the summer of 1995 under three light levels (100%, 60% and 40% of ambient) and two nutrient levels (ambient, and enriched at 3-6X ambient). In September, sods were subcored, and whole system (macrophytes + epiphytes + sediments) fluxes of dissolved oxygen (DO), dissolved inorganic carbon (DIC), ammonium (NH_4), phosphate (PO_4), and nitrate+nitrite (NO_x) were determined in light and dark incubations. Net production and respiration averaged 2148.17 mg O_2 m^{-2} h^{-1} and -915.67 mg O_2 m^{-2} h^{-1}, respectively (Negative values denote uptake). All other constituents, on average, were taken up in the light and released in the dark. Mean light/dark flux rates for DIC, NH_4, PO_4 and NO_x were -43.26/28.49 (mM m^{-2} h^{-1}), -66.07/75.50, -123.60/45.97, and -37.90/23.29 (μmol m^{-2} h^{-1}), respectively. Treatments produced significant differences in whole system net production. Whole system oxygen production was enhanced by nutrient additions under high and low light, but reduced at intermediate light. Variable epiphyte growth may have confounded some treatment effects.

INTRODUCTION

The importance of SAV (submersed aquatic vegetation) communities to coastal and inland aquatic ecosystems is widely recognized. The benefits of SAV communities include habitat and food source for many important commercial, recreational, and endangered species of fish and wildlife, sediment and shoreline stabilization through the baffling of wave and current energy, facilitation of detrital food webs, and amelioration of water quality by nutrient uptake and recycling and promotion of sedimentation. Therefore, changes in the distribution and abundance of SAV have widespread implications to fish and wildlife resources, shoreline geomorphology, and biogeochemical cycles.

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Extensive losses of SAV habitat have been well documented for many coastal areas of the United States. Seagrasses cover less than 10% of their original area in Galveston Bay (Pulich and White 1991), less than 30% in Mississippi Sound (Eleuterius 1987), less than 20% in Tampa Bay (Lewis et al. 1985), and only ca. 1% in Pensacola Bay (Livingston 1987). In contrast, the status of SAV species in inland estuarine and freshwater river systems and bays is very poorly known. Significant losses have been documented for Lake Pontchartrain (Steller 1985; Turner et al. 1980), and Mobile Bay (Stout 1990) while tremendous fluctuations of both biomass and species composition have been noted in Currituck Sound (Davis and Brinson 1983) and the tidal Potomac River (Rybicki and Carter 1986). In the Indian River lagoon losses of SAV coverage have been highly variable over the past 20 years, but in some areas losses have exceeded 95% (Morris and Tomasko 1993).

Most losses of SAV habitat can be attributed to effects of coastal zone population growth and accompanying municipal, industrial, and agricultural development (Neckles 1994). Although direct causes of local declines can sometimes be identified, the majority of habitat loss has been attributed to widespread chronic deterioration of water quality (Livingston 1987, Kenworthy and Haunert 1990). Increased concentrations of suspended sediments, dissolved inorganic matter, and dissolved nutrients in the water column promote poor light conditions and stimulate growth of light-absorbing algae both in the water column and attached to SAV leaf surfaces. Consequently, light transmission to the leaves of submerged macrophytes is less than that necessary for plants to achieve net photosynthesis, and the plants begin to die. Therefore, successful management, mitigation, and restoration efforts depend on understanding the environmental tolerances and requirements of SAV communities to these widespread environmental changes. The objectives of this study were to determine the whole system (macrophytes, epiphytes, sediments) responses of an estuarine SAV community to 3-6 fold nutrient enrichment and 40-60% light reduction, and to determine the effects of the nutrient enrichment and light reduction on system flux rates of dissolved oxygen (DO), dissolved inorganic carbon (DIC), ammonium (NH_4), ortho-phosphate (PO_4), and nitrate + nitrite (NO_x). In addition, we determined the response of the epiphyte community and the benthic microalgal community to the study variables (Rizzo and Boustany, this volume).

MATERIALS AND METHODS

Intact sediment plugs, 15 cm in diameter, containing *Vallisneria americana* were collected on June 1, 1995 from the north shore of Lake Pontchartrain adjacent to Big Branch National Wildlife Refuge. Plants were brought back to the greenhouse and placed in large mesocosm tanks. The tanks were filled with a micronutrient solution (Smart and Barko 1985). Water depth was adjusted to achieve light levels typical of the depth at the collection site (1 m). Nutrients (NH_4, PO_4, and NO_3) were added daily in sufficient amounts to maintain the ambient nutrient concentrations of the sampling area of Lake Pontchartrain. Each week one-third of the water was replaced and the tanks were scraped and the water filtered with a diatomaceous earth pool-filtering system to limit periphyton and phytoplankton growth.

On July 25, we began the experimental treatments and continued them for 2 months. Enriched tanks received daily 3-fold additions of NH_4 and PO_4 and a 6-fold addition of NO_3, based on the nutrient enrichment documented for Galveston Bay, which suffered extensive SAV losses in the 1970's (Stanley 1992). In the absence of historical data for Lake Pontchartrain, we used Galveston Bay as a comparable model of conditions that could have similar effects on Lake Pontchartrain SAV communities. Nutrient concentrations were determined prior to additions in all mesocosms and within 20 min following additions in all enriched tanks to monitor uptake and/or accumulation. We found that added dissolved inorganic nitrogen was being removed between additions, with the exception of the lowest light treatment, where NO_x concentrations increased during the experiment, reaching values near 40 μM. Mean ambient concentrations (± standard deviation) of NH_4, PO_4 and NO_x were 0.92 ± 0.14 μM, 0.41 ± 0.05 μM, and 1.41 ± 0.45 μM, respectively and mean treatment

concentrations were 3.97 ± 1.52, 1.50 ± 0.61, and 11.60 ± 12.13 μM, respectively. Therefore, actual enrichments averaged approximately 4.3X NH_4, 3.7X PO_4, and 8.2X NO_x of ambient. The light reduction treatments were based on secchi disk monitoring data (Cormier et al. 1994), which we used to calculate the degree of light reduction in areas where SAV was formerly present in Lake Pontchartrain. These analyses suggested impacts would occur at 40-60% light reduction.

Following two months of treatment, randomly selected pots were removed from the mesocosm tanks and subcored using 10 cm plexiglass cores [70 cm^2 surface area, 2.5 L volume]. The cores were then taken to the laboratory and filled with the same micronutrient solution used in the mesocosms. Cores were placed in a laboratory incubation system consisting of thermally controlled aquaria, receiving saturating light (600-800 umol m^{-2} sec^{-1}) at ambient mesocosm temperature (25-30 °C). The cores were stirred throughout the duration of the experiment. Total system flux measurements were determined from initial and final concentrations of DO, DIC, NH_4, PO_4 and NO_x for a sequential 3h light and 15h dark incubation. Afterwards, epiphytes were scraped from the plants and both the plants and epiphytes were dried and weighed for biomass determination.

Dissolved oxygen was measured polarographically (Yellow Springs Instruments Model 55) and DIC was measured by infra-red gas analysis (Capni Con 4 Total Carbon Dioxide Analyzer). Automated analysis (Alpkem Flow Solution III autoanalyzer) was used for the nutrient analyses. Two-way analysis of variance (ANOVA) was used to compare the effects of light, nutrient enrichment, and their interaction on fluxes. Results were considered significant at a 0.05 probability level.

RESULTS

Flux measurements

Net production (Fig. 1) ranged from 1635 to 3213.45 mg m^{-2} h^{-1} and respiration ranged from

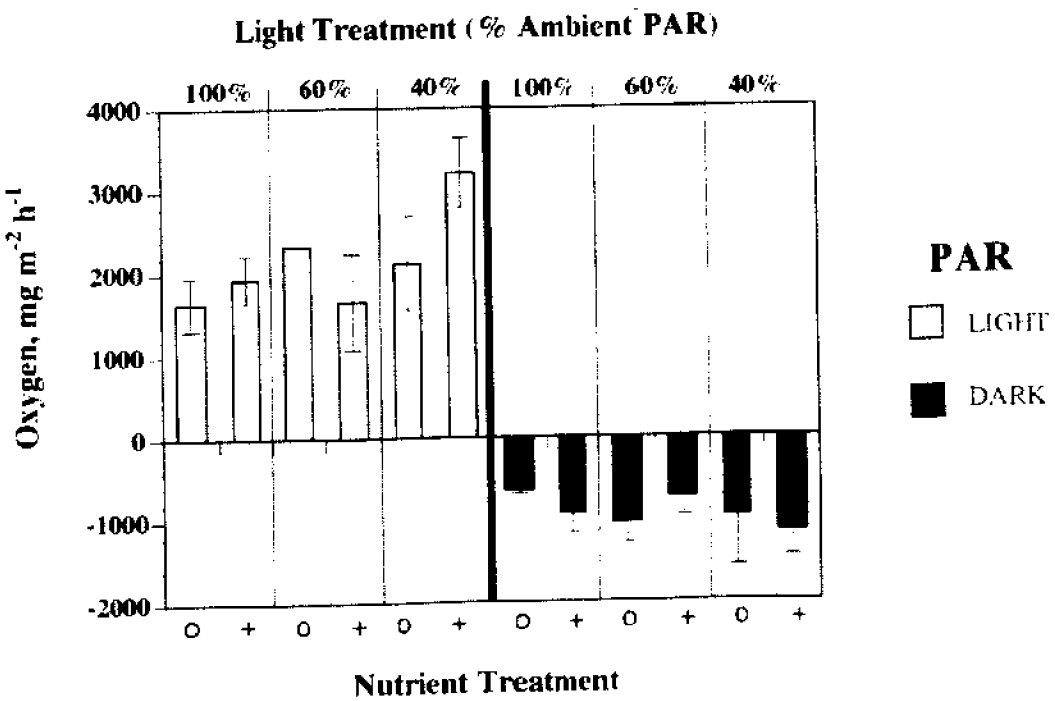

Fig. 1. Mean oxygen fluxes (mg O_2 m^{-2} h^{-1} ± standard error) of whole system for light and dark incubations. Labels designate tank treatments for ambient (o) and enriched (+) nutrients and light (100%, 60%, 40% of ambient light (PAR)). Fluxes into sediments are negative and fluxes out of sediments are positive.

-650 to -1162.88 mg m^{-2} h^{-1}. Surprisingly, the highest rates of primary production and respiration were recorded under the lowest light and highest nutrient treatment, while the lowest rates occurred under ambient (high) light and nutrient treatments. There was a significant interaction between light and nutrient treatments on dissolved oxygen fluxes in the light, but no significant effects under dark conditions (Table 1). In the light, nutrient addition increased photosynthesis under both high and low light, but not at the intermediate level of light reduction.

Dissolved inorganic carbon (DIC) fluxes (Fig. 2) ranged from -0.14 to -73.98 mM m^{-2} h^{-1} in light and 0.12 to 50.23 mM m^{-2} h^{-1} in dark. Consistent with the oxygen data, the highest fluxes for both light and dark experiments were observed under 60% light reduction and nutrient-rich conditions, and the lowest flux rates occurred under low light/ ambient nutrients treatments in both light and dark experiments. However, the treatment effects did not produce significant differences.

Table 1. Results (F prob) of two-way analysis of variance (ANOVA).

VARIABLE		WHOLE SYSTEM Light	Dark
Oxygen	Light	0.0014	0.3580
(mg $m^{-2}h^{-1}$)	Nutrients	0.2717	0.7259
	Interaction	0.0150	0.2819
DIC	Light	0.1426	0.1390
(mmol $m^{-2}h^{-1}$)	Nutrients	0.9693	0.7403
	Interaction	0.0942	0.3875
NH_4	Light	0.9489	0.5858
(umol $m^{-2}h^{-1}$)	Nutrients	0.8713	0.6920
	Interaction	0.6139	0.9331
PO_4	Light	0.8467	0.5223
(umol $m^{-2}h^{-1}$)	Nutrients	0.6838	0.6299
	Interaction	0.4780	0.5673
NH_4	Light	0.8467	0.5223
(umol $m^{-2}h^{-1}$)	Nutrients	0.6838	0.6299
	Interaction	0.4780	0.5673

Similarly, there were no significant effects of treatments on any nutrient fluxes. Generally nutrient fluxes were highest in the low light treatments. Ammonium and phosphate fluxes tended to be higher in enriched treatments, but NO_x fluxes were higher in ambient treatments. In the light, NH_4 (Fig. 3) was taken up at rates ranging from -21.92 to -138.83 µmol m^{-2} h^{-1}. In the dark, NH_4 was generally released from sediments, except for slight uptake in treatments receiving moderate (40%) light reduction and nutrient enrichment. Fluxes ranged from -9.13 to 137.91 µmol m^{-2} h^{-1}. The whole system removed PO_4 (Fig. 4) in the light at rates ranging from -44.75 to -233.81 µmol m^{-2} h^{-1}, and released it in the dark at rates of 11.87 to 126.04 µmol m^{-2} h^{-1}. The whole system removed NO_x (Fig. 5) in the light, at rates ranging from -15.52 to -62.11 µmol m^{-2} h^{-1}. Dark flux rates were more variable showing from slight uptake (-7.31 µmol m^{-2} h^{-1}) to fairly large releases (121.47 µmol m^{-2} h^{-1}).

Epiphyte biomass was not significantly different among treatments. Nevertheless, epiphyte biomass (Fig. 6) comprised a substantial percentage of total system aboveground biomass, 26-31% under high light. Under moderate light reduction and enriched conditions, epiphyte biomass contributed more (52%) to total system biomass than did *Vallisneria*. This may have been responsible for the inconsistent results of nutrient enrichment at the intermediate light level versus the significant enhancement of total system production at the other two light levels. Under ambient nutrient concentrations and moderate light, and under both high shading treatments, epiphytes contributed only about 10% to total system biomass. Including epiphyte biomass as a covariate did not change the results of the flux rate ANOVA's.

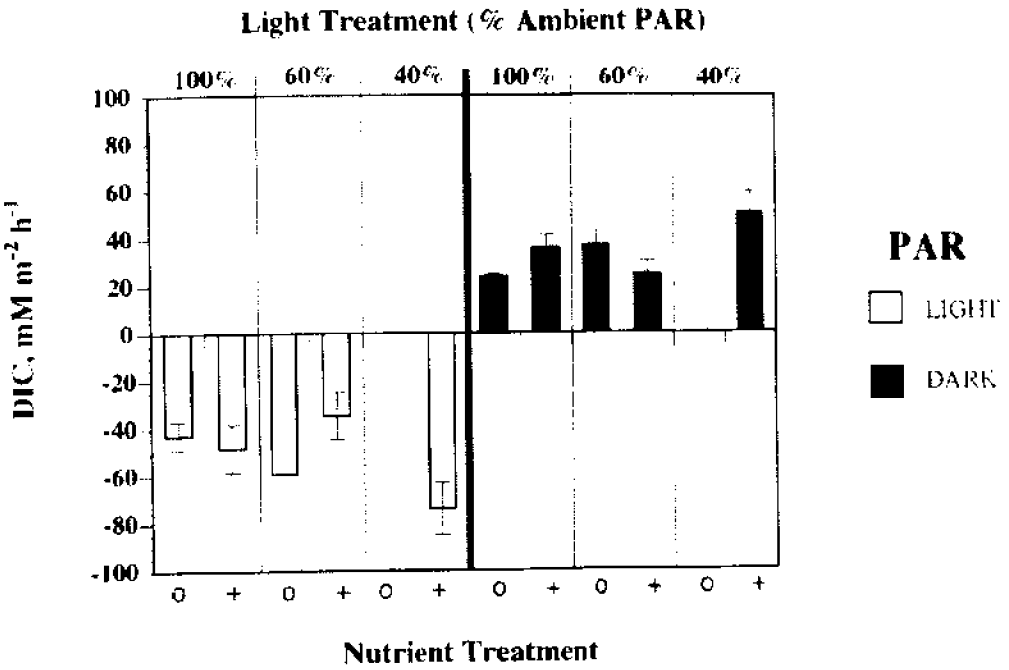

Fig. 2. Mean dissolved inorganic carbon (DIC) fluxes (mM DIC m^{-2} h^{-1} ± standard error) of whole system for light and dark incubations. Labels designate tank treatments for ambient (o) and enriched (+) nutrients and light (100%, 60%, 40% of ambient light (PAR)). Fluxes into sediments are negative and fluxes out of sediments are positive.

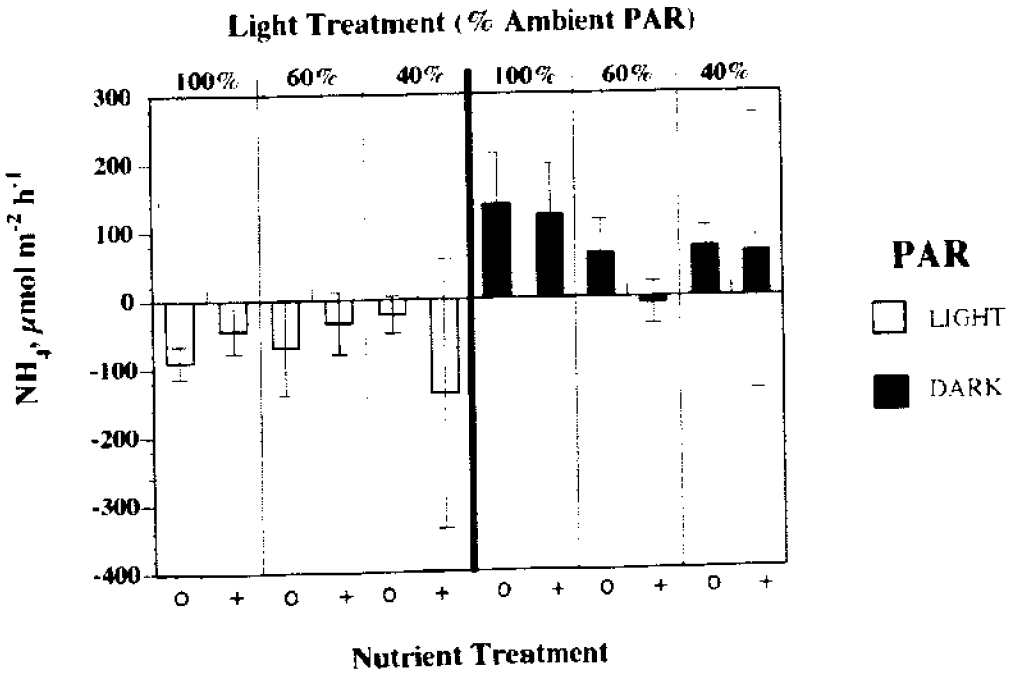

Fig. 3. Mean ammonium fluxes (µM NH_4 m^{-2} h^{-1} ± standard error) of whole system for light and dark incubations. Labels designate tank treatments for ambient (o) and enriched (+) nutrients and light (100%, 60%, 40% of ambient light (PAR)). Fluxes into sediments are negative and fluxes out of sediments are positive.

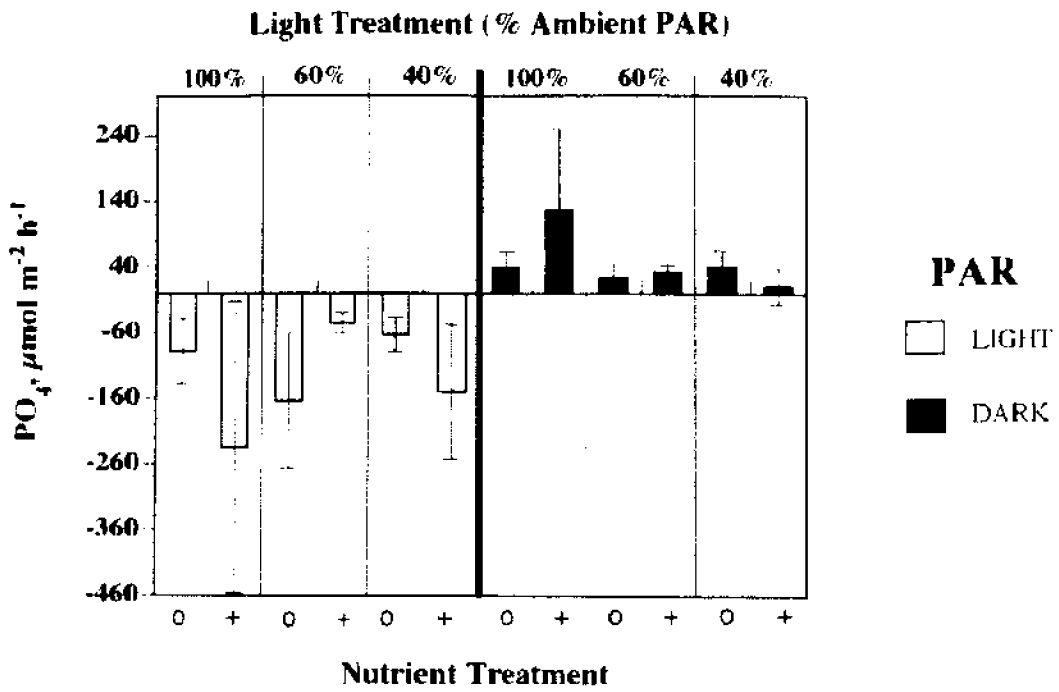

Fig. 4. Mean phosphate fluxes (µM PO_4 m^{-2} h^{-1} ± standard error) of whole system for light and dark incubations. Labels designate tank treatments for ambient (o) and enriched (+) nutrients and light (100%, 60%, 40% of ambient light (PAR)). Fluxes into sediments are negative and fluxes out of sediments are positive.

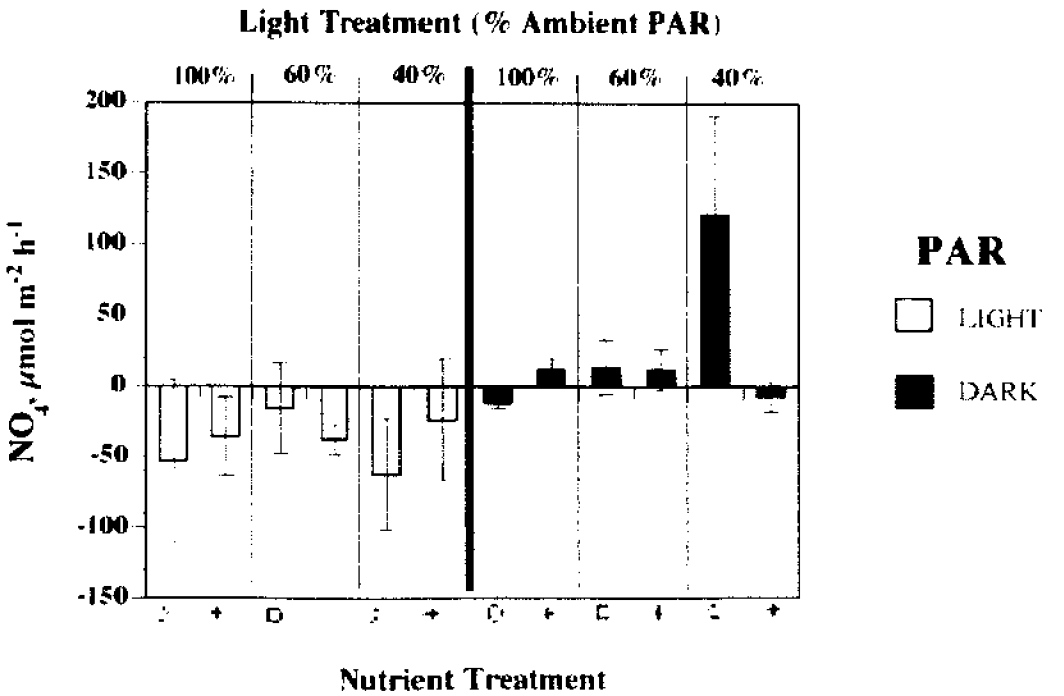

Fig. 5. Mean nitrate+nitrite fluxes (µM NO_x m^{-2} h^{-1} ± standard error) of whole system for light and dark incubations. Labels designate tank treatments for ambient (o) and enriched (+) nutrients and light (100%, 60%, 40% of ambient light (PAR)). Fluxes into sediments are negative and fluxes out of sediments are positive.

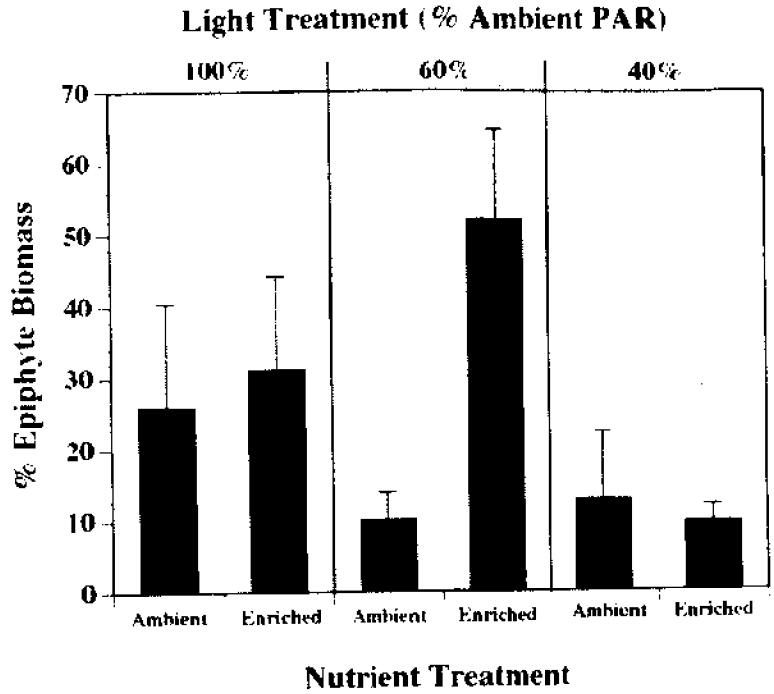

Fig. 6. Percent of total system aboveground biomass (± standard error) consisting of epiphytes for nutrient and light treatments.

DISCUSSION

Effects of Experimental Treatments

We expected increased productivity as a result of nutrient enrichment, as in the high and low light treatments, given the low ambient nutrient concentrations in this system. The depression of net production at the intermediate light level is probably due in part to the heavy epiphyte load. Macrophyte photosynthesis is frequently limited by DIC diffusion through the leaf-water boundary, and/or by competition with phytoplankton (Sand-Jensen and Borum 1991), effects which would be exacerbated by a thick epiphyte layer. Heavy fouling can also severely limit light penetration to the leaf surface (Twilley et al. 1985), an effect probably overridden in this experiment at the high light level. It is possible that actual light penetration to the leaf tissue in our enriched intermediate light treatment could have actually been lower than in the 60% reduction treatment, though we could not make that determination. Shading effects would have to be severe to induce light limitation in *Vallisneria*, which acclimates well to low light conditions (Twilley and Barko 1990). A number of earlier studies have shown significant increases in epiphyte biomass at high nutrient concentrations, to the detriment of the macrophyte hosts (Philips et al. 1978; Borum and Wium-Andersen 1980; Twilley et al. 1985; Jensen and Gibson 1986; Viaroli et al. 1996). If our *Vallisneria* plants were adapted to low light there may have been photoinhibition within the higher light mesocosm treatments as evidenced by the highest rate of production at the lowest light level under nutrient enrichment. The impairment of photosynthesis by super-saturating light intensities (photoinhibition) is well documented (Lund 1965; Talling 1971; Vymazal 1995). Given the high ambient light in the greenhouse, the shading levels may have acted to facilitate protection to plant photosystems rather than impose light limitation.

In such a strongly autotrophic system, the lack of treatment effects on nutrient fluxes was surprising. We expected to see some autotrophic impacts on nutrient fluxes as a result of both foliar uptake and forcing nutrients into the sediments by

creating a favorable concentration gradient by plant uptake in the root zone (Wigand et al. 1997). Although these processes are probably occurring, other microbial and physical processes apparently dominated material exchanges in this system under all experimental conditions. Secondarily, there may have been an epiphyte interaction, since nutrient fluxes, particularly in DIC, NH_4, and PO_4, were also consistently low in the intermediate light treatments (second only to the low light and low nutrient treatment), in both light and dark incubations. Finally, even in enriched treatments, nutrient concentrations were usually very low because of rapid uptake of the nutrient additions and were very often reduced to near detection limits by the time each subsequent addition was made. Consequently, replicate variability increases, and that fact coupled with variability induced by processes within the cores, probably contributed to such high variability in the calculated flux rates that significant differences could not be shown.

One obvious result of the impact of SAV autotrophy on material fluxes was evident in the magnitude of the fluxes. Fluxes of all constituents were an order of magnitude greater than fluxes from unvegetated sediments (Rizzo and Boustany, this volume). Presumably these higher flux values reflect the high metabolic capacity of the system, a combination of macrophytic productivity and coupled heterotrophic response, and possibly enhanced exchanges from the presence of a substantial rhizosphere.

Comparison of Material Fluxes with Other Studies

The relative contributions of primary producers to the metabolism of SAV communities are shown in Table 2. These studies encompass huge differences in scale, species and environment, and consequently show great variability among communities of SAV. Generally, sediments and epiphytes make a greater contribution to total system

Table 2. Relative contribution of autotrophic components to total system primary production (%).

	Scale	Plants	Epiphytes	Sediments	Phytoplankton
Mississippi Sound[1]	Annual	13	46	17	24
Indian River[2]	Sp/Su	72	5	21	2
Tampa Bay[3]	Quarter	17	15	10	58
Indian River[3]	Quarter	7	16	5	72
Dravuni Island, Figi[4].	Fall	38	38	25	--
Terminos Lagoon[5]	Annual	65*	--	5	30
Marion Lake[6]	Annual	24*	--	53	11
Ches. Bay - Zostera[7]	Annual	76*	--	15	9
Ches. Bay - Ruppia[7]	Annual	69*	--	26	5
Grevelingen - Zostera[8]	Annual	16*	--	24	38
Veerse Meer - Ulva[8]	Annual	27*	--	18	55
Present Study	Su	87	8	5	--

*Plants + Epiphytes

[1]Daehnick et al. 1992
[2]Heffernan & Gibson 1983
[3]Jensen & Gibson 1986
[4]Pollard & Kogure 1993
[5]Day et al. 1982
[6]Hargrave 1969
[7]Murray & Wetzel 1987
[8]Nienhuis 1993

productivity, up to 50%, than we found (< 10%). Since rates of benthic microalgal net production were as high as many other sediment microalgal systems (Rizzo et. al. 1996), our low value may have resulted from lack of inclusion of seasonality in our assessment. In our study, benthic production was compared to total system productivity only at the seasonal peak biomass of *Vallisneria*, and therefore represents a minimum contribution rather than a typical one. The particular growth and decomposition patterns of macrophyte species result in tremendous seasonal changes in the magnitude and direction of oxygen and nutrient fluxes (Viaroli et al. 1996).

The comparisons of nutrient fluxes (Table 3) are even more variable than oxygen fluxes and there are fewer studies. The variability of nutrient fluxes and relative contributions of autotrophic components are probably a result of differences between systems, especially regarding species of dominant macrophyte, and of the relative importance of other microbial processes within those systems. In particular, fluxes of NH_4 and NO_x are affected by nitrification and denitrification, which can be expected to vary strongly among systems in response to environmental differences such as oxygen concentration, salinity, light penetration into sediments, sulfide concentrations, and other variables (Risgaard-Petersen et al. 1994). While most studies show much greater whole system nutrient fluxes compared to unvegetated sediments (Table 2), there is often very little difference in NO_x fluxes between whole system and sediments (Falcao and Vale 1990, this study). This difference might be explained by a preference for ammonium as a nitrogen source for seagrasses (Moriarity and Boon 1989).

Table 3. Comparison of literature on relative contribution of autotrophic components to nutrient fluxes.

	Whole System*	Sediments*	Nutrient
Indian River[1] - Spring	0.04 to 0.09	0.004 to 0.03 (10-33%)	PO_4
Ria Formosa[2] - Annual	522 to 761	-58 to 121 (max. 16-23%)	SiO_2
	47 to 96	-5 to 25 (max. 26-398%)	PO_4
	-44 to 96	-25 to -175 (26-398%)	NO_x
	906 to 911	0 to 566 (0-62%)	NH_4
New Caledonia[3] - Summer	-17	-34 to 29 (max. 200%)	NH_4
This Study - Summer	-22 to 138 (light)	-4 to 66 (3-18%)	NH_4
	-9 to 138 (dark)	-28 to 18 (13-311%)	NH_4
	-45 to -234 (light)	-18 to 8 (8-40%)	PO_4
	12 to 126 (dark)	-8 to 10 (max. 8-83%)	PO_4
	-62 to -17 (light)	-3 to 29 (5-19%)	NO_x
	-7 to 121 (dark)	-80 to -1 (14-1143%)	NO_x

*All values µmol m-2 h-1
[1]Zimmerman et al. 1985. Theoretical diffusive flux. *Halodule wrightii*. Range @ 1 site.
[2]Falcao & Vale. 1990. *Zostera* sp. Range multiple sites.
[3]Boucher et al. 1994. Species not given. Mud sediments.

Although macrophytes may be the dominant structural feature in an SAV community, the relative contribution of macrophytes to total system production is highly variable. Summer primary production by *Vallisneria americana* from Lake Pontchartrain is dominated by the macrophyte component and is equivalent to well-developed seagrass systems (Dunton and Tomasko 1994). This system responded to nutrient enrichment but not to light reduction. The lack of significant treatment effects on nutrient fluxes suggests that nutrient fluxes in this system are not as strongly coupled to oxygen dynamics as in other systems (i.e., to oxygenic photosynthesis and aerobic respiration) and underscores the need to assess microbially-mediated processes in these systems as well. However, part of the observed responses may have been due to epiphyte/macrophyte interactions, underscoring the need to account for changes in all autotrophic components when evaluating the ultimate effects of eutrophication on these systems. For example, Jenson and Gibson (1986) compared three different SAV communities representing a range of nutrient regimes and varying degrees of human activity and impacts. By comparison to a study by Pomeroy (1960) two decades earlier, they found that in high nutrient environments there was a shift in the production base from seagrass dominance to phytoplankton. They also found that in two of the three species studied, the percentage of standing stock biomass consisting of epiphytes ranged from 1.9 to 2.5 times greater under high nutrient conditions than at the low nutrient site. Also, the relative contribution of seagrasses to total system production was diminished at the high nutrient sites. In our study, the occurrence of a higher percentage of epiphyte biomass as a result of nutrient enrichment may be indicative of a shift in the relative importance of the macrophyte component in the SAV community towards the algal component. Moreover, this may be symptomatic of changes in community structure as well as local nutrient dynamics. Early detection of such shifts along with prudent water quality information could serve as an effective tool for predicting impending SAV stress and decline. Therefore, a better understanding of the relative contribution of the different autotrophic components of this SAV community to material fluxes appears necessary to effectively predict the response of the community to changes in the light and nutrient environment.

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Effects of the 1997 Bonnet Carré Spillway Opening on Lake Pontchartrain Submersed Aquatic Vegetation

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ABSTRACT: An ongoing study of environmental factors that affect Lake Pontchartrain submersed aquatic vegetation provided information on the effects of the March 1997 Bonnet Carré Spillway opening. *Vallisneria americana* and *Ruppia maritima* abundance was surveyed using the line-intercept method at Pointe aux Herbes, Lacombe, Goose Point, and Fontainebleau State Park during late summer and fall of 1996 and 1997. A *Myriophyllum spicatum* bed near the mouth of Bayou St. John was also monitored. Temperature, salinity, dissolved oxygen, pH, and PAR were measured at most sites before, during, and after the Spillway opening. There was a significant decrease in PAR due to blue-green algal blooms. There was no change in *Vallisneria*, but significant decreases in *Ruppia* occurred between surveys. Abundant growth of the alga *Cladophora* occurred on *Ruppia* and *Myriophyllum*, but not on *Vallisneria*. *Cladophora* growth on *Ruppia* may have made it more susceptible to shading from phytoplankton and uprooting by wave energy. This could have resulted in *Ruppia* being selectively lost from grassbeds. Large surface mats of *Cladophora*, increased turbidity from blue-green algal blooms, and poor water quality from algal decomposition eliminated the *Myriophyllum* bed in Bayou St. John.

Introduction

The Bonnet Carré Spillway runs from the Mississippi River to Lake Pontchartrain (Fig. 1). It was designed to divert water from the River to prevent flooding. Spillway construction was completed in 1931, and the Spillway was opened in 1937, 1945, 1950, 1973, 1975, 1979, 1983, and 1997. Flow through the Spillway occurred during an experimental opening in 1994 and an unauthorized opening in 1995. During 1997, the Spillway was opened on March 17 and closure began on April 2. This study was originally designed to evaluate the effects of a breakwater on grassbeds (submersed aquatic vegetation or SAV), but it provided an opportunity to obtain information on the effects of the 1997 Spillway opening on SAV.

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Lake Pontchartrain is a large, shallow, estuarine embayment (Fig. 1) that has a mean salinity of 4 ppt, a mean depth of 3.7 m and a surface area of 1,630 km^2 (Sikora and Kjerfve 1985). The discharge of Mississippi River water through the Spillway produces short-term effects including freshwater conditions, increased turbidity and plant nutrients, and decreased water temperature. Higher nutrient loading produces long-term effects by increasing the growth of phytoplankton and epiphytic algae that reduces light availability to SAV (Dennison et al. 1993). Blooms of the blue-green algae *Anabaena* and *Microcystis* occurred after the 1997 Spillway opening. Blooms began in May, reached a maximum with cell counts of 10^8 l^{-1} in June, and declined in July (Dortch and Achee 1998). Turner et al. (1998) attributed these blooms to the introduction of nitrogen compounds that are high in Mississippi River water, but are usually low in Lake Pontchartrain. They reported Chlorophyll *a* values of 5 - 15 $\mu g\ l^{-1}$ in years without the diversion and values up to 800 $\mu g\ l^{-1}$ after the diversion.

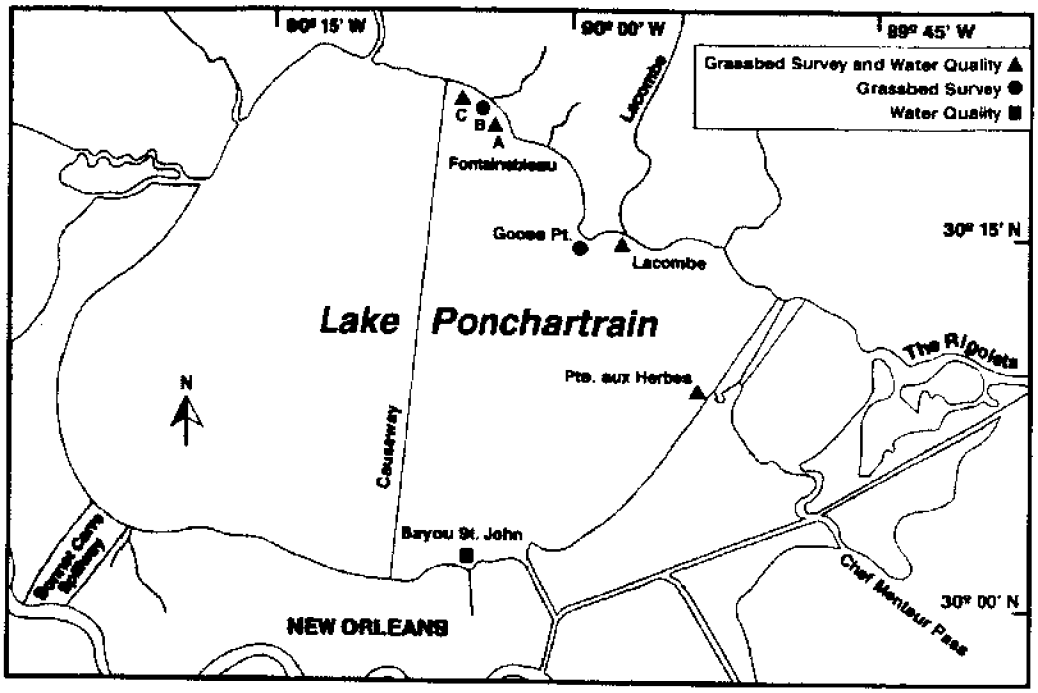

Fig. 1. Map of Lake Pontchartrain showing the seven SAV study sites.

Lake Pontchartrain SAV is dominated by *Vallisneria americana* Michx., a freshwater species that occurs in brackish water, and *Ruppia maritima* L., a brackish water species that occurs in freshwater (Davis and Brinson 1980). *Najas guadalupensis* (Spreng), *Myriophyllum spicatum* L., *Zannichellia palustris* L., *Potamogeton perfoliatus* L. and *Eleocharis parvula* (R.&S.) Link also occur, but are not abundant (Montz 1978). The focus of this report is on *Ruppia* and *Vallisneria*.

Lake Pontchartrain SAV has been in a state of decline since first studied by Suttkus et al. (1954). Vegetation coverage during 1953-1954 has decreased 25 to 35% between 1954 and 1973 (Turner et al. 1980), 50% between 1973 and 1984 (Mayer 1986), and 17% between 1985 and 1992 (Burns et al. 1993). Many explanations for the decline have been suggested (Mayer 1986), but modification of natural shorelines and degraded water quality are the principal causes (Burns et al. 1993).

Montz (1978) conducted a qualitative survey of SAV in Lake Pontchartrain before and after the 1973 Spillway opening. He did not find any observable effects on SAV. Burns and Poirrier (1996) reported that SAV near Fontainebleau State Park was lost during an algal bloom after the 1995 Spillway opening. Persistent blue-green algal blooms did not occur after the 1973 Spillway opening, but have occurred after more recent openings (Poirrier and King 1998). Burns and Poirrier (1996) expressed concern that River discharges might cause SAV decline from reduction of light needed for photosynthesis, and that higher pH values associated with algal growth might adversely affect *Vallisneria* by reducing free carbon dioxide.

Harlin (1995) presented data that supported a generalized shift in the biomass of major plant groups with increasing nutrient input in shallow marine systems. As nutrients increase, the abundance of macroalgae, especially ephemeral green taxa, increases, while SAV declines. *Cladophora* is an epiphyte on SAV in Lake Pontchartrain (Burns et al. 1993). Episodic blooms of this macroalga may be a factor in SAV decline.

Materials and Methods

Study Sites

Because the original study was designed to obtain baseline data on the effects of a proposed shoreline stabilization structure on SAV at Fontainebleau State Park, three study sites (A, B, and C) were established at the Park. Single reference sites were established at Pointe aux Herbes, Lacombe, and Goose Point. Another study site near the mouth of Bayou St. John was added to monitor a *Myriophyllum* bed in January 1997 (Fig. 1).

SAV

Species foliar cover was measured by the line-intercept method (Gertz 1984) with continuous observations along the transect. At each site, five randomly placed line transects were extended 200 m perpendicular from the shore. Water depths at which SAV occurred were recorded as one of five intervals: 0 - 0.31 m; 0.31 - 0.62 m; 0.62 - 0.93 m; 0.93 - 1.24 m; and 1.24 - 1.55 m (0 - 5 ft) on each transect. Field measurements were made from September through October of 1996 and 1997. The size of the Bayou St. John *Myriophyllum* bed was monitored during the winter and spring of 1997 by measuring surface cover.

Water Quality

Photosynthetically active radiation (PAR), pH, temperature, salinity, and dissolved oxygen were measured at Fontainebleau A and C, Lacombe, and Pointe aux Herbes. The following instruments were used: Li-Cor quantum sensors and photometer; an Oakton WD-35615 pH meter and; a model 85 YSI salinity, conductivity, temperature, and dissolved oxygen meter. Water quality was measured monthly from September 1996 to March 1997, weekly from March 20 through May, twice a month during June, July, and August, and monthly from September 1997 through December 1997. Sampling followed general recommendations in Lind (1974), Standard Methods (APHA 1992), and the operation manuals for each instrument. Measurements were made at a depth of 1 m between 9:00 A.M. and 3:00 P.M.

Data Analysis

The foliar coverage sum for each species on a transect was converted to a proportion by dividing the sum by transect length. For statistical analysis, proportions were transformed with the arcsine transformation to ensure normality of the data. Data were analyzed by three-way analysis of variance (ANOVA) and after-ANOVA unplanned contrasts. Residuals were analyzed to test for normality and homogeneous variances. A one-way ANOVA of PAR data was conducted to determine whether values after the Spillway opening (April 3 through July 10, 1997) were significantly different from values obtained at other times (September 1996 - March 1997 and July 24, 1997 - December 1997).

Epiphytes

Cladophora abundance on representative SAV samples was determined from Fontainebleau A and C, Lacombe, and Pointe aux Herbes on May 30, July 3, and September 3, 1997, from depths of 0.3, 0.6, and 0.9 m. Three categories were established: absent (no *Cladophora* observed under a dissecting microscope), present (*Cladophora* wet weight < 50% of wet weight of shoots and *Cladophora*), overgrowth (*Cladophora* wet weight > 50% of wet weight of shoots and *Cladophora*). *Cladophora* growth on *Myriophyllum* and surface growth were observed at Bayou St. John on at least a monthly basis from February through June 1997.

Results

Temperature, pH, and Dissolved Oxygen

Water temperature showed typical seasonal trends that corresponded to changes in air temperature. There was no decline in water temperature at the shallow study sites due to cooler Mississippi River water (Cormier et al. 1992) entering the Lake. Before the Spillway opening, pH varied from 7 to 8.5 and after the opening from 6.4 to 9.7 (Fig. 2). Low pH values were associated with runoff from acidic north shore streams and high values with phytoplankton photosynthesis. Dissolved oxygen ranged from 64 - 200% saturation

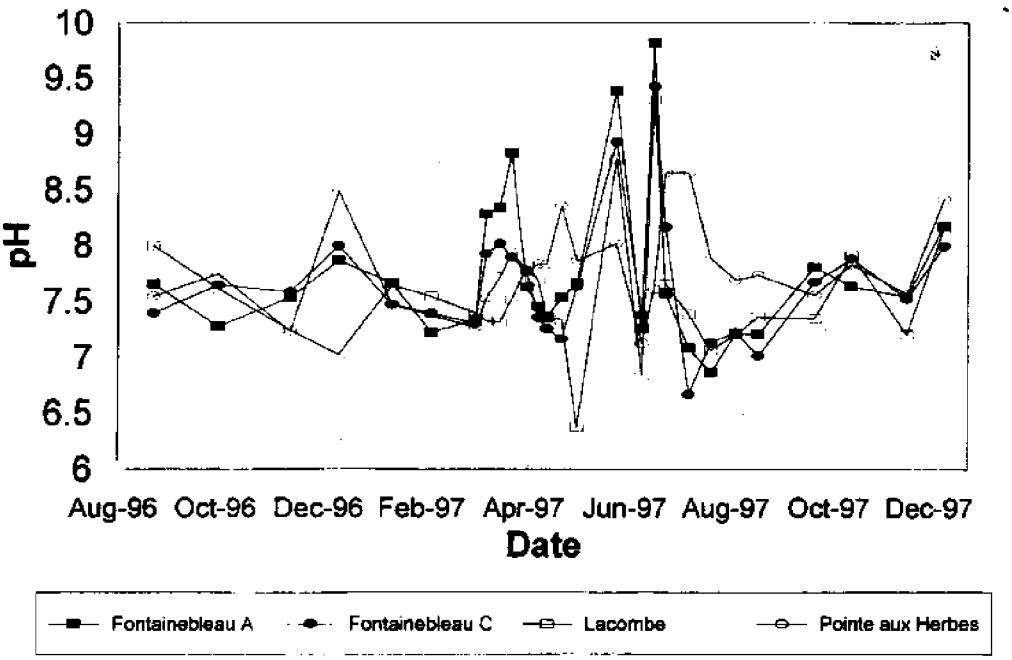

Fig. 2. pH measurements from four water quality study sites from September 13, 1996 through December 22, 1997.

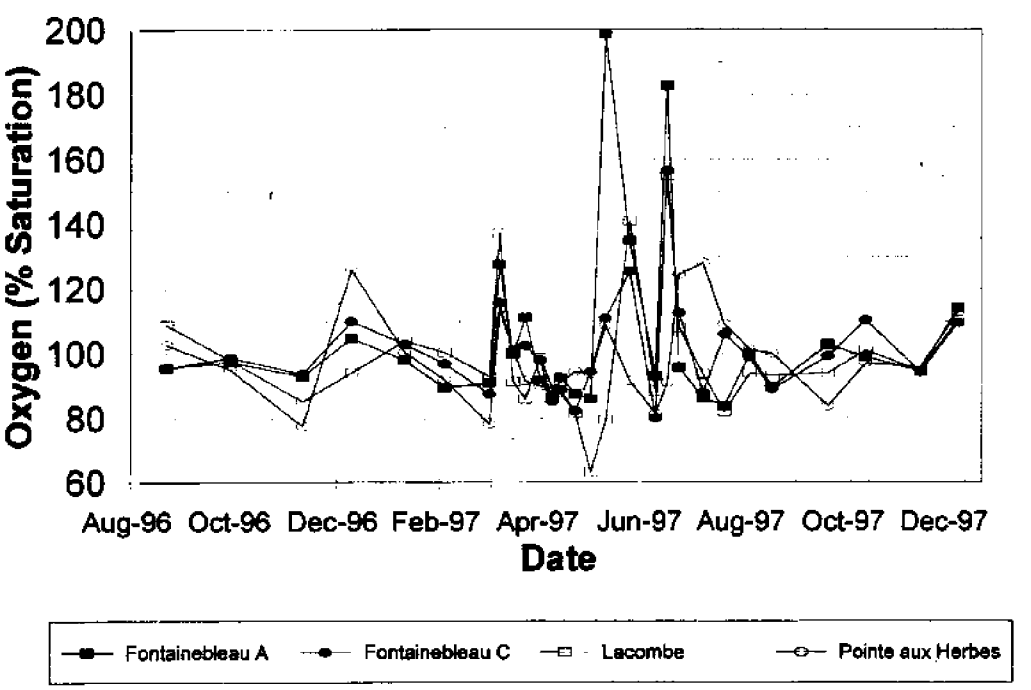

Fig. 3. Dissolved oxygen measurements from four water quality study sites from September 13, 1996 through December 22, 1997.

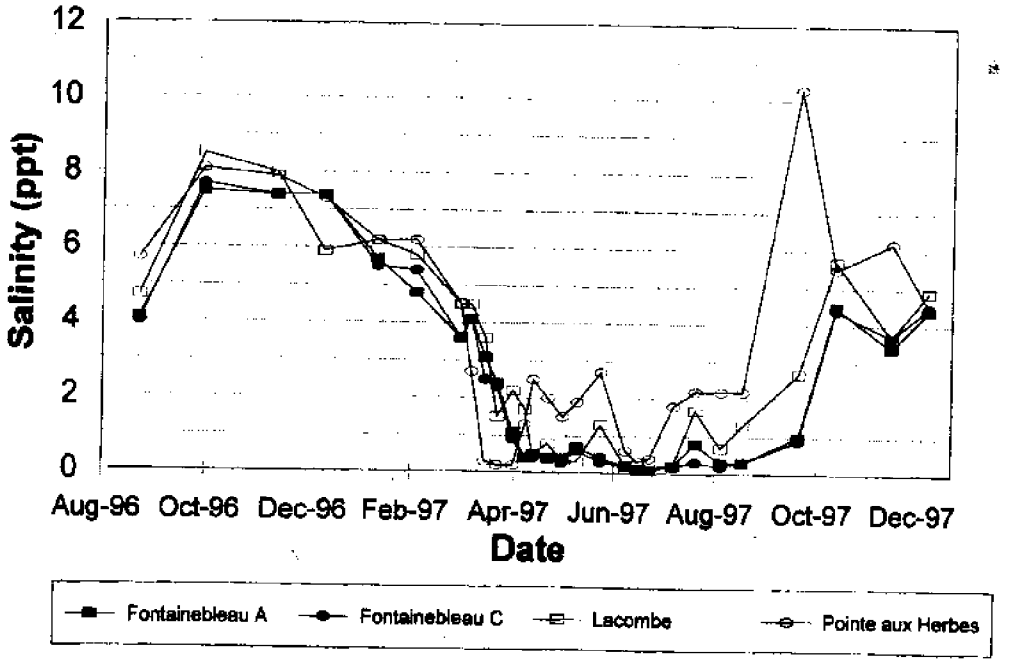

Fig. 4. Salinity measurements from four water quality study sites from September 13, 1996 through December 22, 1997.

(Fig. 3). Oxygen values were higher and more variable after the Spillway opening due to increased phytoplankton (Dortch and Achee 1998; Turner, Dortch and Rabalais 1998), photosynthesis and respiration.

Salinity

There was an overall trend of decreasing salinity to freshwater conditions after the March 17 Spillway opening (Fig. 4). Differences among sites were related to the direction of freshwater flow and distance from tidal passes (Fig. 1). Flow from the Spillway moved along the south shore, and water at Pointe aux Herbes was fresh (< 0.5 ppt) by April 3. Freshwater conditions persisted at this site until April 26. Salinity gradually decreased at the north shore sites after April 3, but freshwater conditions did not occur until April 26. In contrast, salinities at Pointe aux Herbes increased during this time and later fluctuated with rainfall and tidal exchange. North shore sites were fresh through July. On July 3 and 10, discharges from Bayou Lacombe and Bayou Castine lowered salinities at three north shore sites more than when Mississippi River water was present. This is because the Mississippi River is higher in dissolved solids than north shore streams (Cormier et al. 1992). Salinity did not return to the seasonal norm until October 1997.

PAR

Measurements of PAR expressed as a percentage of surface irradiance at 1-m depth are presented in Fig. 5. Low PAR values (mean 6.3%) occurred when Mississippi River water reached the sites, and later during phytoplankton blooms; higher values (mean 24%) occurred at other times. Values from the turbid period (April 3 - July 10, 1997) were significantly different ($p<0.01$) than values from other times (September 1996 through March 1997 and July 24 - December 1997).

SAV

Vallisneria foliar coverage (Fig. 6) was analyzed by three-way ANOVA with fixed factors (sampling period, sampling site and water depth). The 1996 and 1997 sampling periods were not significantly different ($p = 0.658$). Mean foliar coverage differed among water depths ($p < 0.001$), and foliar coverage was greatest at depths of 0.31 - 0.93 m. The inter-

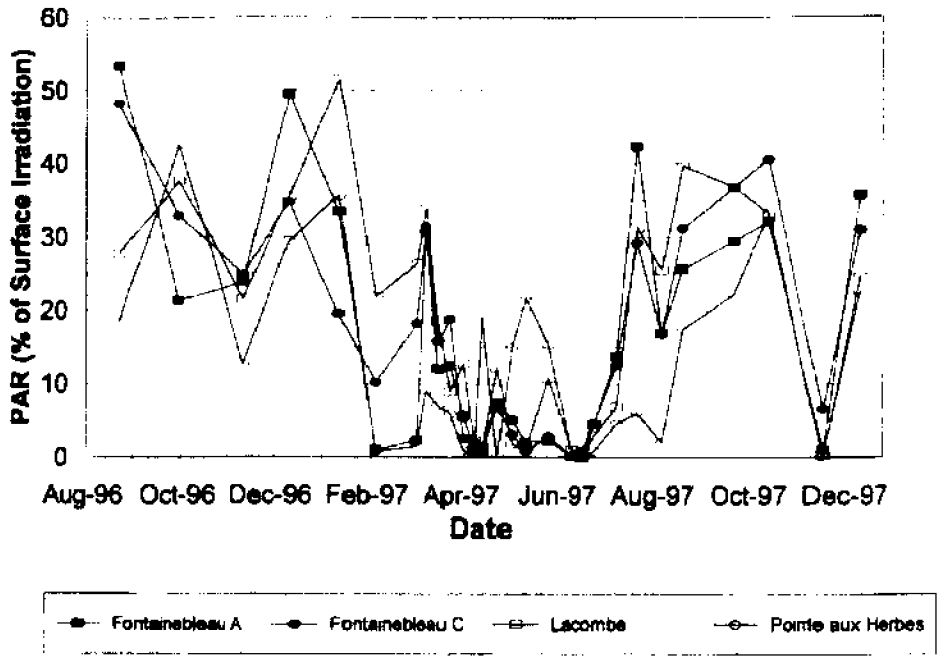

Fig. 5. PAR (Photosynthetically Active Radiation) measurements at a depth of one meter expressed as % of surface irradiance from September 13, 1996 through December 22, 1997.

action of sampling time, sampling site and water depth was not statistically significant ($p = 1.0$). This result shows that there were no significant changes in the depth distribution of *Vallisneria* foliar coverage from 1996 to 1997 at different sites (Fig. 6).

Ruppia foliar coverage was analyzed as above (Fig. 7) and was significantly greater in 1996 than 1997 ($p < 0.001$). There were significant differences among sampling sites ($p < 0.001$) and among water depths ($p < 0.001$). *Ruppia* foliar coverage was greatest at depths ranging from 0.31 - 0.62 m and from 0.62 - 0.93 m.

The interaction of sampling time, sampling site, and water depth was also statistically significant ($p < 0.001$). After-ANOVA cell mean comparisons for *Ruppia* were conducted to examine components of the significant interaction. Statistically significant differences were observed between 1996 and 1997 samples at all sampling sites, and ranged from the shallowest (< 0.31m) at Goose Point to the deepest depth (1.24 - 1.55 m) at Fontainebleau C. The significant differences at individual sites follow: Fontainebleau A, 0.61 - 0.93 m and 0.93 - 1.24 m; Fontainebleau B, 0.93 - 1.24 m; Fontainebleau C, 1.24 - 1.55 m; Goose Point, 0 - 0.31 m; Lacombe, 0.31 - 0.62 m; and Pointe aux Herbes, 0.62 - 0.93 m. The greatest reduction in *Ruppia* foliar coverage occurred at intermediate depths where 1996 coverages were greatest.

The *Myriophyllum* bed in Bayou St. John that was located north of the flood control structure had a surface area of 2,000 m^2 on March 5, 1997. The green filamentous alga, *Cladophora*, was first present as an epiphyte on the *Myriophyllum*. Growth of *Cladophora* continued through April and May and formed 1-m^2 floating blankets over the *Myriophyllum* bed. These *Cladophora* blankets and surface accumulations of phytoplankton shaded the *Myriophyllum* and resulted in a gradual decrease in the size of the bed. By June 1997, the former bed consisted of a few scattered plants with heavy *Cladophora* overgrowth. The remaining plants died during a period (June 16-17) of poor water quality (DO < 0.5 ppm) from the decomposition of accumulated blue-green algae that also caused a fish kill.

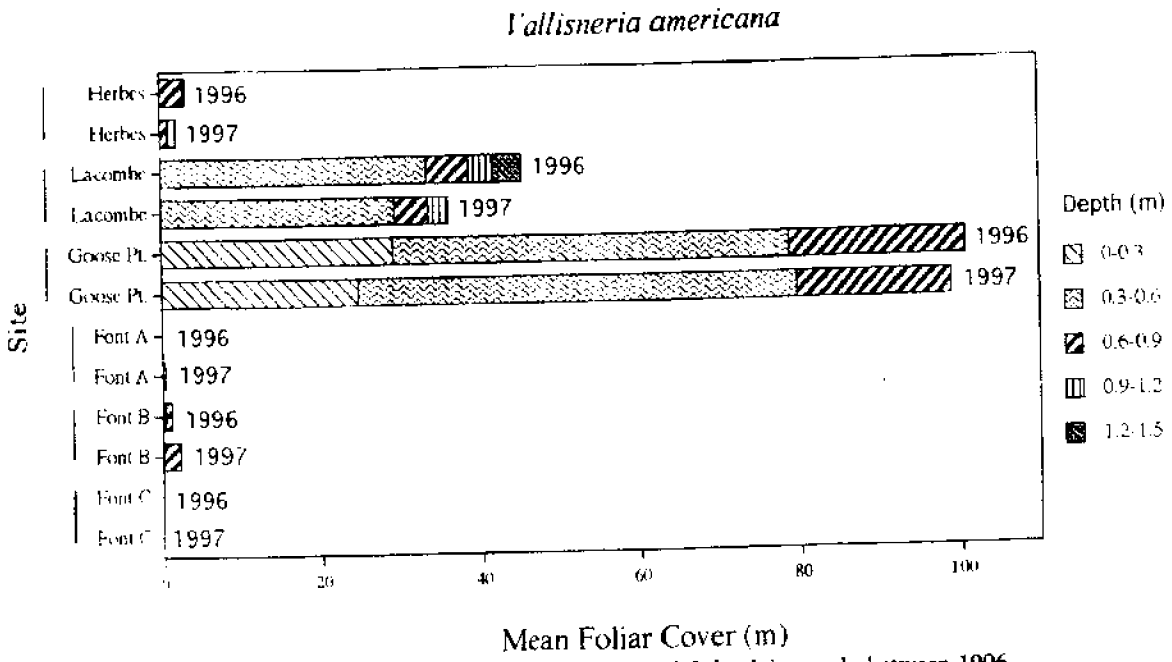

Fig. 6. Differences in *Vallisneria americana* abundance at 0.3 depth intervals, between 1996 and 1997 surveys at six sites.

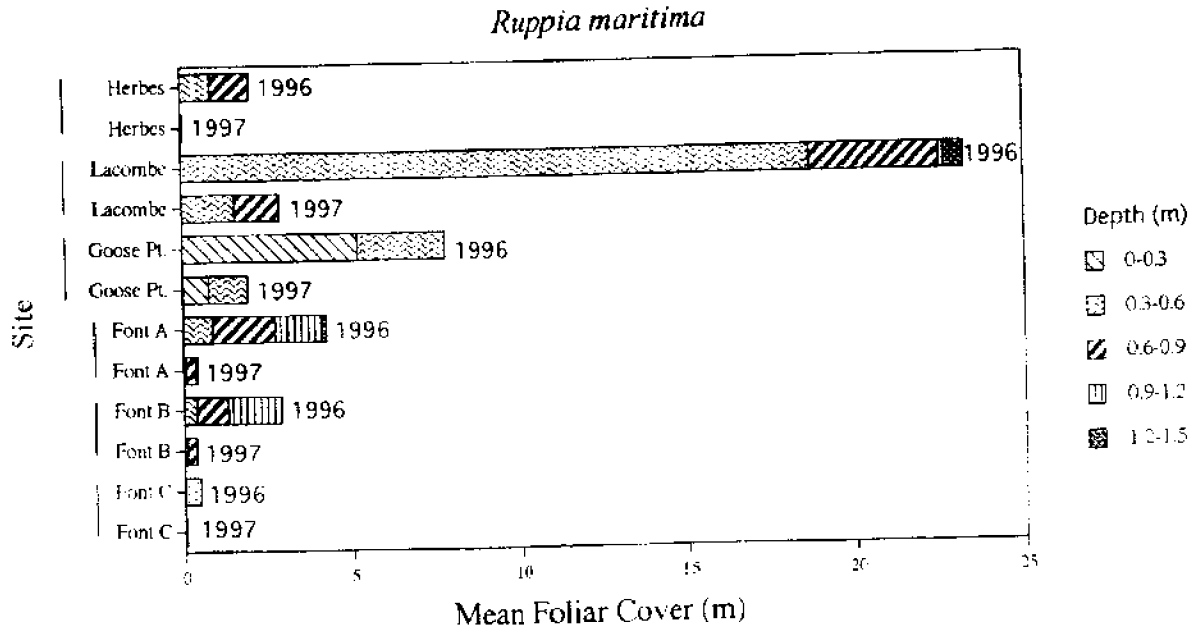

Fig. 7. Differences in *Ruppia maritima* abundance at 0.3 depth intervals, between 1996 and 1997 surveys at six sites.

Small amounts of *Najas* were present at Fontainebleau C in 1997, and small amounts of *Najas* and *Eleocharis* were present at Lacombe and Goose Point in 1996 and 1997. Both species occurred in shallow water, and our data did not indicate a change between sampling periods.

Epiphytes

Cladophora is generally present throughout the year in Lake Pontchartrain with episodes of extensive growth during spring, summer, and fall. In our study, *Cladophora* was absent in the 0.6 m sample from Lacombe and 0.6 and 0.9-m samples from Pointe aux Herbes in May, but present in most other samples (Table 1). *Cladophora* overgrowth on *Ruppia* occurred in 0.3 and 0.6-m samples from Fountainebleau A and C in May, 0.6 m *Ruppia* samples in September and 0.3-m *Najas* samples in September, and 0.6-m *Vallisneria* samples from Lacombe and Pointe aux Herbes in September (Table 1). *Cladophora* was present on *Myriophyllum* in Bayou St. John during February and March, overgrowth occurred in April, and with 1-m^2 floating surface mats developing during May and June.

Discussion

Despite the seemingly adverse conditions for SAV growth after the Bonnet Carré Spillway opening, there was no significant change in *Vallisneria* total coverage, or change with depth at any sites between 1996 and 1997. However, there were significant decreases in total *Ruppia* abundance and changes in depth distribution at all sites.

Of the environmental factors monitored, light reduction due to blue-green algal blooms and growth of *Cladophora* on *Ruppia* are the best explanations for the *Ruppia* decline. Temperature was not important because there was no decrease when Mississippi River water was present at the shallow study sites. Limitations due to a decrease in free carbon dioxide from increased pH (Titus and Stone 1982) cannot explain the persistence of *Vallisneria* and the decrease in *Ruppia* because *Ruppia* is common in alkaline brackish waters. The freshwater conditions that occurred after the Bonnet Carré Spillway opening differed from typical seasonal trends. Salinity is generally highest, 5 - 6 ppt, in November and generally decreases during winter

TABLE 1. Relative abundance of Cladophora on submersed aquatic vegetation (SAV).
A = Cladophora absent P = Cladophora present OG = Cladophora overgrowth

Date	Depth	Fontainebleau A	Fontainebleau C	Lacombe	Pointe aux Herbes
5/30/97					
	0.3 m	OG-Ruppia	OG-Ruppia	P-Ruppia	No SAV
	0.61 m	OG-Ruppia	OG-Ruppia	A	A
	0.91 m	P-Ruppia	P-Ruppia	P-Ruppia	A
7/3/97					
	0.3 m	P-Ruppia	P-Ruppia	P-Ruppia	No SAV
	0.61 m	P-Ruppia	P-Ruppia	P-Ruppia	P-Ruppia
	0.91 m	P-Ruppia	No SAV	P-Najas	P-Vallisneria
9/3/97					
	0.3 m	P-Vallisneria	OG-Najas	P-Ruppia	No SAV
	0.61 m	OG-Ruppia	No SAV	OG-Vallisneria	OG-Vallisneria
	0.91 m	No SAV	No SAV	P-Vallisneria	P-Vallisneria

and spring to a low of 2 - 3 ppt in May (Francis et al. 1994). However, as demonstrated in this study, freshwater conditions from local stream flow can persist for relatively long periods at north shore SAV sites. Freshwater conditions might be regarded as an explanation for the different response between species because *Ruppia* is common in brackish water and *Vallisneria* is common in freshwater (Orth and Moore 1983). However, *Ruppia* also occurs in inland freshwater habitats (Davis and Brinson 1980) and survives periodic freshwater conditions in Lake Pontchartrain (Burns et al. 1993).

Light reduction was an important factor. A significant decrease in PAR occurred at a depth of 1 m from April 3 through July 10, 1997. The April decrease in PAR was due to the introduction of Mississippi River water that is high in suspended silt and clay. Algal blooms were the main cause of the decrease in PAR after May 1997. Values averaged 6.3 % of surface irradiance during this period, compared with an average value of 24% at other times. Minimum light requirement for SAV ranges from 4.4 % to 29.4 % (Dennison et al. 1993). A minimum value of 8.2 % has been reported for *Ruppia maritima* (Duarte 1991). Minimum light requirements for *Vallisneria* are unknown, but it is an efficient carbon fixer at low light intensities (Titus and Adams 1979). Meyer et al. (1943) reported *Vallisneria* to be the most shade adapted of five submersed macrophytes, and found in short-term experiments that *Vallisneria* can maintain 25% of its surface photosynthetic rate at 0.5% of surface light (Meyer et al. 1943). The average value of 6.3% after the spillway opening may have been sufficient to support *Vallisneria*, but not *Ruppia*. This difference in shade tolerance may have contributed to the decrease in *Ruppia* and not *Vallisneria*. However, heavy *Cladophora* growth on *Ruppia* also decreased light availability to *Ruppia* . *Vallisneria* may be able to survive low light intensities by using energy reserves in large roots and tubers.

If light availability was the cause of SAV decline, one would expect that more SAV would have been lost from deeper water. Significant decreases in *Ruppia* occurred in deeper water (0.31 - 1.55 m) at Fontainebleau and Pointe aux Herbes, but at Goose Point and Lacombe the greatest decrease occurred in shallow water (0 - 0.62 m). Although any *Ruppia* present in deep water in 1996 (>0.62 m) was generally absent in 1997, the greatest reduction in *Ruppia* foliar coverage occurred at intermediate depths (0.31- 0.9 m) where 1996 coverages were greatest. Light availability was important, but the growth of *Cladophora* on *Ruppia* also blocked light, and may have contributed to *Ruppia* loss in shallow water.

The effect of wave energy was also important. *Ruppia* with heavy *Cladophora* growth was found uprooted and washed ashore at Fontainebleau and Lacombe after storms in April and May 1997. Heavy *Cladophora* growth may make *Ruppia* more susceptible to uprooting by wave energy. Wave energy could also explain *Ruppia* loss from 0 - 0.3 m where, even with *Cladophora* growth, light was probably not a limiting factor. The growth of *Ruppia*, and *Cladophora* on *Ruppia*, was dynamic (Table 1). *Ruppia* may have been more abundant at Fontainebleau A and C during the spring and summer than during the 1996 and 1997 quantitative surveys. The presence and episodic overgrowth of *Cladophora* on SAV supports *Cladophora* as a cause of *Ruppia* decline.

The cause of more *Cladophora* growth on *Ruppia* than *Vallisneria* is unclear. *Cladophora* grows on *Vallisneria* and can be abundant in areas protected from wave action. This difference in *Cladophora* growth in the littoral zone could be caused by differences in grazing activity related to differences in plant morphology. However, if *Vallisneria* is not easily uprooted by wave action, wave energy could remove *Cladophora* from *Vallisneria*. Weisner et al. (1997) found that *Myriophyllum spicatum* growth was reduced at sites with less wave energy due to increased epiphyte growth.

Terminal growth in *Myriophyllum* and *Ruppia*, but not *Vallisneria*, may make new surface growth prone to algal overgrowth. *Cladophora* overgrowth on *Myriophyllum* and the formation of large surface mats over the plants was the primary cause of the loss of the 1997 *Myriophyllum* bed in Bayou St. John. In 1995, a *Myriophyllum* bed present at the same site also died because of *Cladophora*

overgrowth after the Spillway opening (Burns and Poirrier 1996). In Lake Pontchartrain, growth of *Ruppia* and *Myriophyllum* occurs in late winter and early spring (Burns et al. 1993), but new growth of *Vallisneria* occurs with increasing water temperatures in April. The earlier surface growth of *Myriophyllum* and *Ruppia* may also have made them more susceptible to *Cladophora* fouling.

Conclusions

Based on a comparison of SAV foliar coverage present in 1996 and 1997, the 1997 Bonnet Carré Spillway opening did not affect *Vallisneria* areal coverage. However, a significant decrease in *Ruppia* areal coverage was observed, and a *Myriophyllum* bed in Bayou St. John died. Decreases in light availability to *Ruppia* from blue-green algal blooms, heavy *Cladophora* growth, and the uprooting of *Ruppia* with heavy *Cladophora* growth by wave energy appeared to be the main causes of *Ruppia* decline. The loss of a *Myriophyllum* bed was caused by phytoplankton shading, *Cladophora* overgrowth, and adverse water quality from algal decomposition.

ACKNOWLEDGMENTS

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What is the Threat of Harmful Algal Blooms in Louisiana Coastal Waters?

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ABSTRACT: Harmful Algal Blooms (HABs) have increased worldwide, at least in part in response to escalating coastal eutrophication. Since nutrient inputs and eutrophication have increased substantially in Louisiana coastal waters, the available data on the occurrence of HABs were assembled to assess the threat in this region. Twenty-four HAB species are present, ranging in effects from water discoloration to animal mortality. Four types of toxins which can potentially affect human health have been detected in algae or oysters, including brevetoxins, hepatotoxins, domoic acid, and okadaic acid, although no known human illnesses have occurred. At least two the HAB groups producing these toxins are stimulated by high nutrient availability and others may be as well. In order to keep HAB problems from escalating in Louisiana coastal waters, it is essential that nutrient inputs to coastal areas not be increased.

INTRODUCTION

Harmful Algal Blooms (HABs) refer to what used to be known as "red tides" but now include a greater variety of organisms and phenomena. "Red tides" described red water discoloration, usually, but not always, caused by dinoflagellates, some of which produce toxins that cause human illness and/or fish kills. Blooms are now known to range in color from red to brown to green or blue-green and include other algal groups besides dinoflagellates and also cyanobacteria and autotrophic ciliates. Not all bloom organisms are harmful beyond causing water discoloration or episodic hypoxia/anoxia. Further, visually obvious blooms are not always necessary for deleterious impacts to occur. Thus, the term HABs refers to complex phenomena defined by their impact rather than their appearance.

There has been a global increase in both the numbers of incidents of HABs and the type of organisms that cause the problems (Shumway 1990; Hallegraeff 1993; ECOHAB 1995). Several reasons have been given for this increase: (1) escalating nutrient enrichment of coastal areas, which stimulates blooms; (2) growing global shipping, which transports organisms to new areas; (3) increasing aquaculture and mariculture, which provides micro-environments, magnifies transport opportunities, and enhances surveillance; and (4) increasing numbers of scientists aware of the problem. Although all four are probable factors, it is generally felt that escalating coastal eutrophication is the main

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factor and that future increases in nutrient inputs to coastal areas will magnify the threat.

Nutrient inputs to Louisiana waters have increased to both shelf and estuarine environments with concomitant enhanced eutrophication (Turner and Rabalais 1991, 1994; Rabalais et al. 1996; Parsons 1996). Red tides have been documented in the northern Gulf of Mexico (Perry et al. 1979; Perry 1980; Perry and McClellan 1981a, b; Eleuterius et al. 1981; Maples 1983b, Rabalais et al., 1995), but Louisiana has not suffered the dramatic impacts observed elsewhere. In general it has been assumed that the low salinities and high turbidities of Louisiana coastal waters prevented blooms. Several recent blooms, resulting in fish kills near the Louisiana/Texas border (Robichaux et al. 1998), closure of oyster beds to commercial harvesting (Dortch et al. 1998), and advisories against recreational use of Lake Pontchartrain (Dortch and Achee' 1998), combined with growing public awareness of problems elsewhere, have increased concern about HABs in Louisiana coastal waters. The purpose of this paper is to compile what is known about HAB species and to assess the threats that they pose. The long-term goal is to determine whether the threat is severe enough to warrant a regular monitoring program and how that monitoring should be conducted. Although the focus is on Louisiana coastal waters, pertinent information from other low salinity areas of the northern Gulf of Mexico will also be included in this evaluation.

METHODS

There are three sources for the data compiled in this report.

(1) Published reports (Housely 1976; Perry et al. 1979; Perry 1980; Eleuterius et al. 1981; Perry and McLelland 1981a, b; Maples 1983a, b; Rabalais et al. 1995; Dortch et al. 1997,1998; Dortch and Achee' 1998; Parsons et al. 1998; Robichaux et al. 1998).

(2) Unpublished bloom/fish kill investigations. These were conducted at LUMCON on samples brought in by individuals or as part of investigations by the Louisiana State Departments of Environmental Quality, Health and Hospitals, or Wildlife and Fisheries. Some of the data are from internal agency reports (Latapie 1969, Morrison 1980, Ancelet et al. 1981, Bejarano et al. 1981) and some have been published (Rabalais et al. 1995; Dortch et al. 1998), but all are in databases at LUMCON.

(3) Systematic, routine sampling conducted at LUMCON. These include monthly sampling at a shelf station 20 miles south of Cocodrie since 1990, weekly sampling at a station in the Terrebonne Bay estuary since 1993, sampling during approximately 18 cruises covering various segments of the Louisiana shelf since 1989 (described in Dortch et al. 1997), and sampling over oyster beds east of the Mississippi River in conjunction with the Louisiana Department of Health and Hospitals Oyster Monitoring. All of these data are available as databases at LUMCON and most are available from the NOAA Nutrient Enhanced Coastal Ocean Productivity (NECOP) Data Management Program (Hendee 1994) and/or the National Oceanographic Data Center. Some of these data have been published (Rabalais et al. 1995; Dortch et al. 1997; Parsons et al. 1998 in press; Robichaux et al. in press).

Phytoplankton were preserved in 0.5% glutaraldehyde, size fractionated onto polycarbonate filters (0.2, 3, and 8 mm), and identified and counted using epifluorescence microscopy (Dortch et al. 1997). Some identifications were confirmed with Scanning Electron Microscopy (Parsons et al. 1998).

RESULTS AND DISCUSSION

Harmful Algal Species Present in Louisiana Coastal Waters

Twenty-four potentially harmful species have been identified so far in Louisiana waters, representing five taxonomic groups (Table 1). The criteria for choosing the organisms on the list are that they occur in Louisiana waters and that they cause at least one of three deleterious effects somewhere in the world: water discoloration/bioluminescence, animal mortality, or human illness. Potential impacts are given for each species

Table 1. The possible impacts, temporal and spatial occurrence and bloom characteristics of HAB species in the Northern Gulf of Mexico. ? explained in text. PSP: Paralytic Shellfish Poisoning; DSP: Diarrhetic Shellfish Poisoning; ASP: Amnesic Shellfish Poisoning; Dino: dinoflagellate; Cyano: cyanobacteria; Raph: raphidophyte; Auto cil: autotrophic ciliate; Diat: Diatom; F: freshwater; E: estuarine; c: Coastal; O: open ocean; F: fall; W: winter; Sp: spring; Su: summer; Y: Yes; N: No.

Species	Type	Animal Mortality	Potential Human Illness/Mortality	Location	Season	Bloom Color	Bioluminescence
Alexandrium monilatum	Dino	X		E, C	F	Brown to Red	Y
Alexandrium ostenfeldii?	Dino		PSP?	E	W	Brown	Y
Anabaena spp.	Cyano	X	hepatotoxins neurotoxins?	F, E	Su	Green to Blue-Green	N
Ceratium hircus	Dino			C	Su	Brown	N?
Dinophysis caudata	Dino		DSP?	C	All	None	N
Gymnodinium breve	Dino	X	NSP	E?, C, O	Su,F	Red	N
Gymnodinium sanguineum	Dino	X		E, C	Su	Red to Brown	N
Heterocapsa spp.	Dino			E, C?	Sp	Brown	N
Heterosigma akashiwo	Raph	X		E, C	Sp?	Red	N
Lingulodinium polyedrum	Dino		PSP?	C	Su?	Brown	N
Mesodinium rubrum	Auto cil			E, C	All	Red	N
Microcystis sp.	Cyano	X	hepatotoxins neurotoxins?	F, E	Su	Green to Blue-Green	N
Noctiluca sp.	Dino	X?		E, C	Sp	Red/Orange, Pink	Y
Prorocentrum micans	Dino		DSP?	E, C	All?	None	N
Prorocentrum compressum	Dino		DSP?	E, C	All?	None	N
Prorocentrum gracile	Dino		DSP?	E, C	All?	None	N
Prorocentrum mexicanum	Dino		DSP	?	?	?	N
Prorocentrum minimum	Dino		Venerupin poisoning? DSP?	E, C	All?	Red	N
Pseudo-nitzschia delicatissima	Diat		ASP	E, C?	All	Brown	N
Pseudo-nitzschia multiseries	Diat	X?	ASP	E?	All	Brown	N
Pseudo-nitzschia pseudodelicatissima	Diat		ASP	E, C?	All	Brown	N
Pseudo-nitzschia pungens	Diat		ASP?	C, O?	All	Brown	N
Scrippsiella spp.	Dino			E, C	Sp?	Brown	N
Trichodesmium spp.	Cyano	X		C, O	Su?	Tufts & Puffs	N

(Table 1): an impact without a ? signifies that the impact has been documented in Louisiana waters and an impact with a ? indicates that the impact has not been clearly documented in Louisiana waters or that there is some question about the impact elsewhere as well. Thus, the ? can have several different meanings, as the following examples indicate. (1) *Alexandrium ostenfeldii* has only been tentatively identified and samples are being sent elsewhere for confirmation because of the possibility that it is toxic (Steidinger and Tangen 1996). (2) *Mesodinium rubrum*, a ciliate with cryptomonad symbionts, causes massive red tides in many areas (Lindholm 1985), but although it is abundant in Louisiana waters, it has never been observed to discolor the water. (3) *Anabaena* spp. and *Microcystis* spp. can produce both hepatotoxins and neurotoxins (Sivonen 1996), but in samples from Louisiana only hepatotoxins were measured (Dortch and Achee' 1998). (4) Okadaic acid has been measured in oysters in Louisiana and elsewhere in the northern Gulf of Mexico (Table 2). Some species of *Dinophysis* and many benthic/epiphytic species of *Prorocentrum* produce okadaic acid and its derivatives, which cause DSP. None of the planktonic species abundant in Louisiana waters is known to produce it, but a number have not been adequately tested. While it is possible that okadaic acid may be produced by *P. mexicanum*, this benthic species is relatively rare in water samples. It is possible that either one of the more abundant known species is producing okadaic acid or there are benthic/epiphytic species that are getting into water samples. (5) *P. minimum* is a special case, whose threat is difficult to assess. It causes massive red tides in a number of areas, like the Chesapeake Bay (Luckenback et al. 1993), with no immediately obvious deleterious effects. However in Japan in 1942 blooms led to venerupin poisoning from shellfish consumption, with many deaths (Okaichi and Imatomi 1979). It has been implicated as the cause of other less serious outbreaks of DSP-like human illness (Tangen 1983, Freudenthal and Jijina 1985, Silva 1985), but no further work has been done on the human toxicity of this organism. Finally, it may cause problems with shellfish mortality or development (Luckenback et al. 1993, Wikfors and Smolowitz 1995), is sometimes associated with fish kills (Perry and McLelland 1981 b, Rabanni et al. 1990, Ho and Hodgkiss 1993), and is in the same genus with known okadaic acid producers.

Such an extensive list, including many bloom-forming species, indicates that the low salinities and high turbidities of Louisiana coastal waters do not prevent blooms. In fact, many of the organisms have sufficient motility that they can position themselves at the water surface, thereby avoiding any light limitation that might result from the high turbidities. The high turbidities may make it more difficult for untrained observers to recognize blooms, since many are the color of suspended sediment. The locations where the species are found (Table 1) indicate that many are adapted to low salinity waters. Even *Gymnodinium breve*, the red tide organism which causes major problems in the high salinity waters off of Florida and Texas (Tester and Steidinger 1997), has now been found at relatively low salinities off Louisiana (Dortch et al. 1998).

The long-term goal of this research is to determine the seasons and places where Harmful Algae are likely to occur in order to design a monitoring program if the frequency of blooms warrants it. In Table 1, all the available information about the temporal and spatial occurrence of each species is given. A ? indicates that there is some uncertainty in the distribution of the organism, although the uncertainty can arise from several causes. For example *Prorocentrum mexicanum* has been identified only recently and is too rare to describe its distribution. In the case of *Pseudo-nitzschia*, the genus is easy to recognize with light microscopy, but identifying the toxin-producing species requires either molecular methods or SEM, which we have only used for the last two years (Parsons et al. 1998). Finally, a number of organisms are seen in low to moderate abundance all year round, but a careful quantitative analysis of the entire data set is necessary to determine the times of the year when they are most abundant.

The color of blooms and the occurrence of bioluminescence are also indicated (Table 1) since, sometimes, all of the information together can be diagnostic of the organism present. Because many incidents of water discoloration or bioluminescence are not a cause for concern from a public health

perspective, it would be helpful to develop information which can be used to reassure the public and help public officials make decisions about monitoring. For example, a bloom which discolors the water "tomato soup" color, is bioluminescent at night, and occurs in the spring can only be *Noctiluca* sp., which is probably harmless, although fish tend to be absent when it is present in large numbers. Most blooms, however, will require some monitoring to rule out organisms that are a threat.

One difficulty with establishing a list of Harmful Algal species present in Louisiana coastal waters is that many are not observed in routine sampling, but appear to occur as sporadic blooms. *Heterosigma akashiwo*, *Lingulodinium polyedrum*, and *Microcystis* sp. have never been present and *Alexandrium monilatum*, *A. ostenfeldii*, *Ceratium hircus*, *Gymnodinium breve*, and *Noctiluca* sp. have only rarely been counted during regular sampling. They are either not present at all or are below the limits of detection in our routine methods, 100 to 1000 cells l^{-1} depending on location, time of year, and primarily, suspended particle concentration. Most of the samples containing these organisms have been brought in as special samples by individuals or state agencies because of water discoloration, bioluminescence, or fish kills, and abundances were in the range of 1×10^4 to 1×10^7 cells l^{-1}, depending on the size of the organism. Blooms may also cover large or small geographic areas. Thus, it is highly likely that, despite our long term systematic sampling (>4000 samples), there are other Harmful Algal species which have not yet been detected. This problem also magnifies the difficulty of determining the spatial and temporal distribution of blooms of a particular organism, because the available information is dependent on "chance encounters". An effort is underway to expand the spatial and temporal coverage of both regular sampling and "chance encounters."

HAB Species Which Are Not Now a Problem

There are several groups of organisms that are not in Tables 1 and 2 because they may not presently be a significant problem in Louisiana waters. Some estuaries in Texas have been plagued with the "Texas Brown Tide" (Stockwell et al. 1993). A yearlong study in Terrebonne Bay and the Louisiana shelf has shown that the organism is not present in Louisiana waters, although it is present in Florida (Lopez and Villareal, pers. comm.). Similarly, *Pfiesteria piscicida* and similar organisms do not now appear to pose a significant problem in Louisiana waters because few fish with the typical lesions are present, even in fish kills. However, *Pfiesteria*-like organisms have been isolated from Mobile Bay (Burkholder and Glasgow 1997) and since problems with *Pfiesteria* are enhanced by high inorganic and organic nutrient inputs (Burkholder and Glasgow 1997), it is possible that *Pfiesteria*-like blooms could occur in Louisiana.

Water Discoloration and Animal Mortality

A number of species produce large scale blooms in Louisiana waters that are quite evident, even to untrained observers, including *Alexandrium monilatum*, *A. ostenfeldii*, *Gymnodinium breve*, *G. sanguineum*, *Heterosigma akashiwo*, *Lingulodinium polyedrum*, and *Prorocentrum minimum*. Of these, only *G. breve* and, maybe *P. minimum*, are known to cause human health problems, which will be discussed below. Two others are suspected of causing human health problems, *A. ostenfeldii* and *L. polyedrum*, but the evidence is quite scant (Rabalais et al. 1995; Steidinger and Tangen 1996). However, animal mortality is well documented for several of these species, *A. monilatum*, *G. breve*, *G. sanguineum*, *H. akashiwo*, and *P. minimum* (summarized in Table B1 in Rabalais et al. 1995; Dortch et al. 1998). The most usual manifestation is fish kills, but invertebrate, bird, turtle, and marine mammal mortality can also occur. In some cases is not known whether a toxin is produced or whether the deaths are due to oxygen depletion during the decay of the bloom (e.g. Robichaux et al. 1998). Finally, more subtle ecosystem effects are also possible (Smayda 1992). For example, consumption of toxic algae by larval fish can effect recruitment, although this process would be difficult to detect in the field (Smayda 1992). Thus, even when blooms do not have a human health impact, they may have a significant impact on the ecosystem.

Human Health Impacts

Four toxins, that can cause human illness and, in some cases death, have been measured either in phytoplankton or shellfish in Louisiana coastal waters (Table 2). Two of these, brevetoxins and hepatotoxins, were measured during algal blooms and resulted in actions to protect public health including closure of oyster beds to harvesting east of the Mississippi River for up to 5 months in 1996-1997 (Dortch et al. 1998) and issuance of an advisory against recreational use of Lake Pontchartrain in the summer of 1997 (Dortch and Achee' 1998). Two others, domoic acid and okadaic acid, have been measured during routine sampling. Evaluation of the human health implications of the occurrence of all of these toxins requires a better understanding of the frequency of blooms, the temporal and spatial variation of their occurrence, and the factors which cause toxin production. At present there have been no known human illness or death due to toxic algae in Louisiana.

The abundance of two of the toxin-producing groups is clearly stimulated by nutrients. The *Anabaena* spp. and *Microcystis* sp. blooms in Lake Pontchartrain in the summer of 1997 occurred after the diversion of high nutrient river water into the Lake for flood control purposes (Dortch and Achee' 1998). Analysis of algal abundance and nutrient data indicate that increased availability of nitrogen and phosphorus stimulated the bloom and that nitrogen limitation ended the bloom. A similar bloom occurred in 1995 when river water leaked into the Lake and local runoff was high (Dortch and Achee' 1998). *Pseudo-nitzschia* spp. abundance on the shelf reaches some of the highest levels ever observed anywhere in the world in the plume of the Mississippi River during peak river flow (Dortch et al. 1997), suggesting nutrient stimulation. Now that individual species can be identified on a routine basis (Scholin et al. 1994; Parsons et al. 1998), it will be possible to determine how nutrients stimulate growth and toxicity of each species. Elsewhere, blooms of this group are stimulated by nutrients (Dortch et al. 1997; Fryxell et al. 1997). For the other two groups, the role of nutrients is not clear. Physical transport is the mechanism most frequently cited as leading to *Gymnodinium breve* blooms (Tester and Steidinger 1997), but its nutrient requirements have not be examined systematically. Although coastal blooms may be initiated by transport events, their continuation may be dependent on the availability of nutrients in those areas. Finally, very little is known about the physiological ecology of the known okadaic acid producers, including their nutrient requirements, because most cannot be cultured. Further, it is impossible to determine their relationship to nutrients in the northern Gulf, since it is unclear what species are producing okadaic acid.

Historic Changes in Harmful Algal Blooms

The hypothesis that increasing nutrient inputs to coastal areas has caused the increase in HABs is an attractive hypothesis that has been very difficult to test, because of the lack of historical data on nutrients and algal species composition. Increasing eutrophication has been well documented both on the Louisiana shelf and in estuaries, with large increases in nutrient inputs since the 1950's (Turner and Rabalais 1991, 1994; Parsons 1996) and it would be an excellent area to test the hypothesis. The first record of an algal blooms dates to 1969 (Latapie 1969). From 1969 to 1990 there were 6 reports of red tides (Perry 1980; Perry and McLelland, 1981a; Morrison unpublished, 1980; Ancelet et al. unpublished, 1981; Bejarano et al. unpublished, 1981; Eleuterius et al. 1981; Maples 1983b) and after 1990 there are 18 reports. The apparent increase in frequency may, however, be due solely to increased numbers of observers (e.g. LUMCON, starting in 1989).

The only group for which there are good historical data is the diatom species *Pseudo-nitzschia*, in which some species produce domoic acid which causes ASP. Studies conducted in 1955-1957 (Simmons and Thomas 1962), 1972-1974 (Fucik and El-Sayed 1979), and 1990-1994 (Dortch et al. 1997) are directly comparable and show a large increase in the abundance of this genus since the 1950's (Rabalais et al. 1996; Dortch et al. 1997). Because it is thought that growth of this group of organisms is stimulated by nutrients, the increase can be attributed to increasing nutrient inputs since that time. Investigations of algal remains preserved

Table 2. Summary of outbreaks of toxic algae with human health impacts in the low salinity waters of the northern Gulf of Mexico.

- **Brevetoxins in Oysters (Neurotoxic Shellfish Poisoning, NSP)**
  - * *Gymnodinium breve* (Dortch et al. 1998)
    - —Fall, 1996 in Louisiana, Mississippi, Alabama coastal waters
    - —Oyster beds closed for harvest for 1 to 5 months
    - —First confirmed bloom in low salinity waters
  - * Common HAB species in Gulf of Mexico (Tester and Steidinger 1997)
    - —Normally occurs at high salinities
    - —Causes fish kills and human respiratory problems as well as NSP
  - * Link to nutrient inputs unclear, but being re-evaluated

- **Cyanobacterial Hepatotoxins in Phytoplankton**
  - * *Anabaena* spp. and *Microcystis* sp. (Dortch and Achee' 1998)
    - —Summer, 1997, Lake Pontchartrain
    - —Probably resulted from increased nutrients due to diversion of Mississippi River water
    - —Advisory issued against recreational use
  - * Present in other fresh and very low salinity water bodies in Louisiana
    - —Lake Pontchartrain bloom July 1995
    - —Lake Salvadore bloom January 1995
    - —Lake Borgne July/August 1997
  - * Blooms of cyanobacteria and impacts increasing worldwide due to eutrophication (Pearl 1996)

- **Domoic Acid in Phytoplankton (Amnesic Shellfish Poisoning, ASP)**
  - * *Pseudo-nitzschia* spp.
    - —At least 4 toxin-producing species (Parsons et al. 1998)
    - —High cellular domoic acid concentrations measured (Doucette et al. 1997)
  - * All coastal and shelf waters (Dortch et al. 1997; Parsons et al. 1998)
    - —Most abundant in winter in estuary and in spring/fall on shelf
    - —On shelf peak in abundance corresponds to peak in river flow
  - * Link with nutrient inputs in Louisiana and elsewhere (Dortch et al. 1997)

- **Okadaic Acid in Oysters (Diarrhetic Shellfish Poisoning, DSP)**
  - * Measured at 2 locations in Northern Gulf of Mexico
    - —Mobile Bay, Fall, 1990 (Dickey et al. 1992)
    - —Terrebonne Bay, Jan.-Mar. 1995 (Dickey, pers. comm.)
  - * No species clearly linked to either occurrence
    - —*Dinophysis caudata* and *Prorocentrum* spp. frequently present but none shown definitively to produce okadaic acid
    - —*Prorocentrum mexicanum* known okadaic acid producer, rarely present
    - —Not possible to evaluate link to nutrients at present

in sediments are being used to directly test the link between eutrophication and increased algal blooms in Louisiana coastal waters.

SUMMARY

There are at least 24 potentially HAB species present in Louisiana waters. They have resulted in water discoloration and animal mortalities. Although four toxins that can cause human illness and sometimes death have been detected in phytoplankton or shellfish, there have been no known incidents of human illness in Louisiana. Low salinities and high turbidities do not prevent algal blooms and at least some of the HABs are stimulated by high nutrient availability. For many others too little is known about the conditions leading to blooms to assess the role of nutrients. However, the association worldwide between increasing coastal eutrophication and HABs suggests that it is prudent to avoid increasing nutrient inputs to coastal areas. This is especially true in Louisiana coastal waters where HAB species are present and eutrophication has occurred, but the problems at present are not as severe as in many other areas of the world.

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Nitrogen Losses in Water Flowing Through Louisiana Swamps

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ABSTRACT: A comparison was made of nitrogen concentrations entering and leaving two ecosystems with well-defined input and outlet channels: (1) the Atchafalaya River basin, a relatively natural and large (4,662 km²) alluvial swamp receiving about one-third of the Mississippi River discharge, and, (2) the Bonnet Carré spillway, a 13,450 ha flood peak reduction spillway near New Orleans. Six percent of the total nitrogen was removed from the Atchafalaya River during the transit downstream through the basin. Dissolved nitrate concentration, a major constituent of concern, remains essentially unchanged (+4%) from upstream to downstream monitoring stations. The Bonnet Carré spillway discharge resulted in a comparatively low total nitrogen loading rate of < 0.01 g N m^{-2} d^{-1} and inconsistent patterns in uptake and release among the nitrogen forms. The uptake of nitrate in 7 of 8 examples, contrasted with the net release of TKN and TN, in 5 of 6 and of 4 of 6 examples, respectively. There was a general inverse relationship between the amount of nitrogen retained and loading rates.

These results were compared to literature values for sewage treatment systems and other swamps. Natural and alluvial swamp forests retain a much smaller amount of these elements flooding them than do wetland wastewater flowing overland or belowground. The low retention rates for these large natural systems is attributed to the relatively low nutrient concentration compared to sewage treatment systems, the short duration of the overland flow (<2 weeks), large size, and the general absence of belowground flow. Diverting mainstem Mississippi River water through a flood control spillway (part of a proposed wetland restoration project) will apparently remove a relatively insignificant amount of the total nitrogen loading. The reduction of overbank flooding from flood protection levees did not significantly diminish nutrient loading to the continental shelf.

Keywords: wetland, nutrient cycling, nitrogen, overland flow, swamp, Louisiana, nitrogen, phosphorus, suspended sediments

Introduction

Water quality may change when water flows through or over wetlands because of the various diverse elemental pathways and ecological processes integrating water fluxes with the wetland structure. For example, the nitrogen assimilation rate for natural vegetation ranges from 0.1 to 2 kg ha^{-1} d^{-1} (Nixon and Lee 1986) and potential nitrogen reduction rates may range 10 fold over a 20 °C temperature range (Fig. 13-10 in Kadlec and Knight 1996). The effects of variation in the hydrologic regime are quite robust and pervasive across all wetland types. For wastewater treatment systems, which are especially well documented, there are several aspects that are particularly important and interrelated: constituent concentration, loading rates, retention time, and water depth (Kadlec and Knight 1996). The nutrient quality and loading rates in relatively undisturbed natural systems are usually quite different from that of wetland wastewater

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treatment systems. The former spread comparatively low concentration nutrients in large volumes over large areas; the latter involve engineered systems over smaller areas (typically less than 2 ha) and at nutrient concentrations typical of sewage, agricultural runoff, or industrial sources. Economic considerations and predictability are the major concerns of wastewater treatment systems, not natural systems.

I evaluated these differences in water flowing into and out of natural and engineered systems using water quality data from a large swamp (the Atchafalaya swamp, south Louisiana, USA) and an alluvial floodplain spillway (Bonnet Carré, near New Orleans), and also data for wastewater treatment systems (from the North American Wetland Treatment System Data Base (NAWTDB) and Danish Wetland Treatment System Database (DWTSD) summarized in Kadlec and Knight 1996). In the process, I describe the low percentage of nutrient retention for a natural riverine swamp system. These data are then used to estimate the nitrogen retention rates for a proposed river diversion near New Orleans.

Site Descriptions

The Atchafalaya Swamp

The Atchafalaya River forms from the confluence of the Red River and the one-third of the Mississippi River that is diverted into the Atchafalaya Basin near St. Francisville, LA (Fig. 1). There are no permanent settlements within the depressional basin that is almost entirely an alluvial swamp. Towns located on the edge of the basin whose drainage enters the basin have a total population estimated to be less than 100,000. The 4,662 km^2 swamp basin is a reserved for occasional use as a floodway. Constructed levees on both sides of the main channel constrain water to a largely north to south flow. Floodwater does not flow over these constructed levees and out of the basin. Additional amounts of water from the Mississippi River are diverted during unusually high flood stages through a spillway and into the Atchafalaya basin. These actions are meant to reduce water heights downstream, especially for the benefit of the city of New Orleans, which is mostly at sea level, or below.

The Bonnet Carré Spillway

The Bonnet Carré Spillway (Fig. 1) diverts Mississippi River water into Lake Pontchartrain during exceptionally high water. The 1,325 ha spillway was completed in 1931 and is located about 33 miles (19 km) upstream from New Orleans on the left descending bank. The spillway's designed capacity is 7080 m^3 sec^{-1} and was used to reduce the floodwater heights in 1937, 1945, 1950, 1973, 1975, 1979, 1983 and 1997. There is a proposal for an additional spillway of 10,000 ha to divert 256 m^3 sec^{-1} into the watershed (Day 1997). Another diversion is proposed (and authorized, but without approval; Connor 1996) whose design and operation schedule is known as the New General Design Memorandum (NGDM; conducted after a re-analysis in 1996). The proposed diversion schedule in the NGDM proposes an average monthly flow of 546, 850, 612, and 122 m^3 sec^{-1} during February,

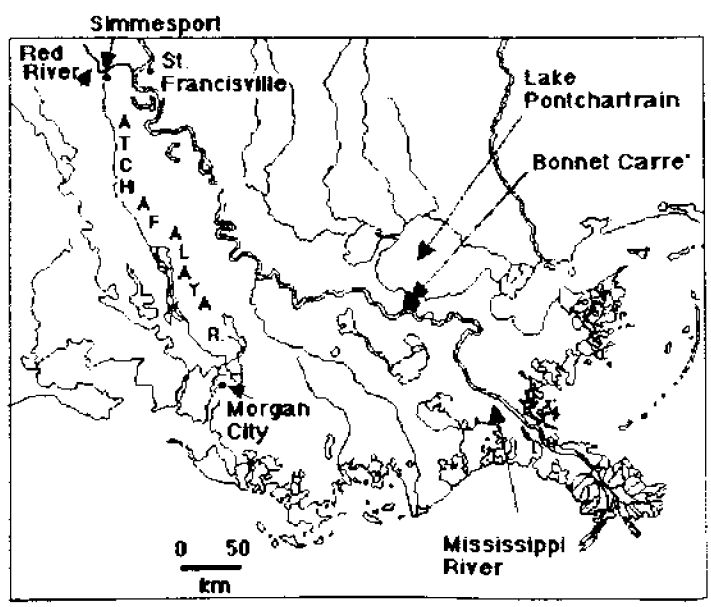

Fig. 1. The location of places mentioned in the text.

March, April and October, respectively. The spillway would be operated in perpetuity. The purpose of these diversions is to influence the salinity regime in the receiving waters and perhaps positively affect fisheries and wildlife habitat, restore and create wetland habitats. These projects have unclear or scientifically contentious effects on wetland restoration (Turner 1997) and swamp forest productivity (Megonigal et al. 1997). If such diversions could reduce the concentration of nitrogen and other nutrients in water flowing over them before reaching the lake, then the occasional noxious and dramatic algal blooms sustained by nutrients reaching the lake might be less of a threat, and the project appear more appealing. A central question of this analysis is how much nitrogen could be removed from overlying waters during an average diversion and with how much land area.

Methods

I analyzed data from monthly water quality monitoring stations at the upstream (northern) end of the basin at Simmesport, LA, and also at the downstream (southern) end south of Morgan City, LA (Fig. 1). Both cities had a population of less than 15,000 in 1990. The sampling program is organized by the Department of Environmental Quality, State of Louisiana. Data for various forms of nitrogen were for monthly sampling trips, which were usually within a few days of each other. Water takes less than 7 days to travel from Simmesport to Morgan City during normal river stages. I used data from these two stations to complete a comparison of the concentrations of various forms of nitrogen for the period 1980 to 1993. Subsets of the data were compiled that included only data for when the Atchafalaya River would be out of its main channel. The nitrogen loading rate from the human population living the Atchafalaya was estimated at 4.4 kg N y^{-1} by using per capita estimates (Vollenweider 1968; cited in Howarth et al. 1996) and a rather generous population estimate of 100,000 persons living within and around the basin. An average discharge for the Atchafalaya River of 17,545 m^3 s^{-1} was used to calculate average loading rates, which is the average 1980 to 1990 flow measured by the US Geological Survey at Simmesport, Louisiana.

Fertilizer use within the basin on agricultural land is certainly less than 10% of the basin's surface area (based on an inspection of aerial photographs), but the precise percentage is not available. I estimated the fertilizer input to the basin by using an average fertilizer use for cropland in the Gulf of Mexico watersheds for 1987 (Turner and Rabalais 1999) and applying this to 10% of the basin's area.

Water from the Mississippi River leaks into the Bonnet Carré spillway when the riverstage rises above the river levee at the spillway structure. The volume of this leakage is empirically related to the height of the river, and described by an equation developed by the New Orleans District, US Army Corps of Engineers. Water samples were collected within the spillway at two places during two leakage periods lasting 3 to 4 weeks each. One sampling location was on a road running parallel to the spillway, near where riverwater enters the spillway. The other location was at the railway crossing just before water leaves the spillway and enters Lake Pontchartrain (Figure 1). Samples were collected 8 times in spring, 1998, at discharges ranging from 9.4 to 55.5 m^3 sec^{-1}. These samples were frozen and later analyzed for dissolved inorganic and organic constituents using EPA Methods 350.1, 350.2, and 353.2.

The North American Wetland Treatment System Data Base (NAWTDB) and Danish Wetland Treatment System Database (DWTSD) summarized in Kadlec and Knight (1996) include information on nutrient loading rates, concentration, volume, and other pertinent information for 189 wastewater treatment systems in North America and Europe. These data were compared with the nitrogen removal rates for the Atchafalaya basin developed in this paper, and other examples for Louisiana from literature sources.

Kadlec and Knight (1996; Example 13-1, page 437) provide a simple formula to estimate the land necessary to remove various percentages of total nitrogen in waters flowing over wetlands. This formula was applied by using an average total nitrogen concentration in the Mississippi River (from 1980 to 1990).

Results and Discussion

Atchafalaya Basin

A comparison of the concentration of nitrate, Kjeldahl nitrogen and total nitrogen (Kjeldahl nitrogen + nitrate) at the upstream (Simmesport) and downstream (Morgan City) monitoring stations is shown in Fig. 2. Table 1 provides the average concentrations for all years and the ratio of nutrients at the upstream and downstream station for paired samples. The nitrate concentrations do not change noticeably from upstream to downstream locations, and the paired samples show a slight increase (4%) at Morgan City. The total nitrogen concentration decreases slightly (6%), primarily because the Kjeldahl nitrogen concentration is reduced by 14%. There was no difference in the result if the data set was constrained to samples taken during flood stage, or below flood stage.

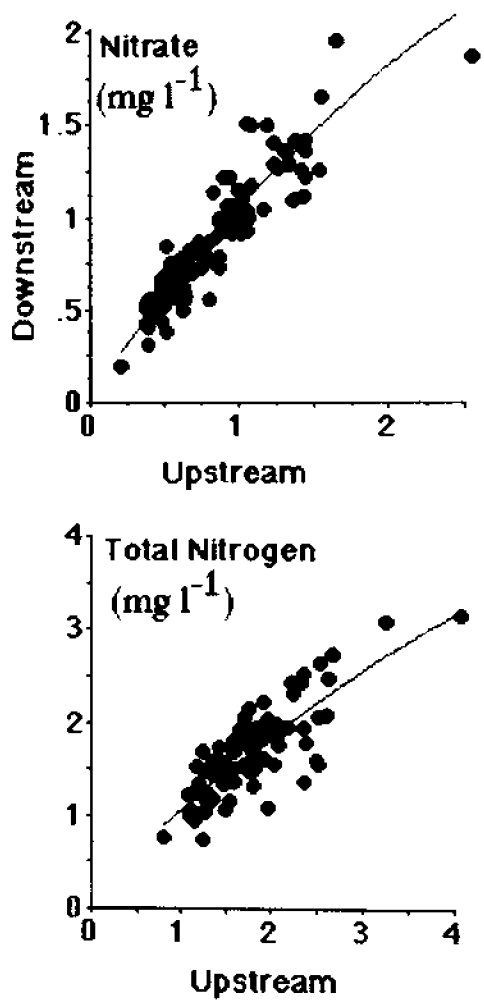

Fig. 2. The concentration of nitrate and total nitrogen (total Kjeldahl nitrogen + nitrate; mg l^{-1}) at Simmesport, Louisiana (upstream) and Morgan City, Louisiana (downstream). The data are from 1980 to 1993 and are collected on a mostly monthly basis. A polynomial fit of the data is shown.

The contribution of sewerage from the population in the basin (4.4 kg N y^{-1} per person) is only 0.07% of the total nitrogen flux through the basin, and thus cannot be the source of the excess nitrate downstream. If all fertilizer applied in the basin was leached as nitrate into the river, then that loading would equal about 1 % of the nitrate flowing into the basin's northern end. This is certainly a high estimate, because not all of the applied fertilizer will enter the river. Howarth et al. (1996), for example, estimated that only about 20% of the applied fertilizer nitrogen ends up as nitrate in large rivers. The approximate 4 % gain in nitrate in water passing through the swamp, not statistically significant, does not appear to be a result of nitrate additions from sewerage or fertilizer released into the basin.

The total nitrogen loading for the Atchafalaya basin was calculated using an average flow and concentration. This value was compared to loadings for the wastewater treatment systems listed in the NAWTDB and DWTSD (Fig. 3). The concentration of TN entering the Atchafalaya River basin is about at 1.75 mg N l^{-1}. The TN load to the Atchafalaya Basin is about 0.38 g N m^{-2} d^{-1}. The proposed diversion through 10,000 ha of a proposed new Bonnet Carré diversion will result in a TN load of 0.39 g N m^{-2} d^{-1}. Compared to the NAWTDB and DWTSD systems, the natural overland flow system in the Atchafalaya basin represents a high load for a surface flow system (but low load for a subsurface flow system), and has a relatively low inflow concentration. This is an important observation, for it reveals distinctions between the hydrology of engineered wastewater systems and natural systems. The Atchafalaya basin loading rates are relatively high, although for water with relatively low nitrogen concentrations, because the water discharge is relatively high. Further, there is generally an inverse relationship between the percentage of nutrients

Table 1. The concentration of nutrients (mg l^{-1}) in the Atchafalaya River at Simmesport, LA. (upstream) and Morgan City, LA (downstream) for 1980 to 1993. Data are from various agency water quality monitoring records (the numbers in the parentheses is for the sample size (n) and ±1 standard error; n/S.E.). The column for 'Change' is for the difference in the average values for paired samples (the number of paired samples is in parentheses).

	Simmesport (upstream)	Morgan City (downstream)	Change (percent)
Total Kjeldahl Nitrogen (TKN) (mg N l^{-1})	0.90 (97/0.030)	0.77 (121/0.019)	-14% (89)
Nitrate (NO_3) (mg N l^{-1})	0.86 (99/0.040)	0.90 (118/0.036)	+4% (89)
Total Nitrogen (TN = TKN+NO3) (mg N l^{-1})	1.79 (92/0.053)	1.66 (115/0.047)	-6% (89)

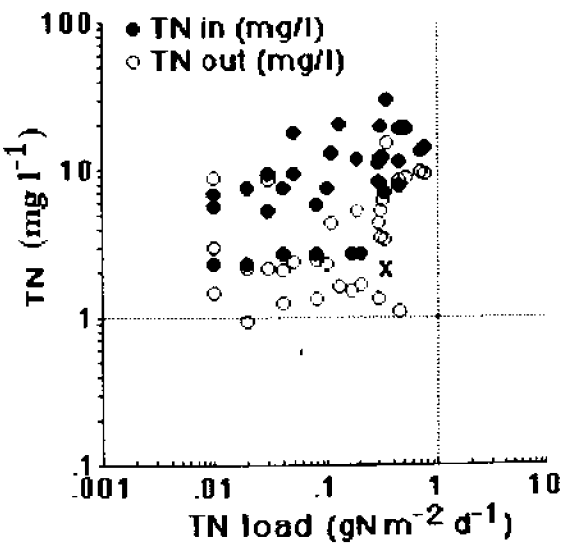

Fig. 3. The relationship between the total nitrogen (TN) concentration (mg N l^{-1}) in the inflow and outflow of surface flow wetland waste water treatment systems compared to the TN load (g N m^{-2} d^{-1}). The data are from Kadlec and Knight (1996). The concentration of TN entering the Atchafalaya River basin is about 1.75 mg N l^{-1}. The TN load to the Atchafalaya Basin is about 0.38 g N m^{-2} d^{-1} and the proposed diversion through 10,000 ha of a new Bonnet Carré diversion will result in a TN load of 0.39 g N m^{-2} d^{-1} (marked with an 'X' in the figure).

retained within a system and loading rates, because key ecological pathways become saturated at high loading rates. Wastewater managers, for example, design and operate their systems to avoid the inefficiencies of high loading rates.

The total nitrogen load for surface water wastewater treatment systems and the removal rate are shown in Fig. 4 for the NAWTDB and DWTSD and the Atchafalaya basin. The uptake rates for the basin are clearly very low in comparison to the engineered systems.

The average loss of nitrogen leaving the Atchafalaya River basin (0.02 g N m^{-2} d^{-1}) is about one-half the average fertilizer application rate for cropland in the Gulf of Mexico estuarine watershed (0.013 g N m^{-2} d^{-1}; Turner and Rabalais 1999). Thus the nitrogen retained by the ecosystem, although a low percentage of the total amount flowing through the system, can be a relatively significant nutrient source if the ecosystem is nitrogen limited.

Bonnet Carré

A plot of the percent retention of nitrate, Kjeldahl nitrogen, and total nitrogen (Kjeldahl nitrogen + nitrate) versus water volume flowing across the Bonnet Carré is in Fig. 5. The spillway

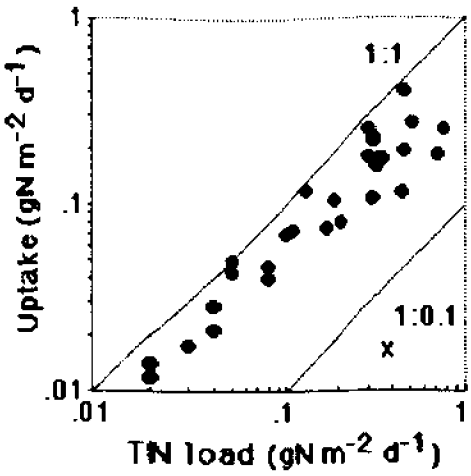

Fig. 4. The total nitrogen (TN) load (g N m^{-2} d^{-1}) for surface water wastewater treatment systems and the uptake rate (g N m^{-2} d^{-1}). The data are from the NAWTDB and DWTSD presented in Kadlec and Knight (1996). The TN load for the Atchafalaya and proposed Bonnet Carré is 0.38 and 0.39, respectively, (g N m^{-2} d^{-1}). The removal rate for the Atchafalaya system is 6% of the total N load, or 0.019 (g N m^{-2} d^{-1}) and shown with an 'X' on the figure.

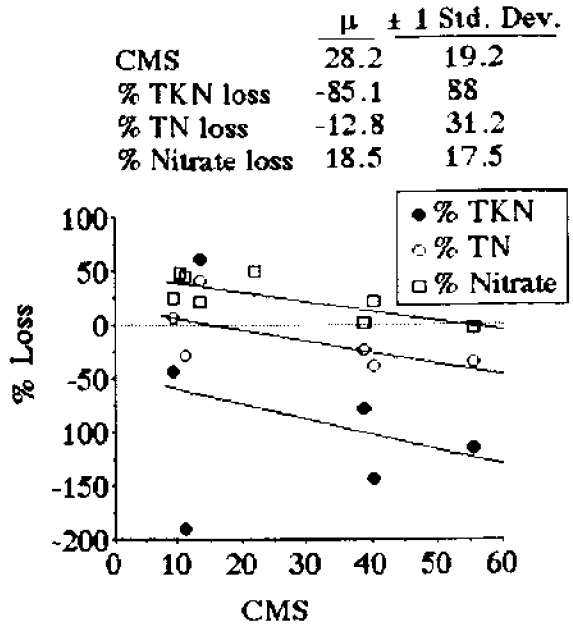

Fig. 5. The relationship between nutrient retention (total Kjeldahl nitrogen (TKN), total nitrogen (TN), and nitrate) and the discharge volume (m^3 sec^{-1}; CMS) of Mississippi River water leaking through the Bonnet Carré spillway in spring, 1998. Negative reduction means that there was a net release. The mean ± 1 Standard Deviation are shown for the variables.

discharge resulted in a comparatively low nitrogen loading rate of < 0.01 g N m^{-2} d^{-1}. The uptake of nitrate in 7 of 8 examples, contrasted with the net release of TKN and TN in 5 of 6, and in 4 of 6 examples, respectively. There was a general increase in retention with lower loading rates. This inverse relationship between nutrient retention and nutrient loading rate is commonly described in the literature. What is not generally appreciated is that some natural overland flow systems release nutrients, rather than take them up. Devito et al. (1989), for example, reported that nitrogen retention rates for five Ontario wetlands were insignificant to negative, and Soranno et al. (1996) showed that some alluvial systems may release phosphorus. The landscape scale is important, too. Arheimer and Wittgren (1994), make the point that nitrogen removal efficiency per area tends to decrease with increasing watershed size. In some cases, what appears to be denitrification, may be the result of groundwater dilution (e.g., Pinay et al. 1998).

Proposed Bonnet Carré River Diversions

A graph prepared by Richardson and Nichols (1985) has been used to estimate the area necessary to remove 50% of the nitrogen load from a proposed diversion of Mississippi River into Lake Pontchartrain (Day 1997). The average concentration of nitrogen in the Mississippi River (mg l^{-1}) and the proposed diversion amount (CMS) were used to estimate that 10,000 ha were needed to reach the loading rate (g N m^{-2} y^{-1}) where 50% of the nitrogen was retained in the Richardson and Nichols (1985) graph. The graph Richardson and Nichols (1985) used was based on data presented by Nichols (1983) that showed the relationship between the percentage nitrogen removed during overland flow and the loading rate of wastewater (sewerage secondary effluent) over 'natural' freshwater wetlands. Nichols (1983) nicely summarized the data and offered examples of net uptake and release of nutrients that depended on hydrologic flows, nutrient loading history, and soil

chemistry. The original data used by Nichols (1983) were based on 8 temperate and 1 Florida wetlands that were in peat soils (i.e., none had the inorganic minerals of the Mississippi River). The data were from 1977 or earlier, and most data were for the period of maximum vegetative growth. The Nichols (1983) data set might be used to provide a preliminary estimate of the nitrogen removal rates for a proposed Mississippi River diversion, but further analysis seemed useful for when one considers the cost of the proposed Bonnet Carré diversion ($US 214 million). I used the data generated here to provide an alternative estimate of the relationship between the loading and removal of nitrogen for the project area.

A nitrogen budget was prepared for the Des Allemands Swamp by Kemp and Day (1985) that has data that could be used to estimate the net amount of nitrogen and phosphorus retained as water flowed through a swamp system. The nitrate and ammonium concentration water flowing through the swamp initially dropped and then began to rise, especially ammonium. However, the average of all measurements of total nitrogen and phosphorus concentration at 8 stations within the swamp were essentially identical, indicating that there was no net loss in these two elements. Kemp and Day's estimate of the net loss of TN and TP was explained as the result of atmospheric deposition, which was estimated with *in situ* sampling. The average total nitrogen concentration in the swamp water was about 1 mg l^{-1}, and in the rainfall, 0.35 mg l^{-1}. The logic of adding the TN and TP weight in rain, but not the volume in rainfall was incorrect (incomplete) mathematical summation of inputs. The TN and TP arrives in a lower concentration in rainwater than in the receiving water, so the addition of TN and TP as rainwater should dilute the swamp water concentration. Further, they did not consider the amount of TN and TP that was intercepted by vegetation before it reached the swamp water. In effect, there was no net TN or TP uptake. There may even have been a net release of TN and TP as water flowed into and out of the study area. These data were compared to those collected during an experimental release of river water through the Bonnet Carré spillway. There was negligible uptake of either nitrogen or phosphorus during various Mississippi River diversions into Lake Pontchartrain (Demcheck et al. 1996). The data generated by the analysis for the Atchafalaya basin have a lower retention rate and a higher loading rate. A comparison of these data with those of Nichols (1983) suggests that a 20% reduction in nitrogen loading is possible for a Bonnet Carré diversion with spillway of 100,000 ha. However, the analysis of short-term releases of water over the existing Bonnet Carré spillway suggests that the nitrogen uptake in diverted water may be negligible, or, if it occurs, will be at very, very low loading rates.

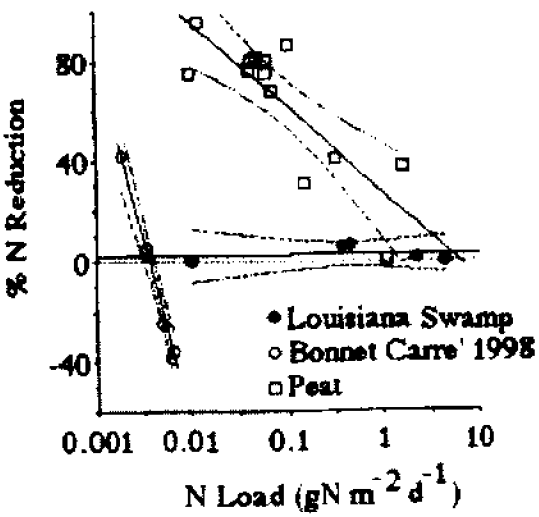

Fig. 6. The relationship between the percentage reduction and loading rates for TN for several Louisiana overland flow swamp sites (filled circles), wastewater systems of natural peat soils (unfilled squares) and leakage through the Bonnet Carré spillway in 1998 (unfilled circles). Data are from this study (Atchafalaya basin), the Bonnet Carré spillway (Demcheck et al. 1996; data collected as part of this study), Nichols (1983) and from Day and Kemp (1985). A linear fit of the transformed data is shown, together with the 95% confidence intervals for each data set. Negative reduction means that there was a net release.

Kadlec and Knight (1996; example 13-1, page 437) provide a formula that may be used to estimate the land area necessary to remove nitrogen at different retention percentages in surface flow over wetland wastewater treatment systems. Kadlec and Knight (1996) make the point with this formula, which is empirically derived, that overland flow systems operating at the lower concentrations (e.g., natural systems) behave differently than wastewater

treatment systems. I changed one input variable to use it, because the background rate that Kadlec and Knight used (1.5 mg N l^{-1}) was higher than after the targeted 50% reduction in the proposed new diversion (i.e., it was changed from the observed 1.76 to 0.88 mg N l^{-1}). In other words, the formula suggests that a 50% reduction in nitrogen cannot be achieved at the existing conditions. If the background rate is lowered to 0.75 mg N l^{-1}, then the area necessary to achieve a 50% reduction in TN is 129,605 ha. A 25% reduction in TN load and a 1.25 mg N l^{-1} background concentration would require 106,450 ha using the same assumptions.

There is clearly a lower retention rate for riverwater flowing over riverine swamps than for wastewater flowing over and through peaty wetlands (Fig. 6). A plausible explanation for this difference is that the peaty systems have more of the nutrient load flowing belowground, compared to the alluvial systems, and therefore a higher nitrogen retention rate.

Furthermore, the natural systems have a relatively very low nutrient concentration compared to that in wastewater, which makes the uptake mechanisms in the soil less efficient at stripping out elements from the overlying water.

Nitrogen Loading and Low Oxygen Zones Offshore

These results can be used to evaluate the influence of flood protection levees, which reduced overland flow, on the flux of nitrogen to the coastal zone and subsequent formation of low oxygen zones. Nutrient loading in the Mississippi River has been linked to the causes of the low oxygen zones of 18,000 km^2 forming near the Mississippi River delta on the continental shelf in most summers (Rabalais et al. 1996). Organic matter from phytoplankton production sinks to bottom waters consuming the oxygen to form these zones under stratified conditions. The phytoplankton production is (at present river concentrations) limited by the riverine nitrogen loading (Turner and Rabalais 1994), which have doubled in the last 30 years (Turner and Rabalais 1991). Only 2-3 % of the annual Mississippi River discharge flowed overbank before engineering works (Kesel 1988). Therefore, a 6 % reduction in the total nitrogen load (from Table 1) in the ≈ 3% of the riverwater no longer flowing overland is a very small change in the nitrogen loading to the sea. The results of these analyses do not support the view that constraining the Mississippi River by flood protection levees caused low oxygen zones to form offshore, or last longer, or be more severe.

Conclusions

The nitrogen loading rates of the alluvial system examined here are relatively high, but the N % retention and nitrogen concentration are relatively much lower than observed for wastewater treatment systems with surface flows. Estimates of nitrogen removal rates for natural systems that are based on wastewater treatment systems are therefore inappropriate. Peaty wetland systems seem to have a much higher rate of nitrogen retention than alluvial systems, which may release nitrogen during flooding events of a few weeks, or less. The prospects that a significant percent uptake of nitrogen as water flowing over large man-made diversion spillways will occur are unsubstantiated, if not rejected. The meager levels of nitrogen retention that will occur if these diversions are built, implies a significant increase in nitrogen loading to the receiving basin. Finally, re-routing Mississippi River floodwaters over the presently-leveed swamplands and former floodplains, although desirable for other reasons, will not significantly reduce nitrogen loading to the continental shelf.

ACKNOWLEDGMENTS

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A Review of Recent Studies of the Ecological and Economic Aspects of the Application of Secondarily Treated Municipal Effluent to Wetlands in Southern Louisiana

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ABSTRACT: Insufficient sedimentation, coupled with high rates of relative sea level rise (RSLR), are two important factors contributing to wetland loss in coastal Louisiana. We hypothesized that adding nutrient-rich treated wastewater effluent to selected coastal wetlands results in four benefits: (1) improved effluent water quality; (2) increased accretion rates to help offset subsidence; (3) increased productivity of vegetation; and (4) financial savings of capital not invested in conventional tertiary treatment systems. To test these hypotheses, we are currently monitoring several forested wetlands that are receiving secondarily treated wastewater in coastal Louisiana. At one site where sedimentation accumulation was measured, rates of accretion increased significantly after wastewater application began in the treatment site (from 7.8 to 11.4 mm yr^{-1}), and approached the estimated rate of regional RSLR (12.0 mm yr^{-1}). No corresponding increase was observed in an adjacent control site. In the same site, surface water nutrient reduction, from the effluent inflow to outflow (1600 m), ranged from 100% for NO_3-N to 66% for total P. At another site, a forested wetland that has been receiving wastewater effluent for 40 years, dendrochronological analysis revealed that stem growth increased significantly in the treatment site after wastewater water applications began, and was significantly greater than an adjacent control. Preliminary results indicate that these sites have the potential to assimilate all effluent nitrogen and varying percentages of phosphorus. Results of avoided costs analyses to evaluate the economic implications between conventional treatment and wetland treatment at three sites indicate savings range from $500,000 to $1.5 million.

Introduction

Numerous studies have shown that wetlands can be effective tertiary processors of wastewater effluent (Kadlec and Knight 1996). Previous studies indicate that both natural and constructed wetlands have been successfully used to purify effluent (Richardson and Davis 1987; Conner et al. 1989; Reed 1991; Kadlec and Knight 1996). Wetlands are efficient at removing excess nutrients and pollutants by physical settling and filtration, chemical precipitation and adsorption, and biological metabolic processes that result in burial,

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storage in vegetation, and denitrification (Conner et al. 1989; Kadlec and Alvord 1989; Patrick 1990). These wetland functions can be especially critical for the coastal regions in Louisiana affected by degraded water quality caused, in part, by inadequate sewage treatment (Louisiana DEQ 1988).

Wastewater effluent may also serve as a restoration tool in coastal wetlands impacted by high rates of relative sea level rise (RSLR). Wetlands have been shown to persist in the face of RSLR when vertical accretion and elevation gain equals or exceeds the rate of subsidence (Delaune et al. 1983; Cahoon et al. 1995). Historically, seasonal overbank flooding of Mississippi river deposited sediments and nutrients into the wetlands of the delta plain. Not only did these floods provide an allochthonous source of mineral sediments, which contributed directly to vertical accretion, but the nutrients associated with these sediments promoted vertical accretion through organic matter production as well as deposition (Nyman and Delaune 1991). This sediment and nutrient source to most forested wetlands in coastal Louisiana has been eliminated since the 1930s with the completion of levees along the entire course of the lower Mississippi River, resulting in vertical accretion deficits (accretion < RSLR), prolonged periods of inundation, lowered productivity and a lack of regeneration (Conner and Day 1988).

In these stressed systems, we hypothesize four primary benefits derived from wetlands wastewater treatment in Louisiana: (1) improved effluent water quality; (2) increased accretion rates; (3) increased productivity of vegetation; and (4) the financial savings of capital not invested in conventional tertiary treatment systems (Breaux 1992; Breaux and Day 1994). The high rate of burial due to subsidence and higher than national average rates of denitrification due to warm temperatures are additional reasons for the use of wetland treatment in Louisiana. Increasing vegetative productivity is especially crucial in many parts of Louisiana where coastal subsidence in the Mississippi Delta results in a relative sea level rise nearly ten times greater than eustatic sea level rise (Conner and Day 1988; Penland et al. 1988). Increasing productivity results in greater root production leading to organic soil formation which can enhance the accretion necessary to offset the subsidence that is contributing to wetland loss.

Since 1988, the Coastal Ecology Institute at Louisiana State University has been working with the U.S. Environmental Protection Agency (EPA), the Louisiana Department of Environmental Quality (DEQ), and several dischargers to assess the impact of forested and marsh wetland wastewater assimilation projects in coastal Louisiana (for a general policy review see Breaux and Day 1994). The dischargers include two municipalities, Thibodaux and Breaux Bridge, and one food processor, Zapp's potato chip factory in Gramercy (Fig. 1). Two additional municipalities (Amelia and St. Bernard) are in the final stages of the EPA-mandated Use Attainability Analyses prior to formal discharge permitting. A feasibility study was recently conducted for wetland treatment of shrimp processing effluent in Dulac, Louisiana. All of the potential and actual receiving wetlands have been hydrologically altered by some combination of levees, spoil banks, highways, oil and gas access roads, or railroad lines. In addition, prior to wetland treatment, all effluent was discharged directly into open water bodies. Wetland discharge provides additional treatment by removing further nutrients from the effluent before entering open water bodies.

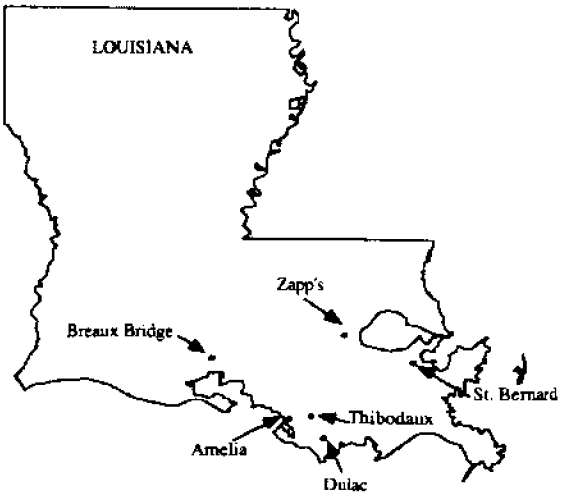

Fig. 1. Wetlands treatment study sites.

To examine the effect of wetland treatment on effluent water quality, sediment accretion, productivity, and economic savings, we review results of studies conducted at Amelia, Breaux Bridge, Dulac, Thibodaux and St. Bernard (Conner and Day 1989; Day et al. 1994; Day et al. 1997a,b; Day et al. 1998; Cardoch et al. 2000). Throughout this review, when we use the term significant, it implies a statistical significance which is documented in the references. Sampling of benthic invertebrate and nekton communities did not indicate any clear effects due to wastewater discharge and results are not presented here. For more information, refer to Conner and Day (1989), Breaux and Day (1994), Day et al. (1994), Day et al. (1997a), Day et al. (1997b) and Pratt (1998). Additional project information can be found in Breaux (1992), Hesse (1994), Delgado-Sanchez (1995), Zhang X. 1995, Blanick (1997), Boustany et al. (1997), Rybczyk (1997).

Improved Effluent Water Quality

We hypothesized that effluent water quality will be improved through efficient nutrient uptake and removal pathways within forested wetlands. Loading rates and percent nutrient reductions for municipal wastewater treatment wetlands are listed in Table 1. Data from the Point au Chene treatment wetland for the City of Thibodaux offers an example for the impact of effluent on water quality.

The Thibodaux site consists of two almost permanently flooded, subsiding, forested wetlands, separated by a slightly elevated bottomland hardwood ridge. Since 1992, the 231 ha wetland on the western side of the ridge has received secondarily treated municipal wastewater at the average rate of 15,140 $m^3 d^{-1}$. The wetland on the eastern side of the ridge, which is not impacted by the effluent, serves as a control site. Baseline monitoring of vegetation, soils, surface water, hydrology, and fauna, at both sites, began in 1988. Extended inundation was documented during the baseline studies (Conner and Day 1989). A comprehensive site description is provided by Breaux and Day (1994) and Rybczyk et al. (1995).

Measurements taken at Thibodaux by Zhang (1995) indicate that effluent water quality was improved as nutrients were significantly reduced and assimilated. The effluent stream was highly nitrified, with NO_3-N being the dominant form of N and soluble PO_4-P accounting for about 77% of the total P in the effluent. After passage through the treatment swamp the concentrations of many water quality parameters at the output station were significantly reduced compared with the influent concentrations. From 1992 through 1996, the mean annual reduction (from inflow to outflow) of NO_3-N the dominant form of nitrogen in the effluent, ranged from 96% to 99% (Fig. 2). At the output station, the NO_3-N concentration was below the detection limit (<0.1 mg l^{-1}) during most sampling periods, indicating that the swamp system was removing NO_3-N. Figure 3 illustrates reductions of NO_3-N concentrations as a function of distanced traveled in the swamp. Within 800 m, concentrations were comparable to those found in the control site. NO_3-N was taken up by growing plants, immobilized to organic N, or removed by denitrification (Boustany et al. 1997). Concentrations of total P in the treatment site varied during the study period. From 1992 through 1994, the mean annual reduction of total P in the treatment site ranged from 33% to 71% from inflow to outflow (Fig. 4, Zhang 1995).

Zhang (1995) described the effects of wastewater effluent on effluent water quality, sediment nutrient concentrations, and the chemical composition of floating aquatic vegetation at the Pointe au Chene site. This study assessed the long term ability of the swamp to treat secondarily treated wastewater effluent from the city of Thibodaux. In general Zhang found that, within the immediate 231 ha treatment zone, N and P concentrations in the water were reduced 100% and 66%, respectively, from effluent inflow to outflow Table 1. In a related review, Rybczyk et al. (1996) concluded that the effective tertiary processing of effluent at this site could be attributed to the following: (1) The dominant species of N in the effluent was the oxidized NO_3-N form and not the reduced species, NH_4-N. These naturally dystrophic wetlands readily denitrify NO_3-N, resulting in a net loss of N to the system as N_2 or N_2O gas (see Boustany et al. 1997);

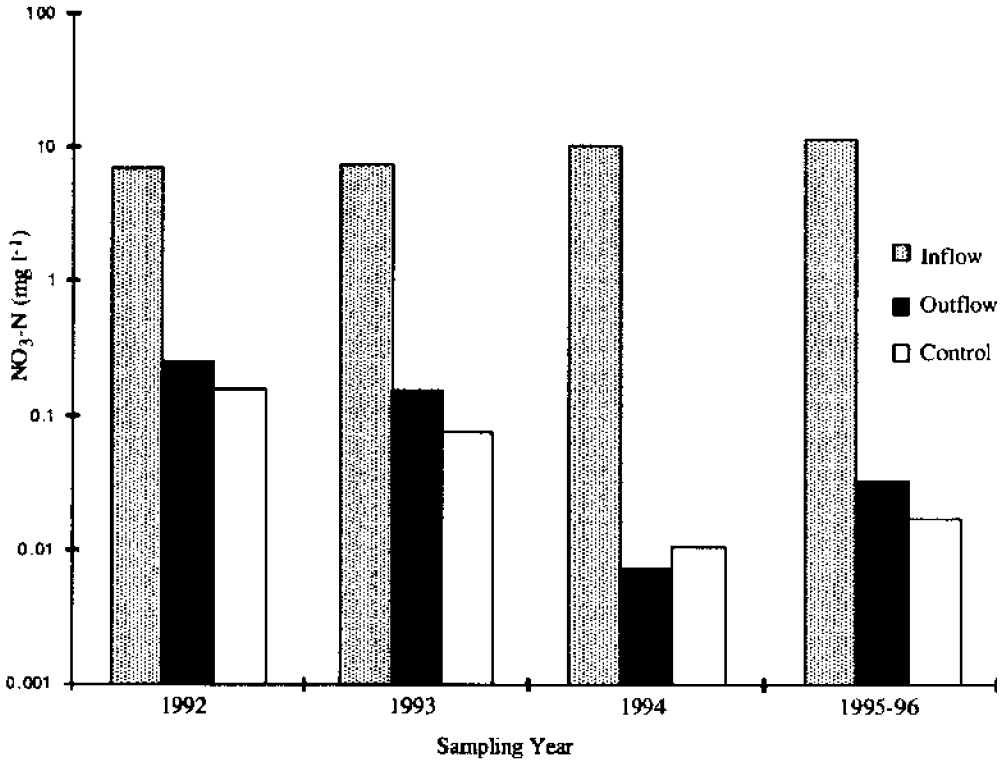

Fig. 2. Mean annual concentrations of NO_3 for the effluent inflow pipe, the treatment outflow, and within the first 100 m of the control site. Inflow concentrations are reduced 96-99%. Note: Logarithmic scale.

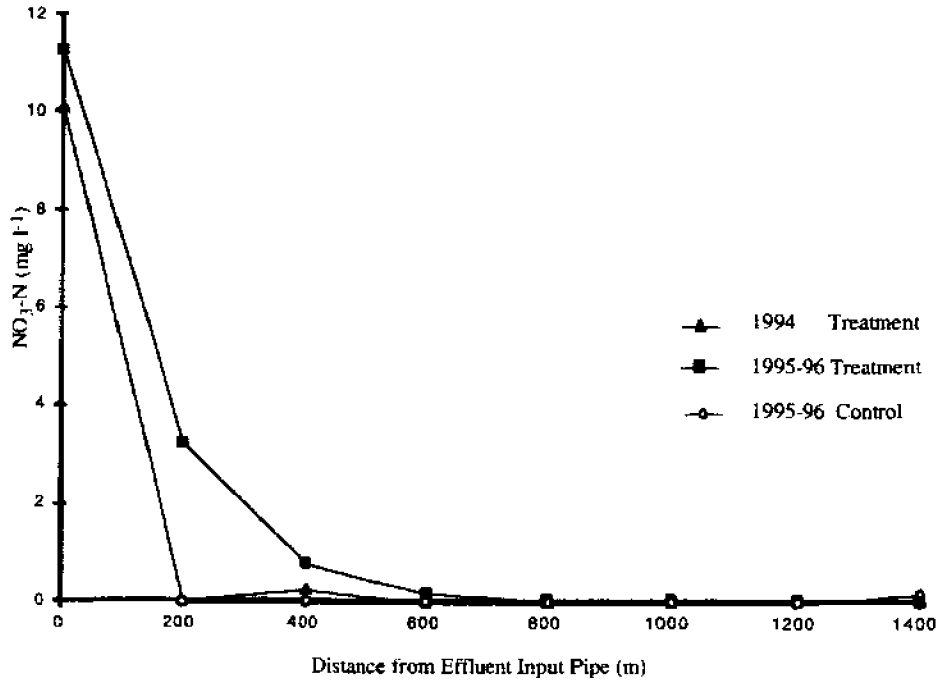

Fig. 3. Mean concentrations of NO_3-N along sample transects in the treatment site during sample years 1994 and 1995, and in the control site for sample year 1995. Nitrate concentrations are reduced 99%.

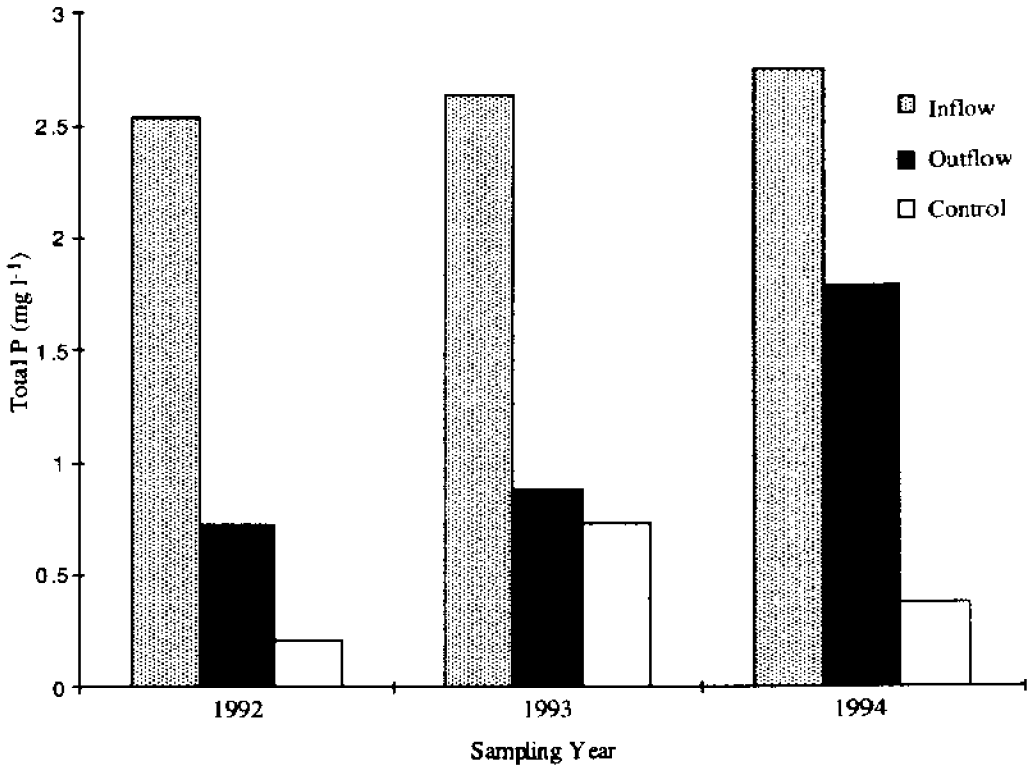

Fig. 4. Mean annual concentrations of Total P for the effluent inflow pipe, the treatment outflow, and within the first 100 m of the control site. Inflow concentrations are reduced 35-71%.

(2) Loading rates are low compared to other wetlands treatment sites. For example, the State of Florida has adopted regulations for wetland wastewater management that established maximum P loading rates of 9 $gm^{-2} yr^{-1}$ for hydrologically altered wetlands (Harvey 1988), an order of magnitude higher that at most of our sites, and; (3) High rates of accretion and burial of sediments in these subsiding systems provide a permanent sink for phosphorus. Two other studies also documented the high rates of denitrification at this site (Crozier et al. 1996; Boustany et al. 1997).

Similar water quality improvements have been documented for the treatment wetlands at Amelia, Breaux Bridge, and St. Bernard (Table 1). High reduction rates of N and P indicate that the wetlands there act as a net nutrient sink and that the sites are effective providers of tertiary treatment. For comparison, in Florida, the tertiary advanced waste treatment (AWT) standards for total N and total P are 3 mg l^{-1} and 1 mg l^{-1}, respectively. For many of these sites, nutrient concentrations have been well below that limit, indicating that tertiary treatment was achieved .

Increased Sediment Accretion

Current evidence indicates that rising water levels are leading to wetland loss, coastal erosion, and salt water intrusion in a number of coastal areas (Stevenson et al. 1988; Sestini 1992). If coastal wetlands, especially those in deltas, do not accrete vertically at a rate equal to the rate of RSLR, they will become stressed due to waterlogging and salt stress, and ultimately disappear (Mendelssohn and McKee 1988). Many wetlands in Louisiana suffer from accretion deficits; that is, they are not keeping pace with RLSR. Discharge of secondarily treated effluent can stimulate biomass production and enhance sediment accretion rates. Maintaining vegetation is crucial to wetlands survival.

Table 1. Loading rates and percent nutrient reductions in at four wastewater treatment forested wetlands in coastal Louisiana. All concentrations are reported as mg l^{-1}.

Site	Treatment Basin (ha)	Nitrogen Loading ($g\ m^{-2}yr^{-1}$)	Phosphorus Loading ($g\ m^{-2}yr^{-1}$)	Nutrient	Effluent Concentration Discharge	Outlet	% Reduction
Amelia[1]	1012	9.8 to 19.6	1.1 to 2.1	TKN	2.98	1	66[5]
				Total P	0.73	0.06	92[5]
Breaux Bridge[2]	1475	1.87	0.94	NO3-N	0.8	<0.1	100
				PO4-P	1	0.2	80
				Total P	2.9	0.3	87
St. Bernard[3]	1536	2	0.42	TKN	13.6	1.4	89.7
				Total P	3.29	0.23	95
Thibodaux[4]	231	3.1	0.6	NO3-N	8.7	<0.1	100
				TKN	2.9	0.9	69
				PO4-P	1.9	0.6	68
				Total P	2.46	0.85	66

[1]Day et al. (1997a).

[2]Day et al. (1993).

[3]Day et al. (1997b).

[4]Zhang (1995).

[5]Reductions calculated for averages of stations T1 and LW stations for 4/96 sampling (Day et al. 1997a). There was no consistent different between the treatments and control sites at Amelia.

Subsidence in deltas leads to a relative sea level rise (RSLR) rate that is often much greater than eustatic rise. For example, while the current rate of eustatic rise is between 1-2 mm yr^{-1} (Gornitz et al. 1982), RSLR in the Mississippi delta is in excess of 10 mm yr^{-1}, thus eustatic sea level increase accounts for only 10-15% of total RSLR in this delta. We hypothesized that adding nutrient rich effluent can increase rates of sediment accretion by promoting production of organic matter and trapping of mineral matter. Evidence from the Pointe au Chene treatment wetland at Thibodaux supports this hypothesis.

The relative sea level rise (RSLR) rate at the Thibodaux treatment wetland (Fig. 5) derived from tidal gauge analysis (Penland et al. 1988), is 1.23 cm yr^{-1} for the period 1962 through 1982. To maintain elevation, soil accretion must equal this rate of RSLR. However, by analyzing ^{137}Cs activity in the soil profile (see Delaune 1978 for a description of methods), it was estimated that background accretion rates in the Thibodaux site averaged only 0.44 ± 0.04 cm yr^{-1} for the same period, leading to an accretion deficit of 0.79 cm yr^{-1} (Rybczyk 1997).

To determine whether wastewater applications stimulated accretion, a feldspar horizon marker technique (Cahoon and Turner 1989) was utilized to estimate accretion rates in the site receiving effluent and in an adjacent control site, both before (1988 - 1991) and after (1992 - 1994) wastewater applications began in the treatment site. Pre-effluent accretion rates averaged 0.78 cm yr^{-1} in the treatment site and 0.52 cm yr^{-1} in the control site and were not significantly different (Fig. 1). After application began, accretion rates in the treatment site (1.1 cm yr^{-1}) were significantly higher than accretion rates measured at the control site (0.14 cm yr^{-1}).

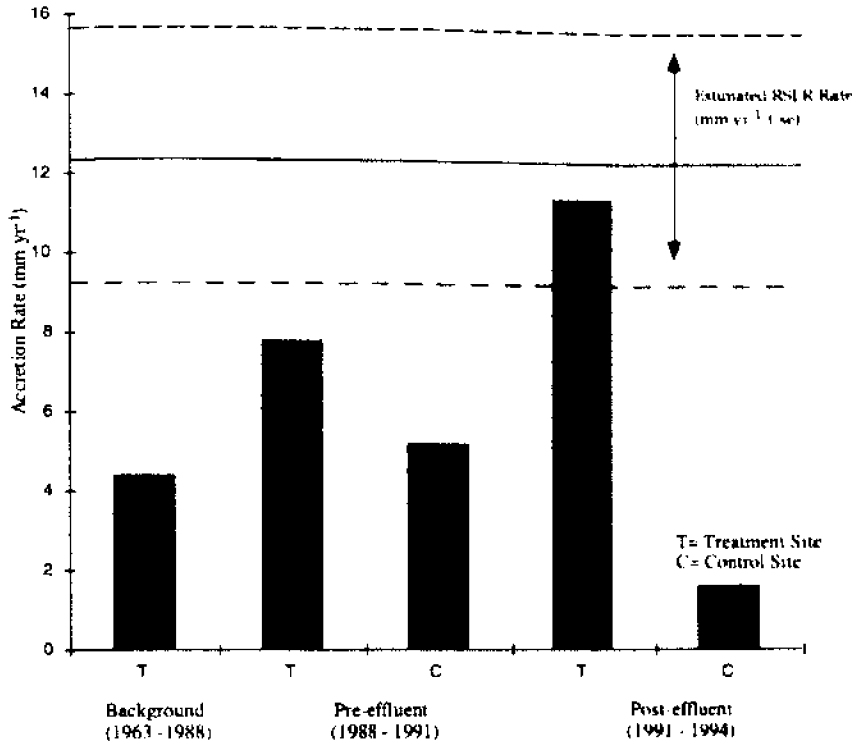

Fig. 5. Background (from ^{137}Cs measurements), pre-effluent, and post-effluent accretion rates, relative sea level rise (RSLR) and accretion balance deficits for treatment (T) and control (C) at the Thibodaux site. T and C were not significantly different during the pre-treatment period, but T was significantly greater during the post-effluent period.

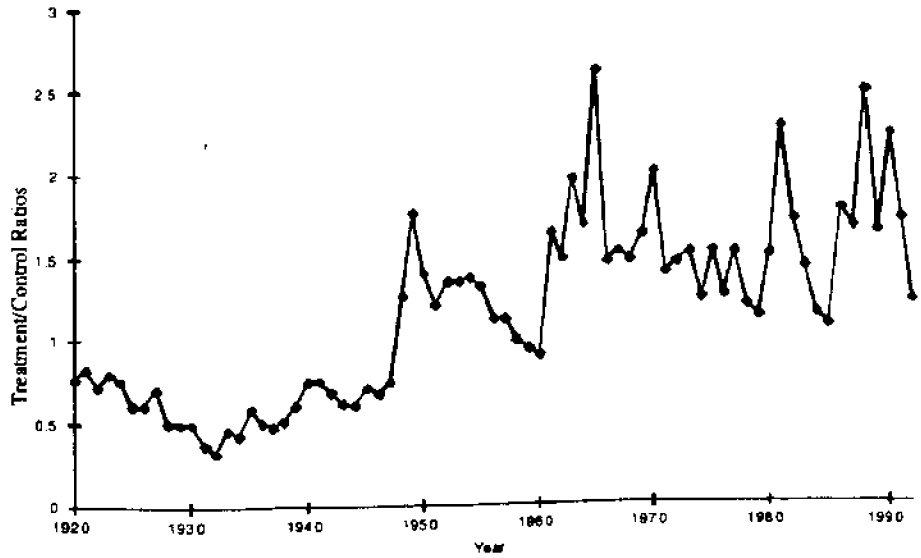

Fig. 6. The ratio (treatment/control) of annual wood stem growth (measured as growth in diameter) for bald cypress trees at Breaux Bridge. Before wastewater applications to the treatment site began in the late 1940s or early 1950s, growth in the control site was significantly higher than growth in the treatment site ($P>0.05$). After wastewater applications began, stem growth was significantly higher in the treatment site. Both sites were similar in size and structure (Hesse et al. 1997).

Additionally, estimated accretion rates in the treatment site fell within one standard deviation of the estimated rate of RSLR in the region (Rybczyk 1997).

Using an integrated field and modeling approach, Rybczyk et al. (1996), Rybczyk (1997) and Rybczyk et al. (1998) focused on the use of wastewater effluent at Pointe au Chene for wetland enhancement and restoration. Their studies revealed that neither aboveground tree production nor annual rates of decomposition were affected by wastewater effluent. Because of increased floating aquatic vegetation production, however, rates of sediment accretion increased significantly after wastewater applications began and fell within one standard error of the estimated rate of relative sea level rise. A site-specific wetland elevation model revealed that wetland elevation in this subsiding region was more sensitive to the uncertainty surrounding estimates of eustatic sea level rise and deep subsidence than to possible effluent-related changes in autogenic processes such as decomposition and primary production (Rybczyk 1997). The model also indicated that nutrient addition alone was not sufficient to lead to long term restoration of the forested wetland and that some mineral sediment input was necessary.

Increased Productivity

Secondarily treated effluent delivers nutrient-rich water to wetlands, stimulating vegetative productivity. Long term impacts on forested wetlands can be assessed by evaluating data from the Breaux Bridge treatment wetland. The treatment wetland at Breaux Bridge is unique because of its long history of discharge to the receiving wetland. The town of 6,000 has been discharging its effluent from an oxidation pond (3,785 $m^3 d^{-1}$) to a 1475-ha cypress-tupelo wetland for almost 50 years (Breaux and Day 1994). Monitoring of the effluent impact site and an adjacent reference site began in 1992. A comprehensive site description is provided by Day et al. (1993).

A dendroecological analysis was conducted (Hesse 1994; Hesse et al. 1997) to determine the long term impacts of wastewater effluent on aboveground productivity (Fig. 6). Stem wood growth from 1920 to 1992 was measured at the treatment site and an adjacent control site. An annual diameter increment ratio was calculated by comparing stem wood growth from the treatment site versus the stem wood growth at the control site. Records indicate that the city began discharging into the forested wetland between 1948 and 1953. Before wastewater application began, Hesse et al. (1997) found statistically significant higher growth in the control site than at the treatment site. However, after onset of treatment, there was increased growth in the treatment site, resulting in statistically significant higher annual diameter increment ratios (Hesse et al. 1997). A spike in the annual diameter increment ratios coincides with the onset of treatment. The sustained elevated trend of ratios in the treatment site illustrates the long term benefits of wetland treatment in this site.

Short term records at this site also confirm these findings. In January 1994, the effluent discharge was switched from the historic wetland (old treatment site), to a new site that had not previously received effluent (new treatment site). In 1992, permanent plots were established at both sites to measure annual litterfall and stem growth (Table 2). There was no statistically significant difference in the total aboveground production between the old treatment site and the new treatment site during 1993 (Delgado-Sanchez 1995). However, during 1994 and 1995, when effluent discharge was switched to the new treatment site, total production was significantly higher at the new treatment site compared to the old treatment site (Delgado-Sanchez 1995). Most of this difference was due to increases in stem wood biomass in the new treatment site and not leaf production.

Similar results have been reported for the other treatment wetlands. For example, a study conducted at Amelia also indicates an increase of primary productivity. The City of Amelia is investigating the ecological feasibility of incorporating the Ramos forested wetland as part of its treatment system to polish secondarily treated sewage effluent (Day et al. 1997a). A year long study on primary productivity indicates enhanced growth in the treatment sites (Table 3). Productivity, as expressed in mean

Table 2. Aboveground production (g $m^{-2}yr^{-1}$ ± se) measured at Breaux Bridge treatment site.[1]

Year	Site	Stem Wood Production	Leaf Production	TotalAboveground Production
1993	Old Treatment*	780 ± 358.5	420	1200.9
1993	New Treatment	677.9 ± 69.21	514	1191.9
1994	Old Treatment	593.2 ± 46.8	547.3 ± 9.2	1140.5
1994	New Treatment*	1383.4 ± 186.4	745.8 ± 8.2	2129.2•
1995	Old Treatment	574.8 ± 187.4	705.2 ± 81.1	1280
1995	New Treatment*	847.7 ± 200.1	763.6 ± 45.5	1611.3•

[1]From Delgado-Sanchez 1995.
*Indicates site receiving wastewater effluent.
•Indicates statistical difference.

litterfall for one year, was statistically significantly higher in the treatment site than in one of the control sites (Day et al. 1997a).

Economic Savings

Conventional wastewater treatment is often very expensive for the loads generated from many of the small communities in southern Louisiana. Wetland assimilation can provide an affordable and effective waste treatment option. In a series of papers, Breaux (1992), Breaux and Day (1994) and Breaux et al. (1995), conducted economic cost benefit analyses of the wastewater treatment operation at Breaux Bridge and Thibodaux (Table 4). They conservatively estimated a capitalized cost savings, using natural wetland wastewater treatment rather than conventional tertiary treatment. At Breaux Bridge, the estimated costs savings was approximately $1.4 million, over a 30-year period. At Thibodaux, there is a potential savings of approximately $500,000. However, it is further

Table 3. Total mean litterfall (g m^{-2}) collect at the Amelia treatment wetland from Sept. 1995-Sept. 1996. Those means with different letter are statistically different (Day et al. 1997a).

Site	Mean Litterfall (g m^{-2}) ± s.e.
Control 1	581.09 ± 35.68
Control 2	42.45 ± 38.24
Treatment	716.65 ± 38.08
Lake1 Site 1	546.06 ab ± 47.24
Lake1 Site 2	666.35 ± 49.52

[1]The treatment site is adjacent to Lake Palourde. Lake Site 2 is connected directly to the treatment site by a small channel. Thus measurements were taken at the lake edge where there is potential influence from the effluent.

Table 4. Cost comparisons for three wetlands treatment projects

Site	Conventional Treatment	Wetland Treatment	Cost Savings
Breaux Bridge[1]	1,500,000	125,000	1,375,000
Thibodaux[1]	1,650,000	1,150,000	500,000
Dulac[2]	2,200,000	700,000	1,500,000

[1]Costs reported in 1992 dollars as per Breaux and Day (1994) and Breaux et al. (1995). Capitalized costs are discounted at 9% for 30 years.

[2]Costs reported in 1995 dollars as per Cardoch et al. (2000). Capitalized costs are discounted at 8% for 25 years.

noted that capitalized savings could be as high as $1,300,000 over a 30-year period, depending upon the disinfection system employed prior to wetland discharge.

Non-toxic, industrial processors, such as shrimp processors, can benefit from using wetlands for their highly seasonal loads. A study was recently conducted to determine the feasibility of using wetlands for treatment of shrimp processing wastewater in Dulac, Louisiana (Cardoch et al. 2000). The avoided cost estimate approach was used to compare costs from conventional on-site treatment of the shrimp processing effluent with costs for wetlands treatment. Conventional treatment would cost approximately $200,000 per year for 25 years, as compared to wetland treatment costs of $64,000 per year for 25 years. This is a potential cost savings of $1.5 million dollars over 25 years (Table 4).

Much wetland treatment has focused on constructed wetlands primarily to provide a high degree of control for treatment. In Louisiana, the dense network of canals and levees have left many wetlands hydrologically isolated and confer the same degree of control as constructed wetlands. With many natural systems plentiful, it is unnecessary to build artificial wetlands in Louisiana, although many have been built. These isolated wetlands provide a practical economic solution for the small communities that are widely dispersed in the coastal zone.

Conclusions

Results from several ongoing and completed studies of wastewater treatement in wetlands indicate that they are achieving the ecological goals of enhancing effluent water quality, stimulating vertical accretion, and increasing productivity. Economically, the savings are substantial for small communities and non-toxic industrial processors. Calculations of nutrient retention and loss via denitrification, plant uptake, and burial indicate that the receiving wetlands should assimilate all of the NO_3-N and more than 50% of the phosphorus given the current loading rates. As water quality regulations become more stringent, and federal grants for sewage treatement improvements become less available, it will be increasingly difficult for small coastal communities to meet the water quality standards. Wetland wastewater treatment could provide an economically viable and effective alternative to expensive conventional tertiary treatment. Additionally, it potentially serves as a means for wetland restoration in the subsiding coastal zone.

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Migration of Speciated Metals in Soils Reclaimed from Ship Channel Sediments

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ABSTRACT: The heavy metals chromium, copper, iron, manganese, nickel, lead and zinc were measured in Calcasieu River sediments, reclaimed soils, soils in a site being reclaimed, and soils in a control site at Sabine National Wildlife Refuge. The metals were selected due to their presence in soil, possible presence in oil from spills, use as micronutrients for plants, or contaminants in biota. The possibility of contaminated soils being used to reclaim subsiding marshland and the effect of these contaminants on existing biota were the motivation for this study. Concentrations of heavy metals were determined using a sequential extraction procedure that partitioned the metals into the fractions a) exchangeable, b) bound to carbonates, c) bound to manganese oxides, d) bound to organic matter, and e) residual metals in a microwave digestion procedure.

Introduction

Trace metals in soils and sediments should be considered as points of concern when habitat restoration projects are undertaken because of their importance as micronutrients and their potential to become toxic pollutants (Davies 1992). The trace metals Cr, Cu, Fe, Mn, Ni, Pb, and Zn were selected for this study because of their importance as micronutrients for plants and/or animals (Davies 1992) and possible presence in oil. It has been shown, through various biogeochemical studies, that plant roots affect the species in which metals exist near the rhizosphere by altering the pH of this environment (Jenny and Grossenbacher 1963; Hauter and Mengel 1988; Gijsman 1990). The form in which a metal exists is believed to greatly affect its bioavailability, and thus its effects on the environment, both positive and negative. The use of sequential extraction procedures for the speciation of particulate metals provides a means for monitoring distribution of particulate metals in different chemical forms (Tessier et al. 1979).

There have been numerous studies in the Calcasieu Lake estuary on metal contamination in water, sediment and biota. Previous studies have measured concentrations of Cr, Cu, Fe, Mn, Ni, Pb, and Zn in Sabine National Wildlife Refuge sediments (Army Corps of Engineers 1987; Beck et al. 1990; Cunningham et al. 1990; Derouen and Stevenson 1987; Schultz 1991; Wade 1994; J. Sneddon unpublished data). Sediments used to construct the Sabine National Wildlife Refuge spoil disposal/wetland creation site varied considerably in physical characteristics along the ship channel dredging area. Differences in physical characteristics of sediments are of concern because of their effect on the chemical processes occurring which in turn alter speciation of metals (Wade 1994).

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Sample Sites

The Calcasieu River/Lake complex contains the sites where the samples were collected. Sediments thought to be contaminated with particulate metals were used to create a marsh habitat at the Sabine National Wildlife Refuge in southwest Louisiana. The habitat reclamation sites, control site, sampling locations, and dredging locations are shown in Figure 1. A brief history of the restoration sites using Calcasieu Ship Channel dredge spoil at the Sabine National Wildlife Refuge is as follows: Site I was created in 1981 and is the oldest of the sites and had a homogeneous environment at the time of sampling. Site II was created in 1993 and was three years old when sampling began. It had a more heterogeneous environment than Site I in terms of topography at the time of sampling. The creation of Site III was completed in early 1996. Site IV was the natural "reference marsh" and was similar to Site I in environment. Site II had areas similar to Site IV which could be used for comparison. Site V was the Calcasieu Ship Channel from which the sediment was taken to create the marsh for Sites I, II, and III.

Materials and Methods

Sediment samples were collected at the sampling locations discussed previously using a teflon coated scoop. The upper 1 cm of topsoil was collected from a 10 cm diameter circle and placed in a plastic container. Three replicate samples were randomly taken at each sample site. At Sites I, III, and IV samples were taken from the interior (IN) above normal water levels and near the waters edge (OUT) to monitor these different environments. Site III was the proposed fill site and samples were taken prior to (PRIOR) and after (AFTER) the site was restored. Site II was sparsely vegetated and samples were collected near clumps of vegetation (NONVEG) and within the vegetated areas (VEG) to monitor the impact of plant activity on metal concentrations. The ship channel sediments (Site V) were taken using a 0.1 m^2 Ekman grab. All sediment samples were stored at 4°C for less than 3 d, weighed, and dried in a forced air oven at <95°C. Upon drying, dry masses were measured so that percent water could be determined. The samples were then sealed in plastic containers and stored in a cabinet at room temperature. All samples were digested within 3 mo of collection.

A procedure closely following that of Tessier et al. (1979) was employed for the sequential extraction procedure. After drying, a fraction of the sediment samples was crushed into a fine powder by using a mortar and pestle and screened through a 20 mesh sieve. Specimens of powdered sediments (2.5 g) were placed into 250-ml Erlenmeyer flasks and sealed using parafilm. The flasks were labeled and the samples were ready for immediate digestion.

(1) *Exchangeable.* The exchangeable metals were extracted from the sediment sample by digesting the sample with 20 ml of a 1.0 M $MgCl_2$ solution adjusted to a pH of 7.0 at room temperature. The mixture was continuously agitated for 1 h using a magnetic stirrer. This solution was then centrifuged in a 50-ml polypropylene centrifuge tube at 7000 rpm for 30 min. The supernatant was poured into a 50-ml volumetric flask and brought to volume using deionized water. This solution was placed in a 40-ml plastic bottle prior to analysis. The residue was washed with approximately 20 ml of deionized water and the solution centrifuged at 7000 rpm for 15 min. The supernatant was discarded.

(2) *Metals Bound to Carbonates.* Metals bound to carbonates were extracted from the residue of the first extraction with 20 ml of a 1.0 M $NaC_2H_3O_2$ solution, adjusted to a pH of 5.0 with acetic acid. Following continuous agitation at room temperature for 4 h using a magnetic stirrer, the mixture was centrifuged in a 50-ml polypropylene centrifuge tube at 7000 rpm for 30 min. The supernatant was poured into a 50-ml volumetric flask and brought to volume using deionized water. This solution was stored in a 40-ml plastic bottle prior to analysis. The residue was washed with approximately 20 ml of deionized water and the solids separated by centrifugation at 7000 rpm for 15 min. The supernatant was discarded.

(3) *Metals Bound to Iron and Manganese Oxides.* Metals bound to iron and manganese oxides were extracted from the residue of the second

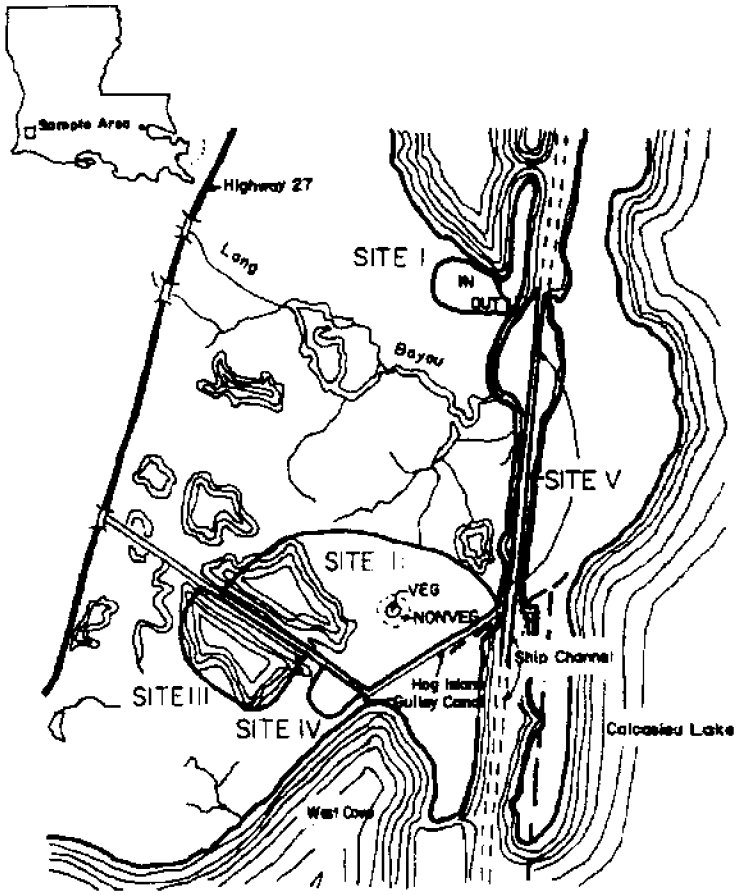

Fig. 1. A map of the sampling area including the locations of the five sampling sites and the types of samples collected at the sites.

extraction with 50 ml of a 0.04 M $NH_2OH{\cdot}HCl$/25% $HC_2H_3O_2$ solution. The solution was kept at a temperature of 96°C ± 3°C for a 5.5-h period and agitated continuously for 30 min. using a magnetic stirrer. The mixture was then centrifuged in a 50-ml polypropylene centrifuge tube at 7000 rpm for 30 min. The supernatant was poured into a 50-ml volumetric flask and brought to volume using deionized water. This solution was stored in a 40-ml plastic bottle prior to analysis. The residue was washed with approximately 20 ml of deionized water and the solids separated by centrifugation at 7000 rpm for 15 min. The supernatant was discarded.

(4) *Metals Bound to Organic Matter.* Metals bound to organic matter were extracted from the residue from the third fraction with 7.5 ml of a 0.02 M HNO_3 solution and 12.5 ml of a 30% H_2O_2 solution, adjusted to a pH of 2.0. Following continuous agitation for 2 h using a magnetic stirrer at a temperature at 85°C ± 2°C, an additional 7.5 ml of 30% H_2O_2 solution was added. The pH of the mixture was adjusted to a pH of 2.0 and the agitation continued at a temperature of 85°C ± 2°C for 3 h. After allowing the mixture to cool to room temperature (approximately 45 min.), 12.5 ml of a 3.2 M $NH_4C_2H_3O_2$/20% HNO_3 solution was added. This mixture was diluted to approximately 50 ml using deionized water, agitated continuously for 30 min and the solids extracted by centrifugation in a clinical centrifuge at 7000 rpm for 30 min. The supernatant was poured into a 50-ml volumetric flask and brought to volume using deionized water.

This solution was then stored in a 40-ml plastic bottle prior to analysis. The residue was washed with approximately 20 ml of deionized water and the solids separated by centrifugation at 7000 rpm for 15 min. The supernatant was discarded.

(5) *Residual.* Metals in the residual form were extracted from the residue of the fourth extraction by placing the residue in a microwave digestion bomb and adding 10 ml of concentrated 70% HNO_3. The bomb was then placed into a microwave oven for 30 s at full power. The bomb was then taken out and allowed to cool to room temperature for 1 h. The residue in the bomb was placed in a 50-ml polypropylene centrifuge tube, diluted to approximately 20 ml using deionized water and centrifuged for 30 min. The supernatant was poured into a 50-ml volumetric flask and brought up to volume using deionized water. This solution was stored in a 40-ml plastic bottle prior to analysis. The residue was then discarded.

Metal concentrations were determined using Flame Atomic Absorption Spectrophotometry (Cr, Cu, Fe, Mn, Ni, and Zn) and Graphite Furnace Atomic Absorption (Pb). Duplicate samples, matrix and reagent blanks, and standard additions were used to check for reproducibility, contamination and possible matrix effects. Contamination was not found in any blank samples, and the sample matrix did not affect the analysis.

Statistical Groupings of Sampling Stations

Sampling stations were placed in three groups based upon observed differences in metal concentrations. The station groups were: Group 1 - Site I (OUT), Site III (PRIOR), and reference marsh (Site IV); Group 2 - Site I (IN), Site II (NONVEG), Site III (AFTER), and Site V; Group 3 - Site II (VEG). To test whether the selected groupings could be separated and distinguished by metal concentration, a multivariate analysis of variance (MANOVA) was used. Group differences with respect to metal concentrations were highly significant ($p < 0.05$).

Since no information is obtained to determine the individual metals causing the differences within groups, a univariate analysis of variance (ANOVA) was used to examine the effect of each metal. The concentrations of Fe, Mn, and Zn were found to vary significantly between the station groups. The remaining metals showed no group differences. To further understand the relationship of the metals in each group, a discriminant analysis was run using Duncan's multiple range test to confirm that group means were different. The results (Table 1) indicate that an individual observation could be correctly placed in the predicted grouping 91% of the time.

Table 1. Discriminant analysis of metal data classifying metal concentrations in each station grouping.

		Predicted Group Membership[a]		
Actual Group	No. of Cases	1	2	3
1	45	38 85.0%	5 10.5%	2 4.5%
2	105	2 1.8%	95 90.6%	8 7.6%
3	133	0 0.0%	10 7.8%	123 92.2%

[a] Percent of group cases correctly classified were 91.22%.

Results

The samples collected were selected based upon their location and the different characteristics of the sample sites (i.e., vegetated sediment, non-vegetated sediment, ship channel sediment, etc.). The mean concentrations for Cr, Cu, Fe, Mn, Ni, Pb, and Zn are given in Table 2 for each sampling station. The values reported are means for sums of the five separated fractions for each station. Figure 2 shows the variation of mean metal concentrations at each sample site. Elevated concentrations of Mn were found at Sites I, II, and V compared to the reference site (Site IV). A similar pattern was found for Fe concentrations with the inclusion of elevated levels at Site III, while Zn concentrations were found to be elevated at Sites I and II. Figure 3 shows the effect on the Fe and Mn concentrations after ship channel spoil has been placed on Site III. It is clear that vertical mixing has not occurred.

The results obtained using the sequential extraction procedure and summing the fractions yield results that agree well with earlier studies on Sabine National Wildlife Refuge sediments (Army Corps of Engineers 1987; Beck et al. 1990; Cunningham et al. 1990; Derouen and Stevenson 1987; Schultz 1991; Wade 1994; J. Sneddon unpublished data). The mean metal concentrations measured in this study are compared in Table 2. Concentrations of Cu, Cr, Mn, and Zn for ship channel sediments found in these studies ranged from 10-59 ppm for Cr, 7-27 ppm for Cu, 0.8-2% for Fe, 590-670 for Mn, 6-18 ppm for Ni, 8-22 for Pb and 36-84 ppm for Zn. The means for all totals performed in the current study compare favorably and fall within the ranges measured previously (Table 3).

To further test the reliability of the method used, a subsample was digested to yield total metal concentrations in the sample (Table 4). The agreement between the measurements is within the variation between samples. Total metal concentrations, however, are generally higher, perhaps reflecting observed losses of sample during transfer following centrifugation or other systematic errors.

Table 2. Mean metal concentrations and standard deviations by sampling site in ppm unless noted. Numbers in parentheses are the number of samples used in calculating means.

Site	Chromium	Copper	Iron (%)	Manganese	Nickel	Lead	Zinc
I	18.5 (6) ±2.0	11.0 (6) ±3.1	1.82 (6) ±.61	210 (6) ±90	20.4 (6) ±2.5	6.4 (5) ±.7	47.1 (6) ±5.4
II	21.8 (22) ±2.9	14.5 (22) ±6.2	1.95 (11) ±.55	640 (22) ±120	19.5 (11) ±3.0	8.3 (9) ±.9	55.5 (22) ±11.5
III (before)	19.5 (6) ±2.5	10.0 (6) ±3.5	0.41 (6) ±.22	151 (6) ± 30	10.4 (6) ±2.8	6.8 (4) ±.8	36.5 (6) ±4.1
III (after)	18.1 (6) ±3.8	13.6 (6) ±1.8	1.91 (3) ±.35	648 (6) ±145	12.1 (3) ±3.2	5.6 (2) ±.4	41.0 (6) ±8.1
IV	23.6 (14) ±1.8	10.6 (4) ±2.7	0.35 (4) ±.11	56.2 (3) ±9.1	17.8 (4) ±2.5	4.3 (3) ±.6	33.0 (4) ±10.1
V	21.4 (14) ±4.5	9.9 (14) ±3.2	2.10 (12) ±.61	580 (14) ±220	14.6 (12) ±3.5	5.6 (9) ±.8	41.6 (14) ±15.6

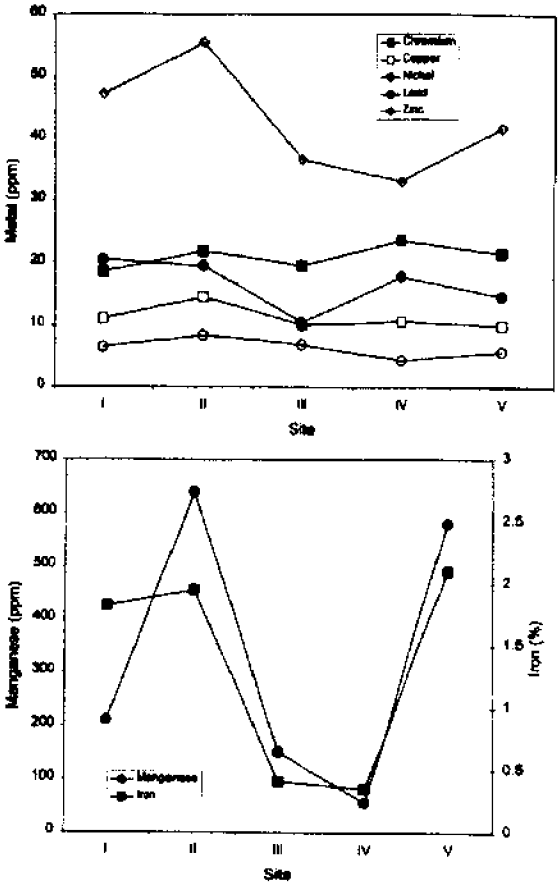

Fig. 2. The variation of mean metal concentrations across each sampling site.

Table 5 gives the mean concentrations (ppm) for the studied metals for each fraction separated and for each sample site. For example, copper at Site 1 (OUT) varies from 0.3 ppm in fraction 1 (ion exchange) to a high value of 4.7 ppm in fraction 5 (residual). Table 6 gives the same results for the metals Fe, Ni, and Pb.

Concentrations of Cr, Cu, Mn and Zn are compared for Site V, Site II (VEG), and Site II (NONVEG) samples in Table 5. For Site II (VEG) samples, greater than 85% of Cu is bound in the organic and residual fractions, while 50% of Cr is contained in the residual fraction, with the remainder spread evenly throughout the other fractions. Site V and Site II (NONVEG) samples have approximately 50% of the Mn bound in the carbonate and manganese oxide fractions, whereas Site II (VEG) samples have over 75% in the

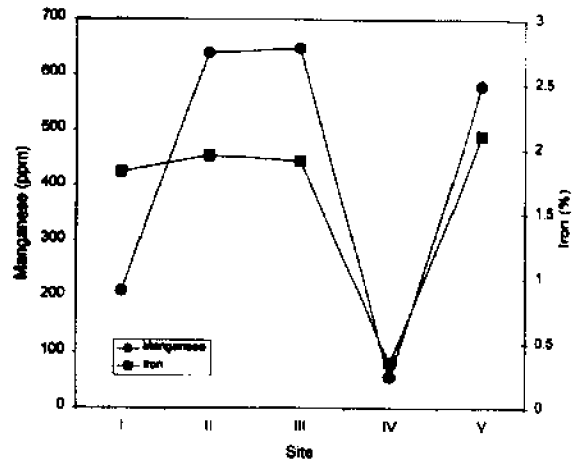

Fig. 3. The variation of mean concentrations of Fe (%) and Mn (ppm) across each sampling site after Site III was restored using dredge spoil.

manganese oxide and organic phases and a small increase in the residual phase. This change in pattern is unique for Site II and may suggest uptake of Mn by the local vegetation and subsequent buildup in detritus.

Discussion

Concentrations of Cu, Cr, Pb, and Ni are not significantly different among Site II (VEG), Site II (NONVEG), and Site V sediments. This trend should be expected in typic soils for the sampling area. Copper tends to be immobile in soils with a pH lower than 6 and chromium usually exists as the chromic (+3) ion in soils with a low pH, allowing it to complex with organic materials and adsorb to clays and other minerals, which renders Cr immobile. The pH of the soils at the sampling sites averaged 5.3 (C. E. Proffitt unpublished data) for the surficial sediments and could cause the observed trend (Gambrell 1994).

For Site V sediments, a definite correlation for manganese can be seen between samples collected before dredging (BD) and those collected after dredging (AD). The highest concentrations of Mn are contained in fraction 2 for BD samples and in fraction 4 for AD samples. The most probable cause for this variation occurring after dredging would appear to result from differences in grain size and composition of the sediments (Wade 1994).

Table 3. Comparison of metal concentrations from previous studies on Calcasieu Ship Channel sediments taken in the vicinity of Sabine National Wildlife Refuge.

Investigator	Cr (ppm)	Cu (ppm)	Fe (%)	Mn (ppm)	Ni (ppm)	Pb (ppm)	Zn (ppm)
DeRouen (1986)	19	27	—	—	—	10	45
ICF Kaiser (1989)	25	7	—	—	6	8	36
Cunningham (1990)	28	17	1.97	589	18	8	36
Schultz (1991)	34	15	1.71	670	15	9	84
Army Corp. (1993-94)	10	11	—	—	14	12	36
Wade (1994)	59	8	0.77	—	—	20	61
Sneddon (1995)	17	18	—	—	—	22	70
Average all Investigators	27	15	1.48	630	13	13	53
SITE V (This Work)	21	10	2.10	580	15	6	42

The total Mn concentrations for Site I indicate a difference in the INNER (332 ppm) and OUTER (92 ppm) sampling sites. Site I (OUT) samples have significantly lower Mn concentrations than Site I (IN). Complete coverage of Site I by dredge spoils from the Ship Channel occurred during the same dredging periods, indicating that the variations of Mn should not be as large as observed. One possible explanation for this discrepancy may be the location of the sampling sites. Site I (OUT) is located just west of the Ship Channel, very near Long Bayou and adjacent to several minor streams (Figure 1). Site I (IN) is located in the center of Site I far from open water or any waterways. The soils at Site I are typic regional soils and have a very acidic surface layer (US Dept. Agriculture 1995). This may allow acid leaching to occur at Site I (OUT), causing depletion of Mn. Leaching appears to have occurred at Site I (IN) since the mean Mn concentration has decreased from typical spoil concentrations of about 900 ppm to the current mean concentration of about 300 ppm. Another explanation may be that Site I (OUT) was never covered by spoils, thus measured Mn may reflect concentrations in typic soils represented by Site IV, the reference marsh.

Site IV (reference site) contained two sampling locations, PR-L (above ambient water levels) and PR-W (covered by water). Pronounced differences of Mn concentrations exist for the first two fractions (Table 5). PR-L samples have higher Mn concentrations in the exchangeable fraction, with a marked decrease in Mn for carbonate fractions. For PR-W, the trend is reversed; fraction 1 contains low concentrations of Mn with fraction 2 containing higher concentrations. Manganese exists in soils primarily in the divalent state which is mobile and predominates in acidic soils; but when the pH rises above 6, Mn^{+2} combines with carbonates. Thus, the greatest concentration of Mn should occur in fraction 1 for PR-L and fraction 2 for PR-W because flooding of marsh sediments should buffer pH to 7 (Gambrell 1994).

Site III (AFTER) samples were collected six months after the site was covered with Ship Channel dredge spoils. Table 3 indicates that Mn resides mainly in fraction 2 at Site III (AFTER) compared to Site V which has high concentrations in fractions 2-4. Site III (PRIOR) shows high concentrations of Mn in fractions 1-4. The pronounced shift of Mn from fraction 1 samples collected at Site III (PRIOR) and fraction 4 of the Site V samples into fraction 2 of Site III (AFTER) can be explained using the same reasoning as that for Site IV-L and IV-W. A fraction of Mn contained in fraction 1 of Site III (PRIOR) sediments should be converted into fraction 2 when Ship Channel dredge spoil are placed on Site III. Water covered the entire site as sediments settled and dewatered resulting in a shift

Table 4. The concentrations (ppm) of metals (except Fe in percent) in subsamples compared to the concentration in the summed fractions (in parenthesis) measured in the sequential extractions.

SAMPLE SITE	REPLICATE NUMBER	Cr (ppm)	Cu (ppm)	Mn (ppm)	Fe (%)	Ni (ppm)	Pb (ppm)	Zn (ppm)
SITE I								
	IN-B	20.1 (21.0)	12.5 (10.6)	162 (155)	19200 (18600)	19.1 (23.2)	6.7 (6.4)	60.6 (54.5)
	OUT-A	19.5 (17.4)	11.1 (10.8)	104 (94.4)	12100 (12500)	17.4 (17.3)	7.2 (6.8)	31.9 (32.6)
SITE II								
	NONVEG-B	24.1 (20.4)	24.9 (23.0)	455 (424)	14600 (16900)	20.7 (19.8)	6.8 (7.1)	55.4 (48.4)
	VEG-C	30.0 (28.2)	20.7 (18.4)	709 (688)	22000 (20300)	21.1 (19.7)	6.4 (6.7)	61.9 (54.4)
SITE III								
	PRIOR-A	17.4 (18.2)	8.5 (8.4)	128 (122)	16100 (14000)	16.2 (17.2)	6.4 (6.2)	35.3 (34.2)
	AFTER-A	16.7 (20.8)	16.6 (15.2)	591 (589)	9100 (8700)	14.9 (11.8)	7.2 (5.5)	39.4 (36.0)
SITE IV								
	IN-A	22.4 (22.0)	9.0 (5.0)	79.4 (60.8)	2600 (2400)	10.8 (8.6)	6.7 (6.6)	33.0 (29.2)
	OUT-C	30.1 (22.4)	14.9 (12.6)	50.8 (43.6)	4100 (3500)	14.1 (12.4)	5.9 (4.3)	42.7 (37.0)
SITE V								
	BD-A	31.1 (28.4)	8.0 (7.6)	978 (998)	9000 (8700)	12.9 (11.8)	5.4 (5.5)	36.9 (30.6)
	AD-A	24.7 (19.6)	5.6 (4.8)	461 (453)	7100 (6800)	13.1 (12.8)	5.8 (5.8)	35.4 (38.2)

of Mn from fraction 1 in the Site III (PRIOR) sediments to fraction 2. As stated earlier, Mn^{+2} can combine with organic matter when the pH is above 6, inferring that some of the Mn in fraction 4 (organic) in Ship Channel dredge spoils will be shifted into fraction 2 when encountering the increased pH in water into which it was placed. Shifting of Mn in the Site III (PRIOR) and Site V samples from the exchangeable and organic fractions should produce the measured initial increase in Mn in fraction 2 for Site III (AFTER) samples. Theoretically, some Mn in fraction 2 will be shifted into fraction 1 as water levels fall and pH is lowered by the acidic soils.

The data in Table 5 illustrate a shift in Mn from fraction 2 of Site II (NONVEG) samples to fractions 3 and 4 of vegetated samples at the same site. A plausible explanation for the shift of Mn into fraction 3 (iron/manganese oxides) is that oxidation of Mn^{+2} increases in aerated soils having initially low pH that are periodically flooded with alkaline, saline water (Gambrell 1994). When vegetation begins to grow on the soils, the root systems allow better aeration and, thus, greater access to O_2 by soil Mn. Plants also can alter the nearby soil chemistry ("rhizosphere effect") by changing the pH and/or redox potential by exuding protons or chelating agents near the roots (Otte et al. 1995; Simmers et al. 1981). This allows for higher solubility of deficient metals resulting in chelation and complexation of toxic metals into a chemical form that can be absorbed by the roots (Otte et al. 1995; Simmers et al. 1981). Changes in pH and Eh in the rhizosphere by the roots of *Spartina anglica* play important roles in determining the chemical speciation of elements and their resulting mobilities (Otte et al. 1995). Eh values were found to be 400 mV higher around the roots of *Aster tripolium* than for *S. anglica* (Otte et al. 1993). They concluded that this effect results from the greater flooding in the areas occupied by *S. anglica* due mainly to differences in elevation. This conclusion was supported by high salinities found in soils around *S. anglica*.

The bioavailability of metals is also affected by the translocation of the metals from the roots to the stems and leaves (Otte et al. 1993). This may not seem to be within the realm of effects caused by chemistry occurring in soil. However, certain metals cannot be transported from the roots to the plant tops resulting in accumulation near plant roots unless factors in soil, such as alkalinity, phosphate levels, or base cation concentrations, allow for a change in metal speciation. Otte et al. (1993) have reported that Cu and Zn are higher in the roots of *S. anglica* than in the shoots; however, the opposite was true for *A. tripolium*. Differences in soil chemistry due to frequent flooding appears to have an important role in this difference.

A final factor leading to bioavailability is the speciation of metals. This factor is related to some of the mechanisms which aid or hinder translocation. Plants can only uptake metals in specific ionic forms or certain chemical compounds or complexes. Metal species that are very insoluble and are adsorbed onto surrounding minerals greatly hinder bioavailability. Soluble metal species that do not adsorb on surrounding minerals tend to be transported more quickly through the soil in forms that are more readily adsorbed by plant roots and are thus more readily available to the food web.

As manganese oxides are produced they are precipitated and accelerate the oxidation of Mn. Mn^{+2} is selectively adsorbed by manganese oxides and as manganese oxides are formed surface area is increased, thereby increasing the rate of adsorption of Mn^{+2}. The shifting of Mn into fraction 4 of the vegetative samples is probably due to an increased supply of organic rich surface matter resulting from decaying vegetation. When the pH is above 6, Mn^{+2} ions form complexes with organic matter causing an increase in Mn concentrations in fraction 4 for the vegetative samples.

Summary

The use of sequential extraction procedures to study metal distributions in sediments can yield useful information related to bioavailability to organisms. The reproducibility of the method correlated well with previous measurements of metal concentrations in the study area and with results for digestions of collected subsamples suggesting that this method can provide accurate and useful results. No evidence was found to suggest that Calcasieu Ship Channel spoil would contaminate restored marsh with heavy metals at the Sabine National Wildlife Refuge. However, in areas that are heavily vegetated the concentration profiles for Fe, Mn, and Zn become modified when compared to ship channel spoil and in nonvegetated areas surrounding vegetation. The concentration changes appear to be related to changes in pH and/or Eh caused by the alteration of soil chemistry by vegetation. The roots of *Spartina* species have been found (Simmers et al. 1981; Otte et al. 1995) to change the pH and/or Eh in the rhizosphere by the release of hydrogen ions or chelating agents. This may result in changing the chemical speciation of

Table 5. Mean concentrations (ppm) ± SD of Cu, Cr, Mn, and Zn in each fraction at each sample site.

SITE I OUT PPM BY FRACTION				
FRACTION	ANALYTE			
	Cu	Cr	Mn	Zn
1	0.3 +/- 0.1	2.0 +/- 0.8	16.9 +/- 3.4	4.5 +/- 2.5
2	0.9 +/- 0.7	2.5 +/- 0.3	20.0 +/- 8.7	6.3 +/- 3.4
3	0.2 +/- 0.0	2.7 +/- 0.6	13.3 +/- 5.8	5.1 +/- 0.9
4	4.5 +/- 2.9	3.2 +/- 0.9	26.7 +/- 7.6	9.7 +/- 2.3
5	4.7 +/- 3.4	7.1 +/- 0.6	15.1 +/- 5.2	10.6 +/- 4.5

SITE I IN PPM BY FRACTION				
FRACTION	ANALYTE			
	Cu	Cr	Mn	Zn
1	0.2 +/- 0.0	2.5 +/- 0.6	59.0 +/- 29.7	5.9 +/- 3.1
2	0.3 +/- 0.1	2.4 +/- 0.7	146.7 +/- 206.4	7.3 +/- 2.3
3	0.2 +/- 0.0	2.9 +/- 0.9	63.3 +/- 49.3	10.9 +/- 2.0
4	4.8 +/- 1.4	4.1 +/- 0.7	41.7 +/- 17.6	13.1 +/- 1.9
5	6.0 +/- 0.8	7.9 +/- 0.2	20.9 +/- 3.7	16.5 +/- 1.0

PR-L PPM BY FRACTION				
FRACTION	ANALYTE			
	Cu	Cr	Mn	Zn
1	0.4 +/- 0.3	2.5 +/- 0.1	51.9 +/- 31.4	6.1 +/- 0.5
2	0.2 +/- 0.0	2.0 +/- 0.0	16.7 +/- 7.6	4.5 +/- 0.5
3	0.2 +/- 0.0	2.3 +/- 0.1	21.7 +/- 12.6	4.5 +/- 0.6
4	1.7 +/- 0.8	2.9 +/- 0.4	31.7 +/- 7.6	6.0 +/- 0.2
5	8.3 +/- 5.0	10.4 +/- 1.7	13.7 +/- 1.4	13.5 +/- 4.0

PR-W PPM BY FRACTION				
FRACTION	ANALYTE			
	Cu	Cr	Mn	Zn
1	0.5 +/- 0.1	2.4 +/- 0.5	24.6 +/- 6.6	2.4 +/- 0.2
2	0.6 +/- 0.4	2.3 +/- 0.6	55.0 +/- 17.3	5.7 +/- 0.6
3	0.2 +/- 0.0	2.5 +/- 0.1	30.0 +/- 8.7	8.9 +/- 1.2
4	2.9 +/- 1.6	3.5 +/- 0.4	33.3 +/- 2.9	8.5 +/- 2.0
5	6.1 +/- 1.4	8.3 +/- 1.7	17.6 +/- 2.3	13.7 +/- 1.5

NEW SITE PPM BY FRACTION				
FRACTION	ANALYTE			
	Cu	Cr	Mn	Zn
1	0.9 +/- 0.9	1.3 +/- 1.0	14.5 +/- 3.4	1.5 +/- 0.3
2	1.2 +/- 0.7	2.7 +/- 1.4	386.7 +/- 28.4	6.1 +/- 0.6
3	0.5 +/- 0.6	2.5 +/- 1.3	180.0 +/- 75.7	10.1 +/- 1.2
4	6.1 +/- 1.0	4.8 +/- 0.2	60.0 +/- 10.0	11.0 +/- 4.2
5	4.9 +/- 0.9	6.8 +/- 2.3	7.3 +/- 0.3	12.3 +/- 7.1

REFERENCE SITE PPM BY FRACTION				
FRACTION	ANALYTE			
	Cu	Cr	Mn	Zn
1	0.4 +/- 0.3	2.9 +/- 0.6	13.1 +/- 3.1	5.4 +/- 1.6
2	0.2 +/- 0.0	2.6 +/- 0.4	10.0 +/- 4.1	3.2 +/- 1.0
3	0.4 +/- 0.3	3.0 +/- 0.6	13.8 +/- 4.8	5.9 +/- 2.9
4	3.5 +/- 2.0	5.8 +/- 0.2	17.5 +/- 6.4	9.9 +/- 7.2
5	6.1 +/- 1.9	9.3 +/- 1.0	1.8 +/- 0.5	8.6 +/- 4.9

SC PPM BY FRACTION				
FRACTION	ANALYTE			
	Cu	Cr	Mn	Zn
1	0.8 +/- 0.5	2.7 +/- 0.8	25.4 +/- 13.7	5.6 +/- 10.1
2	1.0 +/- 0.5	2.4 +/- 0.5	206.0 +/- 202.0	5.8 +/- 3.0
3	0.9 +/- 1.2	2.5 +/- 0.5	156.0 +/- 144.0	9.9 +/- 1.7
4	4.3 +/- 2.4	3.6 +/- 0.6	84.0 +/- 48.2	8.0 +/- 3.7
5	3.6 +/- 2.0	10.3 +/- 2.8	13.4 +/- 5.1	9.9 +/- 4.1

NON-VEGETATED PPM BY FRACTION				
FRACTION	ANALYTE			
	Cu	Cr	Mn	Zn
1	0.5 +/- 0.4	2.5 +/- 0.8	17.7 +/- 7.7	2.8 +/- 1.7
2	0.8 +/- 0.6	2.1 +/- 0.8	237.0 +/- 68.6	6.6 +/- 4.6
3	0.5 +/- 0.6	2.7 +/- 0.6	236.0 +/- 69.8	10.4 +/- 3.4
4	4.6 +/- 2.1	3.6 +/- 0.9	117.0 +/- 57.1	9.4 +/- 2.2
5	7.1 +/- 3.4	8.5 +/- 1.5	27.3 +/- 9.7	17.7 +/- 5.1

VEGETATED PPM BY FRACTION				
FRACTION	ANALYTE			
	Cu	Cr	Mn	Zn
1	0.4 +/- 0.2	3.0 +/- 1.3	8.1 +/- 7.7	2.3 +/- 1.0
2	1.0 +/- 0.7	2.4 +/- 1.4	97.7 +/- 66.2	6.1 +/- 2.1
3	0.5 +/- 0.4	3.1 +/- 1.2	350.0 +/- 89.2	20.8 +/- 6.4
4	4.1 +/- 2.3	3.6 +/- 1.2	163.0 +/- 29.8	11.5 +/- 11.4
5	9.7 +/- 9.4	10.2 +/- 2.4	29.1 +/- 9.1	23.9 +/- 11.3

Table 6. The mean concentrations (+/- SD) for the metals Fe, Ni, and Pb for each fraction separated and for each sample site.

SITE II VEGETATED METAL CONCETRATION (ppm) BY FRACTION			
	ANALYTE		
FRACTION	Fe	Ni	Pb
1	1.5 ± 0.7	0.6 ± 0.0	0.15 ± 0.07
2	37.5 ± 24.7	0.7 ± 0.2	0.65 ± 0.49
3	4370 ± 180	4.3 ± 1.0	1.80 ± 0.71
4	2460 ± 2040	5.1 ± 0.7	2.30 ± 0.28
5	12400 ± 500	7.0 ± 1.4	1.40 ± 0.57

SITE II NON-VEGETATED METAL CONCENTRATION (ppm) BY FRACTION			
	ANALYTE		
FRACTION	Fe	Ni	Pb
1	2.1 ± 1.1	<0.5	0.38 ± 0.23
2	33.9 ± 25.5	0.6 ± 0.2	0.72 ± 0.12
3	3270 ± 1270	4.9 ± 1.0	2.12 ± 0.61
4	4310 ± 890	5.7 ± 1.2	2.98 ± 0.48
5	11900 ± 780	8.8 ± 0.9	1.63 ± 0.21

REFERENCE SITE METAL CONCENTRATION (ppm) BY FRACTION			
	ANALYTE		
FRACTION	Fe	Ni	Pb
1	3.3 ± 1.5	<0.5	<0.1
2	6.7 ± 2.9	<0.5	0.37 ± 0.15
3	370 ± 160	0.7 ± 0.2	1.23 ± 0.42
4	2070 ± 470	2.7 ± 1.2	1.93 ± 0.92
5	710 ± 160	6.1 ± 1.2	1.20 ± 0.20

NEW SITE METAL CONCENTRATION (ppm) BY FRACTION			
	ANALYTE		
FRACTION	Fe	Ni	Pb
1	1.7 ± 1.2	<0.5	0.15 ± 0.07
2	92 ± 48	0.6 ± 0.2	0.30 ± 0.0
3	3510 ± 640	1.7 ± 0.5	1.40 ± 0.28
4	2840 ± 440	3.7 ± 0.7	2.70 ± 0.56
5	2230 ± 240	6.9 ± 2.0	1.15 ± 0.49

SHIP CHANNEL METAL CONCENTRATION (ppm) BY FRACTION			
	ANALYTE		
FRACTION	Fe	Ni	Pb
1	7.2 ± 11.6	<0.5	0.14 ± 0.10
2	20.4 ± 31.1	1.0 ± 0.7	0.31 ± 0.08
3	2680 ± 930	2.0 ± 0.8	1.65 ± 0.22
4	4330 ± 2100	3.8 ± 1.1	2.81 ± 0.65
5	6410 ± 3720	5.8 ± 1.3	1.58 ± 0.45

PROPOSED SITE METAL CONCENTATION (ppm) BY FRACTION			
	ANALYTE		
FRACTION	Fe	Ni	Pb
1	2.8 ± 1.3	<0.5	<0.10
2	30.8 ± 21.2	<0.05	0.35 ± 0.19
3	1880 ± 60	6.1 ± 1.5	1.88 ± 0.15
4	3740 ± 1020	2.6 ± 0.6	2.72 ± 0.26
5	8410 ± 190	9.3 ± 1.8	1.58 ± 0.22

SITE I-IN METAL CONCENTRATION (ppm) BY FRACTION			
	ANALYTE		
FRACTION	Fe	Ni	Pb
1	3.0 ± 1.7	<0.5	0.13 ± 0.08
2	31.7 ± 41.9	1.0 ± 0.4	0.47 ± 0.21
3	4070 ± 200	5.1 ± 1.6	1.67 ± 0.31
4	4230 ± 850	4.5 ± 0.5	2.43 ± 0.21
5	10600 ± 1450	10.5 ± 2.1	1.53 ± 0.22

SITE I-OUT METAL CONCENTRATION (ppm) BY FRACTION			
	ANALYTE		
FRACTION	Fe	Ni	Pb
1	2.3 ± 0.6	<0.5	0.15 ± 0.07
2	25.0 ± 13.2	1.0 ± 0.4	0.30 ± 0.02
3	3610 ± 930	5.5 ± 1.0	1.40 ± 0.28
4	3460 ± 830	3.5 ± 0.7	3.05 ± 0.64
5	7060 ± 3320	9.0 ± 1.9	1.55 ± 0.21

metals and mobilities in soil and perhaps bioavailability. Other changes in metal concentrations appear to be related to the low pH of regional typic soils and the periodic flooding of the sample area. The metals Cr, Cu, Ni and Pb were found to be essentially immobilized by the local soil chemistry

ACKNOWLEDGMENTS

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COASTAL LAND LOSS AND WETLAND RESTORATION

Wetland Loss in the Barataria Estuary: Empirically-Defined Relationships and Multiple Working Hypotheses

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ABSTRACT: This paper reviews four types of data analyses used to examine the usefulness of four hypotheses about the causes of the dramatically high coastal wetland losses (0.86% yr^{-1}) in the Barataria Bay watershed, Louisiana. These hypotheses are that land loss is the result of: (1) hydrologic changes resulting from an extensive dredged canal and spoil bank network, (2) a decline in sediments in the Mississippi River during the 1950s, (3) Mississippi River navigation and flood protection levees, and (4) salinity changes. These four landscape scale analyses reveal several patterns in land loss: (1) The formation of ponds is highest near canals, whereas ponds that revert to wetland are relatively scarce closer to canals; (2) When canal density is high, land-loss rates are high, and when new canal dredging slows, land-loss rates stabilize or decrease; and, (3) The amount of land loss per area of dredged canal is highest where the hydrologic gradient is highest (from tides), and not in the estuarine headwaters. The hydrologic change hypothesis is the only acceptable hypothesis of the four hypotheses tested. It explains the rise and fall of land-loss rates in time and space in the Barataria watershed and is a more efficient explanation than alternative hypotheses.

Introduction

The possible causes for the dramatic coastal wetland losses in Louisiana (0.8%/yr from 1955/56 to 1978) are well-known and many (Turner 1990; Boesch et al. 1994). Putting these causes in order of importance is useful for management and restoration programs, and can be formally attempted by formulating testable hypotheses. The hypothetico-deductive method is particularly useful when it yields results that eliminate competing hypotheses, or leads to a refinement of the original conjecture that describes and explains phenomena. A general problem with coastal science, however, is that the scale of estuarine landscapes is such that we cannot easily tease apart and manipulate the ecosystem in a laboratory, or clone the estuary, or use replicated plots in landscape scale experiments. Indirect extrapolations, such as done using computer and statistical models, are also fraught with mis-applications of the dearth of data, but that is sometimes all we have to work with.

The results of testing hypotheses about the causes and consequences of Louisiana's coastal wetland losses may not be trivial of course, because the rationale for restoration and rehabilitation is often to reverse or remove the stressors causing the conversion of land to water. For example, if salinity increases caused land loss, then the idea is that restoration will occur if additional freshwater is introduced to the estuary. Understanding the cause-and-effect relationships between wetland loss (or gain) is important when, as in the case of this coast, restoration projects are often large (>100 ha) and expensive (>$500,000), and the losses and rehabilitation success are consequential to the dependent fauna, flora and inhabitants.

I review here some tests of the hypothesis "extensive hydrologic changes in the Barataria Bay

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estuary are causally related to the land losses this century." I then compare the strength of this hypothesis to some of the other hypothesized causes of land loss on this coast. There are laboratory and small-scale field trials that support various hypotheses. It seems to me that the most reliable interpretations are based on what happens in the field, and not on the results of computer models, laboratory studies or conceptual diagrams.

The test results discussed herein are derived solely from data derived at a landscape scale. The data set is restricted to a discussion of the Barataria watershed. This watershed is a significant component of the Louisiana coastal zone (314,000 ha) and there are a variety of habitat data available on it. Its eastern boundary is the Mississippi River from which occasional overflowing waters are hypothesized to deliver enough sediments and freshwater to significantly influence the balance of land loss or gain in the receiving watershed, and whose re-introduction would restore the estuary's wetlands. Improving our understanding of the ecological processes operating in this watershed might assist in the management of others.

Direct and Indirect Causes of Wetland Loss

Wetland loss is essentially the same as land loss on this coast (Baumann and Turner 1990). We can discriminate between wetland loss that is a direct consequence of human activities, and the losses that are an indirect consequence of various other factors. The initial habitat conversions from human activities, or "direct impacts", are about 12% of the total land losses in the Louisiana coastal zone from the 1930s to 1990 (Britsch and Dunbar 1993). These direct impacts are almost exclusively the result of dredging for oil and gas exploration and recovery, as well as navigation channels. Additional direct impacts arose from failed agricultural impoundments. The 'indirect losses' make up the remaining 88% of all land losses. Some of the causes of these indirect losses, or impacts, include reductions in sediment supply, dredging, from subsurface fluid withdrawal, or hydrologic alterations. The ratio of direct:indirect impacts resulting from human activities may vary under influences such as global sea level rise, climate changes, soil type, geomorphic framework and age, subsidence or management.

Four Hypotheses

Four hypotheses about the causes of indirect wetland losses in Barataria Bay will be addressed here (adapted from Turner 1997):

H1. The indirect consequences of dredged canals and the resulting spoil banks have led to the majority of land loss since the 1930s.

H2. A decline in suspended sediments of the Mississippi River during the 1950s led to the majority of the indirect land losses.

H3. Mississippi River navigation and flood protection levees, mostly built during the 1930s, led to the majority of the indirect land losses.

H4. Saltwater intrusion from offshore to inland has caused the majority of the indirect land losses since the 1930s.

The effect of geological subsidence and sea level rise are not included in this list because both factors have remained relatively stable this century when the land-loss rates rose and fell. Local subsidence caused by oil and gas fluid withdrawal in Louisiana has been estimated to be relatively insignificant compared to soil subsidence rates (Martin and Serdengecti, 1984; Suhayda 1987). There are clearly long-term variations in wind, and therefore sea level. However, the trend in water-level rise for the last 80 years is essentially linear, and there is no acceleration in relative sea level rise at any tide gage site with a long-term record up to the 1990s (Turner 1991).

The four hypotheses identified above were examined using data on a variety of habitat changes obtained from photo-interpretations of both 7 1/2' and 15' quadrangle maps that cover the Barataria watershed. These analyses include documentation of the number of new ponds forming near canals, the number of ponds filling in near canals, and the amount of land loss and canal density over various

time periods. The major question asked is: "Does land loss result from the hydrologic changes arising from dredged canals and the consequential spoil bank parallel to the canal?"

Spatial Relationships Between Land Loss and Hydrologic Change: 7 1/2' Quadrangle Maps

A newly dredged canal is typically >20 m wide and is 5 m deep and has a spoil bank built from the dredged materials that is several meters wide and many times higher than a natural levee. If hydrologic changes cause wetland loss, then land-loss rates should be higher nearer, rather than farther, from a dredged canal and spoil bank. Wetland fragmentation into ponds is presumed to be the first stage of wetland loss. This hypothesis was examined using data on the spatial distribution of different sized ponds found in 1955/56 and 1978 from sixty-three pairs of USFWS 7 1/2' quadrangle habitat maps (Turner and Rao 1990). Ponds that had merged or enlarged to became part of a larger open-water body during the interval were identified and not included in the analysis. The total land loss examined represented 38% of the total land loss for the coastal zone in the same period. Some ponds found in 1955/56 were not present in 1978, and the vast majority of ponds were new. The ponds in Barataria watershed (and elsewhere) between 1 and 50 ha were the most numerous (n = 1104) and mostly formed after 1955/56 (n=935).

The appearance of new ponds and the persistence of existing ponds was positively related to the distance from the canal (Fig. 1, top panel). More than half of all new and persisting ponds, but less than 10 % of the ephemeral ponds, were within 1 km of a canal. If canals had no effect on new pond formation, then the distribution pattern of new, persisting and ephemeral ponds should overlap. The frequency distribution of these ponds do overlap within three km of the canal. The greatest disparity between the distribution of ephemeral and new ponds was within 1 km of the canal.

The area of new ponds between 0 and 60 ha that formed between 1955/56 and 1978 in each quadrangle map was positively related to the area of canals in 1978 (Fig. 1, bottom panel). A linear

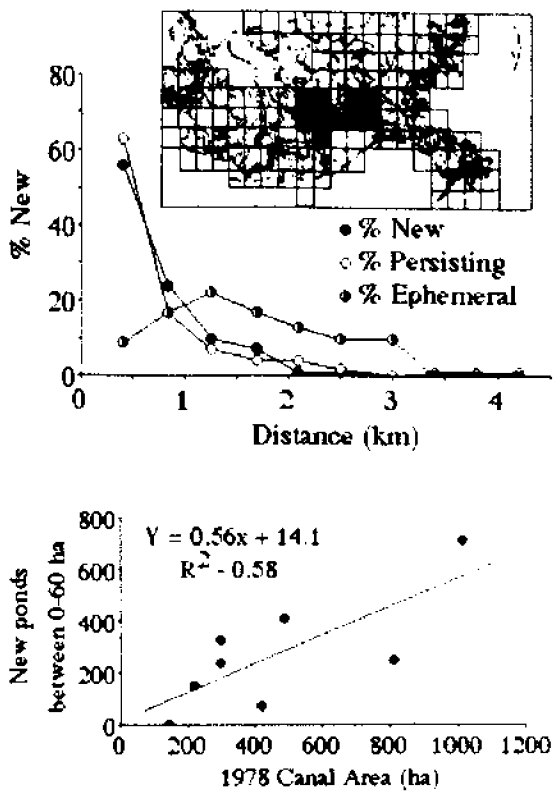


Fig. 1. Top: The relationship between the percent of ponds that are new, persisting, and ephemeral (for the interval 1955/6 to 78) and distance to the nearest canal. Bottom: The relationship between the area of new ponds <60 ha formed between 1955/6-78 and canal surface area (ha) in Barataria estuary (adapted from Turner and Rao 1990).

regression of the two variables (pond area and canal area) gave an intercept statistically indistinguishable from zero ($p = 0.03$). These patterns were also documented in the neighboring St. Bernard and Terrebonne watersheds (Turner and Rao 1990). The hypothesis that canals and spoil banks caused new pond formation is not rejected.

Temporal Relationships Between Land Loss and Hydrologic Change: Salt Marshes

The temporal relationships between canal area and land loss for the St. Bernard, Barataria, and Terrebonne watersheds were investigated by Bass and Turner (1997) using aerial photographs.

Twenty-seven salt marsh sites were used in the analysis that included 1,215 ha of land loss from 1955/56 to 1990 in a study area of 6,367 ha. The land was > 95% salt marsh habitat at all sites and for all map dates. Six sites were in St. Bernard watershed, nine sites in Barataria watershed, and twelve sites in the Terrebonne watershed. The study boundaries were determined by stable landmarks. Bass and Turner (1997) conducted an ANOVA (Analysis of Variance) on the combined data.

There was a significant relationship between canal density (canal area/land area) and the percentage open water (slope = 1.85, Pr>/T = 0.0013) for all sites combined. The slope is defined as the change in Y axis (open water) with respect to the X axis (canal density). There was a 1.85% increase in non-dredged open water in addition to the actual canal area dredged for every one percent increase in canal density for the 35-year study interval. The ANOVA provided an estimate of the Y intercept (land loss) without the area of canals dredged since 1955/56, which was 0.71%/y for the Barataria watershed. This intercept represents an estimate of the 'background' land-loss rates without the effects of canals existing before the 1955/56 mapping interval. It does not represent the land-loss rates without any canals of any age. The effects of canals were additive to the effects of time (Pr>F = 0.0001), which also had a statistically significant interaction with watershed (Year*Watershed; Pr>F = 0.0001). Bass and Turner (1997) concluded that canals and associated spoil banks were significantly and directly correlated with wetland loss. No differences in the effect of canals on land loss were detected among watersheds (i.e., the canal* watershed interactions were not significant).

An intriguing aspect of the relationship between canal density and land loss is the stabilization and decline in the area of open water (exclusive of dredged surface area) when the new dredging activity ceased (in 1974 to 1978, and 1988 to 1990). This result implies that land loss may be reversible in these salt marshes.

Spatial and Temporal Relationships Between Land Loss and Hydrologic Change: 7 1/2' Quadrangle Maps

Scaife et al. (1983) analyzed the USFWS 7 1/2' quadrangle maps of habitat change for the Louisiana coastal zone in a search for patterns between land loss (1955/56 to 1978) and various geological, physical, and anthropogenic factors. Their analysis of the St. Bernard delta complex included mapping units of the present day Barataria watershed. There was a very strong linear relationship ($R^2 = 0.94$; $p < 0.01$) between canal density (in 1978) and land loss from 1955/56 to 1978, when normalized for the amount of land present in 1955/56; when the percentage of land dredged was high, the percentage of land loss was also high (Fig. 3).

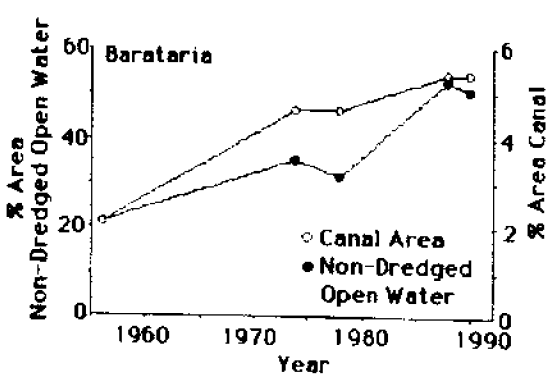


Fig. 2. The percent non-dredge open water and percent canal area vs. year for salt marsh sites in Barataria estuary (adapted from Bass and Turner 1997).

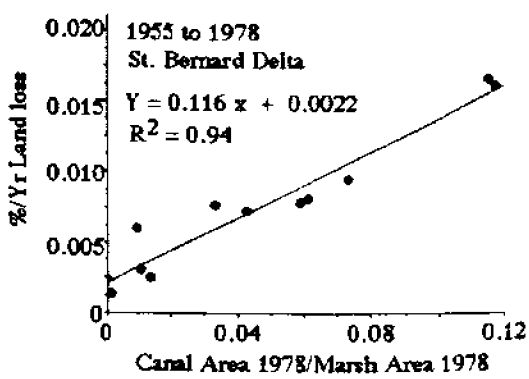


Fig. 3. Land-loss rates from 1955/56 to 1978 in 7 1/2' quadrangle maps selected because they include the geologically-defined St. Bernard delta complex, which underlies part of the modern Barataria watershed (adapted from Scaife et al. 1983).

Spatial and Temporal Relationships Between Land Loss and Hydrologic Change: 15' Quadrangle Maps

Data from 15' quadrangle maps are available (Britsch and Dunbar 1993) to test for a spatial and temporal relationship between land loss and hydrologic changes. Britsch and Dunbar's land inventories for coastal Louisiana from the 1930s to 1990 were derived from colored overlays on 15' base maps (approximately 63 X 10^3 ha). They used these overlays to map the open-water habitat changes between mapping intervals. These authors used a consistent photo-interpretation method that is slightly different from methods others have used. The data set is based on gross land-loss rates, rather than net land-loss rates, and represents the only data set of its kind (for Louisiana) that goes back to the 1930s and that covers the whole coast in a consistent manner. The mapping dates were from the 1930s (range 1931 to 1949; μ=1934), the mid-1950s (range 1951 to 1958; μ=1957), 1974, 1983 and 1990. Britsch and Dunbar (1993) classified *Man-made loss* as land that became open water as a direct consequence of human modification. *Natural loss* was all other land loss. The 'Man-made loss' in the Britsch and Dunbar (1993) analysis is the same as what I considered to be direct land loss in this paper. The 15' quadrangle maps included all of the Barataria watershed.

Turner (1997) used all of these data except those maps with >85% open water, or <10^4 ha land within a 15' quadrangle map. Only one of nine maps that included the Barataria watershed was excluded in that analysis. This one map included the Delta Farms region, where an agricultural impoundment failed in the 1960s and the area became open water. This dramatic conversion to open water represented the vast majority of land loss from the 1930s to 1990 for that quadrangle map.

There was a strong, positive relationship between the direct land loss within each 15' map in the Barataria watershed and the indirect land loss (Fig. 4; $p < 0.01$ in all cases). The intercept was zero, or less, indicating that no significant net land loss would occur without dredging. Identical results were obtained for the neighboring Terrebonne and St. Bernard watersheds (Turner 1997).

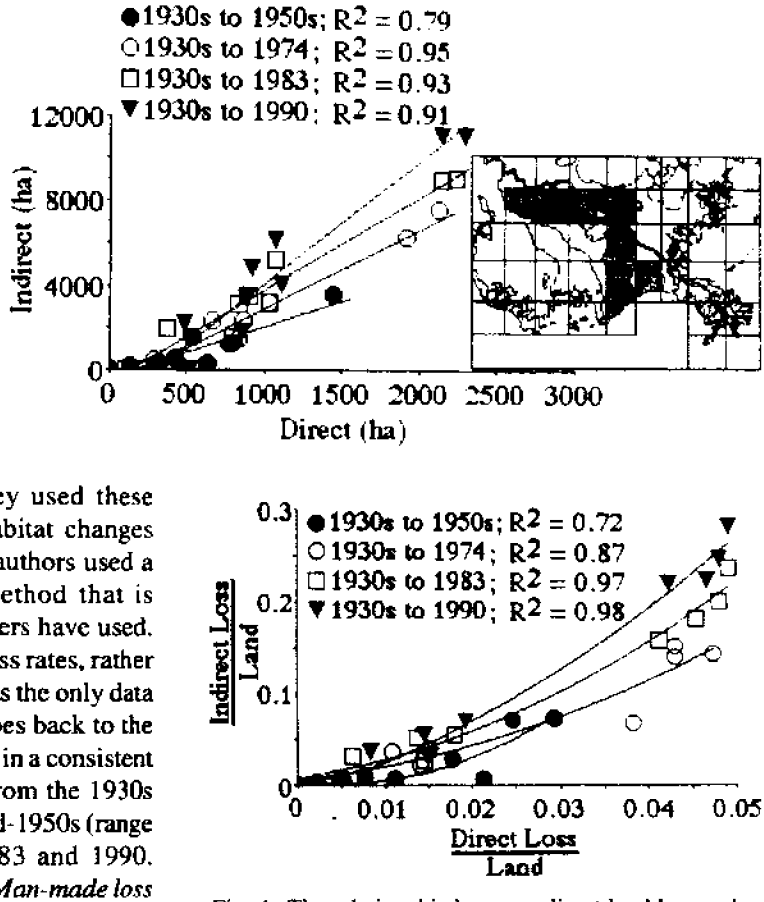


Fig. 4. The relationship between direct land loss and indirect land loss (primarily canal density) in the Barataria watershed for eight 15' quadrangle maps analyzed by Britsch and Dunbar (1993). No data was excluded. Four different mapping intervals are compared: 1930s to 1950s, 1930s to 1950s, 1930s to 1974, and 1930s to 1990. Top: Area of direct land loss vs. area of indirect land loss (ha). A linear regression of the data is shown together with the Coefficient of Determination (R^2). ($p < 0.01$ in all cases). Bottom: The percent indirect land loss vs. the percent indirect land loss. A polynomial fit of the data is shown together with the R^2 for each data set ($p < 0.01$ in all cases).

This data set was also used to plot the relative changes in land loss since the 1930s. The first derivative of the polynomial regression equation describing the cumulative land loss as a function of time yielded an instantaneous land-loss rate whose trajectory approached zero in the 1990s. land-loss rates declined as rapidly as they rose. These estimated instantaneous land-loss rates parallel dredging activity, but land loss lagged dredging activity by several years (Fig. 5).

Tests of Competing Hypotheses

There are many reasons why it is not easy to develop landscape-scale data and then use them fruitfully to test competing hypotheses about how hydrologic alterations of the landscape affect the area of coastal wetlands. Underneath the surface of the Louisiana coast are deltas of different ages and composition. The amplitude and energy of the tides within and among watersheds is not homogenous. The density and timing of dredging activity varies among watersheds because of economic and anciently-defined geological factors. Furthermore, if the four hypotheses identified earlier in this paper are complementary, or if the causal significant terms interact, then the effects of dredging should be difficult to tease out because of a low signal-to-noise ratio. In other words, if the competing hypotheses are individually strong or interact with each other, then a plot of direct and indirect land losses should look like the scatter-plot of data points shown in the bottom side of Fig. 6, not the rather strong linear regression shown in the top panel, which is the result for the Barataria watershed ($R^2 = 0.98$; $p = 0.001$). This result suggests that the effect of canal dredging and hydrologic change on land loss is quite robust.

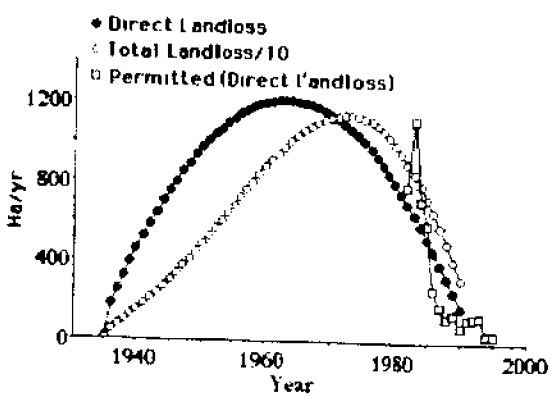


Fig. 5. Land loss and canal dredging over time for the Louisiana coastal zone (from Turner 1997). Permitted area (open squares) is based on the area dredged each year that is permitted by the State's coastal zone management program. The other two estimates are based on a statistical fit of the Britsch and Dunbar (1993) land loss data for the whole coast.

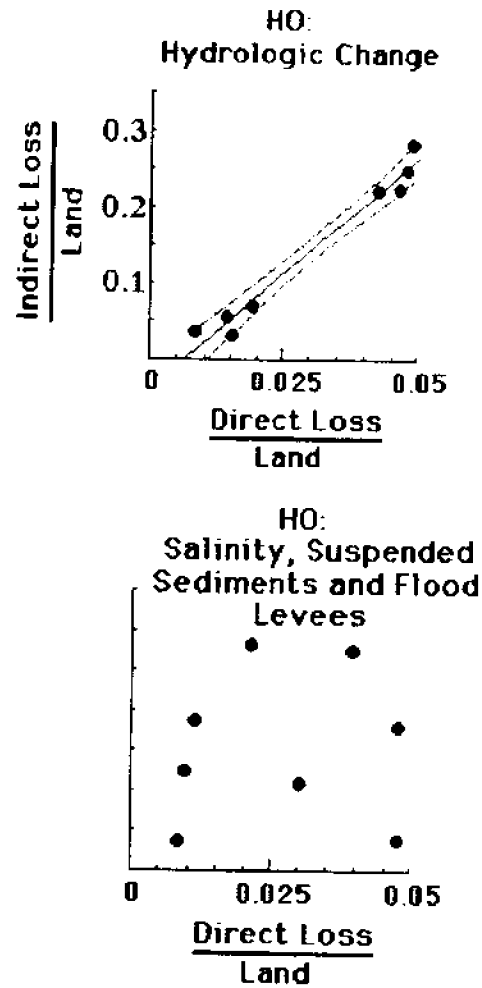


Fig. 6. Top Panel: the hypothesized relationship between indirect and direct loss if the hydrologic change hypothesis is correct. Bottom Panel: the hypothesized relationship between indirect and direct loss if the hydrologic change hypothesis is not correct or if several hypothesis are significant individually or acting in concert. The left panel (from Fig. 4 in Turner 1997) is the actual data for the Barataria watershed.

Another test of competing hypotheses is to compare the relationship between land-loss rates and hydrologic change in the Barataria watershed for the years before and after the significant drop in suspended load occurred in the mid-1950s. If this sediment-load decline was an additional cause of wetland loss, then the intercept of the land-loss and canal-density regression line should be higher after the 1950s, than for data for before the 1950s (Fig. 7). If the decline in sediment loading were an insignificant influence on land loss (during the mapping interval), then there would be no difference in the slope of the regression lines for the two data sets. The hypothesized influenced was not observed (see Fig. 4), so we should reject this competing hypothesis.

The relative effects of increased estuarine salinity (which are not generally observed on this coast; Wiseman et al., 1990) and flood protection and navigation levees on land-loss rates can also be tested with the data set of Britsch and Dunbar (1993; Turner 1997). The hypothesis suggests that land-loss rates will increase if freshwater flow into estuarine headwaters is restricted and plants become stressed by an increase in salinity, or, because of the reduced sediment loading. There are at least three linked assumptions if this hypothesis is true: (1) saltwater changes must occur, throughout the watershed, and not just in one small part of it, (2) these changes must be significant plant stressors and lead to vegetation losses, and, (3) other emergent plant communities do not successfully re-colonize any new niche created.

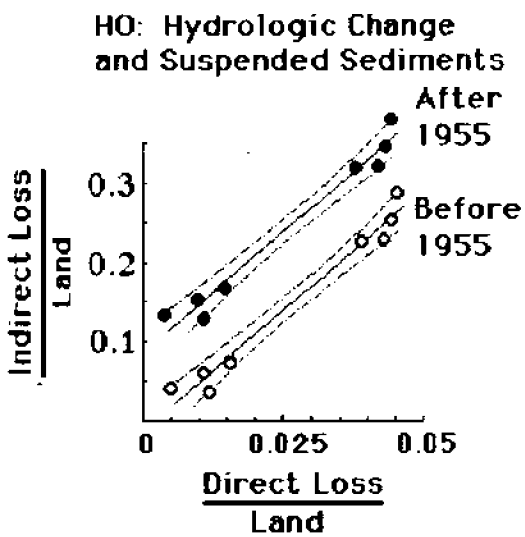
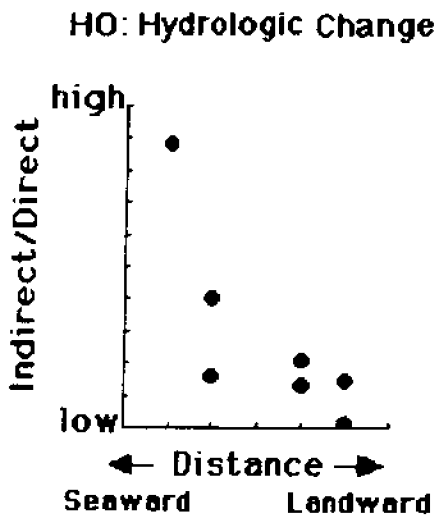


Fig. 7. The hypothesized relationship between indirect and direct land loss for two mapping intervals if suspended sediment loading in the 1950s has an effect on land loss. This pattern was not observed (see Fig. 4).

The 'saltwater intrusion' and the 'sediment deprivation' hypotheses were tested by assigning each 15' base map a number to denote its relative position inland from the estuarine entrance (described in Turner 1997). The quadrangle map closest to the estuarine entrance was the lowest numbered map, and the quadrangle map furthermost from the sea was numbered the highest.

These two hypotheses (salinity stress and sediment deprivation) can be rejected for several reasons, in addition to the previously described relationships. The estuarine headwaters are where overbank flooding previously occurred (but not every year). Plant stress resulting from sediment-deprivation or salt water intrusion should be: 1) lowest for the plants that are most adapted to high salinity and nearest the remaining suspended sediment supply (the estuarine entrance), and, 2) highest for those plants least adapted to salt, i.e., those plants located in the freshwater marshes. If salinity stress occurs, and if it leads to land loss, then the amount of indirect land loss per direct land loss should theoretically have been highest in the lowest salinity zone (bottom panel in Fig. 8). This result was not observed. The indirect:direct land loss ratio declines, not increases, going from salt to freshwater zones along a gradient from the estuarine mouth to inland. This ratio does not decline in the region where plant stresses should be most sensitive to salinity stress (Fig. 9; $R^2 = 0.80$; $p = 0.016$). There is more land loss per dredged channel in the high-salinity zone of the estuary than in the low-salinity zone, and land loss with dredging where flood protection levees are most likely to reduce overbank flooding. Although saltwater intrusion may be significant locally, it does not appear to be a major indirect factor driving land loss this century in this basin. These two hypotheses are therefore rejected.



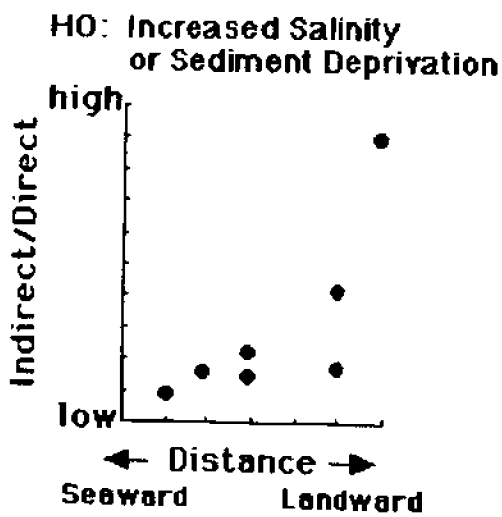


Fig. 8. The hypothesized relationship between the indirect loss rates (Y axis) vs. direct losses (X axis) from the 1930s to 1990 for each estuary as a function of distance from the coast (data are for the 15' quadrangle maps examined by Britsch and Dunbar (1993). Only quadrangle maps with <85% open water and >8100 ha are included. The top panel indicates the results necessary to support the hydrologic change hypothesis, and the bottom panel is the anticipated result if increased salinity, decreased suspended sediments or both is significant. The actual results are that shown in the top panel and Fig. 9.

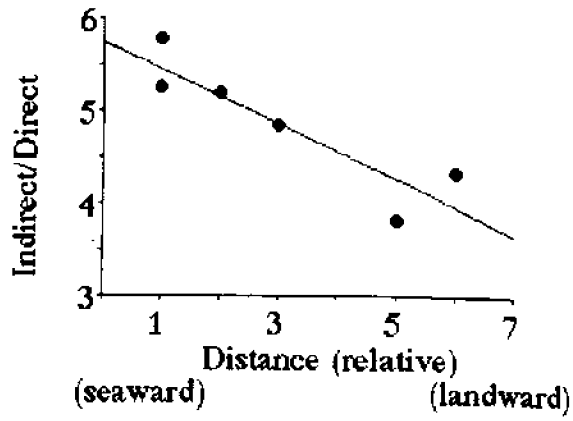


Fig. 9. The relationship between the land loss per area canal (Y axis) and the distance to the coast in 15' quadrangle maps. Land loss is from Britsch and Dunbar (1993) and is for the 1930s to 1990. The distance measure is a simple map sequence from coast to inland.

Conclusions

The hypothesis that hydrologic change is the primary cause of wetland losses in the Barataria watershed was tested in various ways using data on landscape-scale habitat changes. Comparative tests of competing hypotheses were also made to examine the relative strength of the influence of salinity changes, suspended sediment load reduction, and flood protection levees on land-loss rates. The results were consistent among several tests. The formation of ponds is highest near canals, whereas ponds that revert to wetlands are relatively scarce near canals. When canal density is high, land-loss rates are high, and when canal dredging slows, then land-loss rates stabilize or decrease. The amount of land loss per area of dredged canal is highest where the hydrologic gradient is highest (from tides), and not in the estuarine headwaters. The hydrologic change hypothesis is the only acceptable hypothesis of the four that were tested. It explains the rise and fall of land-loss rates in time and space in the Barataria watershed and is a more efficient explanation than alternative hypotheses (the application of Occam's Razor).

These conclusions obviously have some consequences for conserving the existing wetlands and for their rehabilitation and restoration. One

consequence is that the role of plants in wetland stability should be considered as more than a reaction to patterns in inorganic sediment rates. Plants are major contributors to wetland accretion. Plant ecologists could contribute to wetland restoration by investigating belowground plant organic storage pools; wetland hydrologic studies should be expanded to include landscape-scale interactions. We have much to learn.

Landscape-scale analyses bring a unique set of complications to ecosystem science. Obviously landscapes are not homogenous, and therefore variability is introduced when comparing one watershed to another. Further, varying the size of the measuring unit may confound detection of the actual pattern. An analogy might be the example of trying to determine if the earth is curved. Although the earth's surface is easily seen as curved when flying long distances, its shape does not appear to be round when walking. One cannot quantitatively detect this curvature with ordinary surveying instruments as easily along a 10 m path, as along a 10 km path. If a 100 km measuring path (or 100 km^2 plot) has a mountain range, then an even longer path may be needed to compensate for the local variation introduced by geological history. The relationship between hydrologic change and land loss may not seem evident by analyzing a 1-ha pixel, instead of a 100-ha pixel, or if the wetland is encompassed by stranded beach ridges, or embedded within the Mississippi River delta, or isolated by remnant channel levees buried a few meters below the surface. This situation leads one to understand that the interrelationships between wetland hydrology and plant health may extend over kilometers, and that evaluating these connections may involve some rather innovative and humbling failures before we fully appreciate the proper scales necessary to investigate and quantitatively predict the consequences of coastal wetland management.

ACKNOWLEDGMENTS

These analyses accumulated over many years, and especially through the often vigorous discussion with colleagues and the hardwork of the various co-authors and students who shared the research efforts described herein. Students of the Louisiana State University course "Wetland Loss, Restoration and Management" are especially thanked for their patience and willingness to learn, share, and critique the various ideas that survived here, and for help in burying others.

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Pattern and Process of Land Loss in the Louisiana Coastal Zone: An Analysis of Spatial and Temporal Patterns of Wetland Habitat Change[a]

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ABSTRACT: An earlier investigation (Turner 1997) concluded that most of the coastal wetland loss in Louisiana was caused by the effects of canal dredging, that loss was near zero in the absence of canals, and that land loss had decreased to near zero by the late 1990's. This analysis was based on a 15-minute quadrangle (approximately 68,000 ha) scale that is too large to isolate processes responsible for small-scale wetland loss and too small to capture those responsible for large-scale loss. Herein, we conduct a further evaluation of the relationship between direct loss due to canal dredging and all other loss from 1933-1990 using a spatial scale of 4,100 ha that accurately captures local land-loss processes. Data sets for the Pontchartrain and Breton basins did not meet the criteria for parametric statistics. Regressions of other wetland loss on canal area (i.e., direct loss) for the Birdfoot, Terrebone, and Calcasieu basins were not significant. Significant positive curvilinear relationships were isolated for the Barataria (R^2=0.40) and Mermentau (R^2=0.29) basins, indicating that the extent of canals is significantly related to wetland loss in these basins. A significant negative relationship (R^2=0.361) was found for the Atchafalaya coastal basin which had statistically lower loss rates than the other basins as a whole. When the data were combined for all basins, 9.2% of the variation in other wetland loss was attributable to canals. All significant regressions intercepted the Y-axis at positive loss values indicating that some loss occurred in the absence of canals. We agree with Turner that canals are an important agent in causing wetland loss in coastal Louisiana, but strongly disagree that they are responsible for the vast majority of this loss. We conclude that wetland loss in the Mississippi delta is an ongoing complex process involving several interacting factors and that efforts to create and restore Louisiana's coastal wetlands must emphasize riverine inputs of fresh water and sediments.

From the Symposium *Recent Research in Coastal Louisiana: Natural System Function and Response to Human Influence.* Rozas, L.P., J.A. Nyman, C.E. Proffitt, N.N. Rabalais, D.J. Reed, and R.E. Turner (editors). 1999. Published by Louisiana Sea Grant College Program.

[a]**This paper is a condensed version of an article that can be found in Volume 23 of Estuaries and is printed here with the permission of the Estuarine Research Federation.**

Introduction

Turner (1997) analyzed wetland loss in coastal Louisiana wetlands from 1932 to 1990 using statistical analyses of land to water changes in 15-minute quadrangle maps. He concluded that virtually all of the loss was caused by canals, that land loss was near zero in the absence of canals, and that land loss rates declined to near zero by the late 1990s. Turner explicitly excluded saltwater intrusion and the leveeing of the Mississippi River as important factors contributing to land loss because land loss was highest near the coast and because regressions of direct versus indirect land loss had zero intercepts. In this paper and in Turner (1997), direct land loss refers to wetland which becomes water when canals are dredged and indirect land loss refers to all other wetland loss. Turner's conclusions contradict Penland et al. (1996), who concluded that about 46% of wetland loss in coastal Louisiana has occurred through natural processes.

The objective of this paper is to carry out a further evaluation of the causes of land loss in the Louisiana coastal zone, particularly with respect to the effects of canals. A more detailed version of this paper (Day et al. 2000) may be found in the journal *Estuaries*. This reduced version has been printed with permission of *Estuaries*.

Turner (1997) stated four hypotheses about the coast wide causes of land loss and tested them with various statistical analyses of direct and indirect land loss rates from 15-minute quadrangle maps which contain about 68,000 ha. We have incorporated the essence of these in the following hypotheses which we test in this paper.

Hypotheses

1. Direct land loss (i.e., due to canals) is quantitatively related to land loss in general, both for individual hydrologic basins and for the entire coast, indicating that most land loss can be attributed to canals.

2. When direct land loss is zero, other land loss is close to zero (i.e., the intercept in regressions of other land loss on direct land loss is zero).

3. Land loss is highest near the coast and decreases inland. This implies that land loss is highest in salt marsh and lowest in fresh marsh and that saltwater intrusion has not been an important factor in land loss.

4. Restricted riverine input has had little impact on land loss. A corollary to this is that land loss in wetlands in the Atchafalaya Delta region, which is not leveed and receives approximately one third of the total flow of the Mississippi River, has been the same as that of other basins with similar density of canals. If land loss in the Atchafalaya delta region is low, it would suggest that while lack of riverine input may not have directly caused land loss, riverine input is essential to building new wetlands and reducing loss of existing wetlands.

Materials and Methods

Data Base

Britsch and Dunbar (1993) quantified wetland to water changes during four mapping intervals: early-1930s to mid-1950s (referred to in this paper as 1932-1955; the actual dates of the different maps used varied slightly), 1956-1973, 1974-1982, and 1983-1990. They aggregated their direct and indirect land loss data to the standard 15-minute quadrangle map scale (~68,000 ha or 168,000 acres). This is a convenient scale for maps because it is large enough to present a considerable amount of detail but small enough so that the number of maps which encompass the entire coastal zone (about 50 maps) is not excessive. Britsch and Dunbar also produced six spatial maps that were color coded for each mapping interval (Britsch and Dunbar 1996, Fig. 2 of Day et al. 2000).

However, the 15-minute quadrangle scale is generally too large for statistical analysis of site-specific patterns of wetland loss in coastal Louisiana. The patch size of most of the land loss that has occurred in the coastal zone is considerably smaller than the scale of a 15-minute quadrangle. An examination of the maps of Britsch and Dunbar (1993) shows that most 15-minute quadrangles

contain separate patches of wetland loss that are not spatially or functionally related to each other (see Fig. 2 of Day et al. 2000). For example, in a detailed study of the proximity of canals and wetland land loss (Leibowitz 1989), no relationship existed for land loss greater than 5 km from the canals.

Data Analysis

To better approximate the location and scale of many of the land loss processes and patterns (including oil and gas fields where most canals occur), we subsampled the maps of Britsch and Dunbar (1996) at a 4,100 ha (10,100 acres) or 6.4 by 6.4 km cell size, which approximates the size of an average oil field network or a large marsh management impoundment (Cahoon and Groat 1990). *A priori*, the total number of cells (n=121) was determined by budget constraints. These cells were arranged along 19 transects positioned nearly equidistant from each other, such that the number of cells per unit area of the coast was roughly equal. We did not include transects which would have been practically all water (eg., transects in Lake Calcasieu or Lake Pontchatrain). However, to ensure that each basin received at least n=7 cells, it was necessary to place some transects closer together (e.g., the birdfoot hydrologic unit is very narrow compared to the others). The transects were positioned perpendicular to the local coastline (either the Gulf shoreline or interior bay margins as in the Terrebonne, Barataria, Breton, and Pontchartrain basins) and varied in length because they extended to the inland extent of the mapped wetland, but did not overlap one another. While this process was not completely random, the placement of the transects was made without regard to overall land loss patterns and, once the coastal cell of each transect was placed, all cells along the transect were fixed in place.

In each cell, direct (due to canal dredging) and all other land loss (termed natural loss by Britsch and Dunbar 1996) was determined for each sampling interval. Since Britsch and Dunbar only classified changes from land to water, the change from marsh to spoil bank adjacent to each canal is not counted as loss. Britsch and Dunbar did not classify land loss by habitat type. To determine the wetland habitat type in each cell, we overlaid vegetation classification maps on the habitat loss maps. For the 1932-1955 loss period, we used the 1949 map of O'Neil (1949). In a similar manner, for the 1955-1973, 1974-1982, and 1983-1990 loss intervals, we used vegetation classification maps for 1968 (Chabreck et. al. 1968), 1978 (Chabreck and Linscombe 1978), and 1988 (Chabreck and Linscombe 1988), respectively.

Regression and ANOVA analyses were performed using the SYSTAT 6.1 general linear model. Specific hypothesis tests between direct land loss (i.e., due to canals) and other land loss include: (1) other land loss in the entire coastal zone and in each hydrologic basin is statistically related to direct land loss and (2) when direct land loss is zero, other land loss is zero; (3) land loss is highest near the coast and in salt marshes (for this test, distance from the coast was defined by the sequence of each cell in the subsampling transects); and (4) land loss in the Atchafalaya Delta hydrologic unit is not different from that in other hydrologic units. Hypotheses 1 and 2 were tested with regression models, whereas 3 and 4 were tested as separate one-way ANOVAs using basin, cell number in the transects, and marsh type as categorical variables.

We did not eliminate cells containing less than 15% land in 1933 as in Turner (1997) because this potentially creates two biases. First, by definition, these cells are located on shorelines where the primary cause of land loss is wave erosion. Penland et al. (1996) reported that about 35% of land loss was due to wave erosion. Second, very few oil field canal networks are located directly on the Louisiana coastline and hense elimination of coastal cells may improve the regression fit.

Log transformations of the dependent or independent variable, or both, were required in several instances when the data did not meet the criteria for parametric analysis and when relationships were curvilinear. Nonparametric regression was required for the Pontchartrain and Breton Sound data sets (Burkes and Dodge 1993). When data contained an observation with excessive leverage (i.e., > 0.55) the results are presented with and without that observation (Hair et al. 1998). For the

Barataria, Atchafalaya, and combined data set, one observation required omission because it produced an excessive (>±3.0) Studentized residual (Hair et al. 1998). Unless specified, differences were deemed statistically significant at alpha = 0.05.

An attempt was made to analyze the data standardized to the amount of land present at each site in 1932 (as in Turner 1997). That is, the direct and other loss values were proportionalized to the amount of land in each cell at the beginning of the study. These data required an arcsine square root transformation to meet normality and homogeneity of variance assumptions (Sokal and Rohlf 1995).

Results

Regression: For all the data standardized to the amount of land present in 1932, no significant relationships occurred, regardless of transformation (Fig. 1). Interestingly, when cells were removed that contained less than 15% land in 1932 (see Fig. 4 of Turner, 1997, for comparison), the fit did become marginally significant ($F_{1,113}$ = 4.16, $p<0.044$) albeit weak (R^2=0.036). For the analyses by basin, standardization did not improve the fits; therefore, all subsequent interpretations are limited to the non-standardized data.

Statistically significant relationships between other land loss and direct land loss were not obtained for the Birdfoot, Terrebonne, and Calcasieu hydrologic units (best fits producing $p\geq0.12$, Fig. 2). Positive relationships existed for the Breton Sound ($F_{1,5}$=10.37, p=0.023, Fig. 3a), Barataria ($F_{1,22}$=56.97, p<0.0001, Fig. 3b), and Mermentau basins ($F_{1,20}$=10.73, p=0.004, Fig. 3c), producing R^2=0.675, R^2=0.721, and R^2=0.349, respectively. For the Barataria data set, omission of the single observation on the Y axis (with leverage=0.57, Fig. 6a) decreased the R^2 from 0.721 to 0.470. Interestingly, in the Atchafalaya, which is a basin characterized by high input of riverine inorganic sediments, a significant negative relationship existed

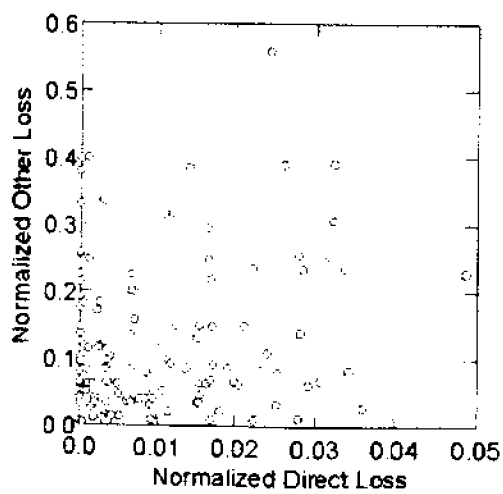

Fig. 1. Direct wetlands loss and other wetlands loss for all basins normalized to the land present in each quadrat in 1933 (as in Fig. 4 of Turner 1997). n = 121.

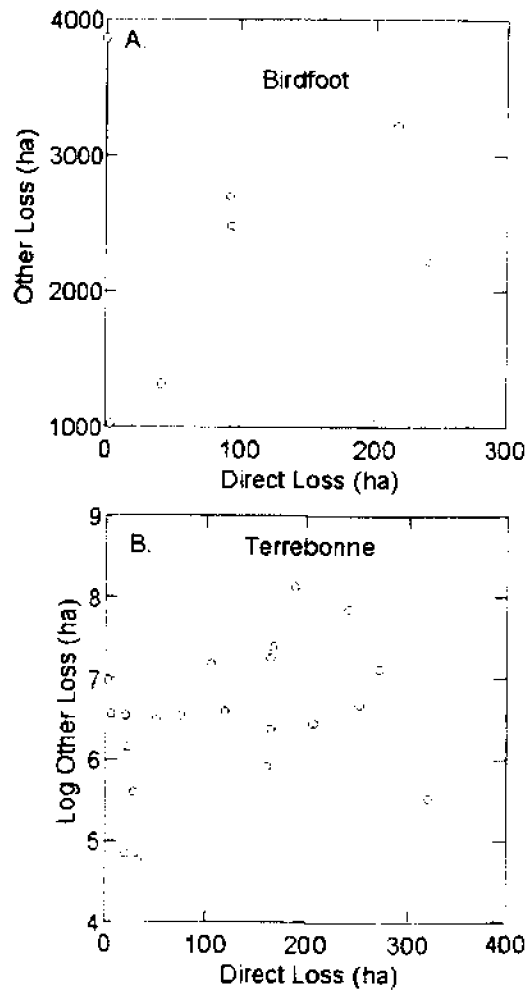

Fig. 2. Direct land loss and other land loss for the (A.) Birdfoot and (B.) Terrebone hydrologic units from 1933-1990. Data for Calcasieu Basin are not shown as direct loss was zero for all quadrats.

between other wetland loss and direct loss ($F_{1,16}$=4.93, p=0.041, Fig. 3d), producing R^2=0.236. For the three parametric fits (Barataria, Mermentua, and Atchafalaya), the intercept (b_0) was statistically greater than zero, indicating that there was wetland loss in the absence of direct losses.

The regression of other wetland loss on direct wetland loss with all of the data combined could account for only 9.2% of wetland loss attributable to canals ($F_{1,118}$=12.00, p=0.001, R^2=0.092, Fig. 4). The intercept of b_0=644 ha was highly significantly different from zero (t=6.60, p<0.0001).

ANOVA: The data set with all basins combined was also used to test for differences in other wetland loss from 1932 to 1990 due to effects of basins, the distance from the coast, and marsh type (Fig. 5). The basin effect was highly significant for the raw ($F_{7,112}$=9.56, p<0.0001, Fig. 5a) and standardized ($F_{7,111}$=8.71, p<0.000) data and the *a priori* linear contrast of the Atchafalaya versus other basins was highly significant ($F_{1,112}$ =12.97, p<0.0001 and $F_{1,111}$ =11.37, p<0.0001 for the raw and standardized data, respectively), despite relatively low loss rates in the Breton Sound and Pontchartrain basins (Fig. 5a). Land loss did not differ from the coast inland for the raw data ($F_{7,108}$=1.32, p=0.247), but did for the normalized data ($F_{7,107}$=2.68, p=0.014) with the highest losses occurring in both coastal and the most inland cells (Fig. 5b). Wetland loss did not differ for different marsh types ($F_{2,169}$=2.72, p=0.069), but displayed a similar pattern as that of distance from the coast (Fig. 5c).

Discussion

The results of our analysis indicate that canals have been an important factor in land loss, but that the relationship with land loss varies between basins. For Barataria Basin, up to 72% of other wetland loss was statistically associated with direct losses due to canals. However, omission of a single outlier decreased the R^2 to 0.47, or by 35%. From both an ecological and statistical perspective, we believe that a single observation with that much leverage warrants omission. There was no significant relationship between direct land loss and other land loss for

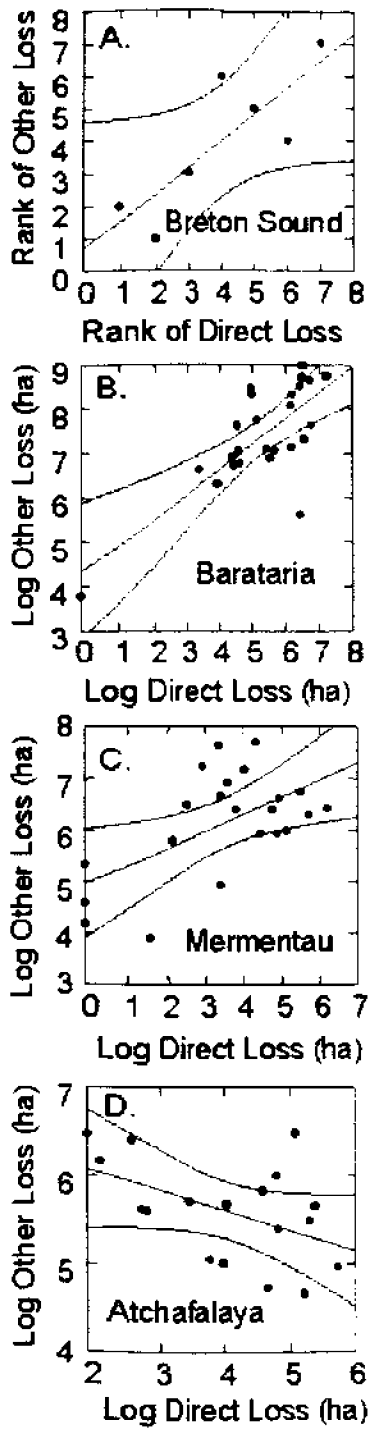

Fig. 3. Regressions of other wetlands loss on direct wetlands loss for (A.) Breton Sound, (B.) Barataria, (C.) Mermentau, and (D.) Atchafalaya basins from 1933-1990. 95% confidence intervals of the mean are included.

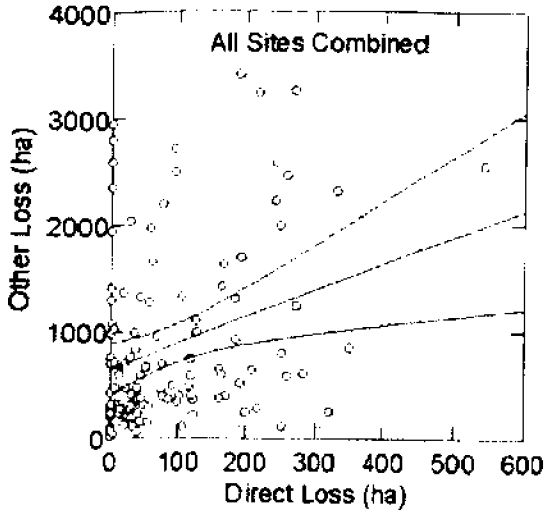

Fig. 4. Regression of other wetland loss on direct wetland loss for all hydrologic units combined for period of 1933-1990. 95% confidence limits of the mean are included.

several of the basins (Pontchartrain, Birdfoot, Terrebonne, and Calcasieu). This does not mean, of course, that land loss has not been caused by canals in these basins. Rather, it indicates that other processes are also important and mask the potential statistical relationship, or that insufficient statistical power existed due to low sample sizes. There was no clear pattern in land loss with distance from the coast or for different types of marsh, and in general the Atchafalaya hydrologic unit, with the highest riverine input, has had a significantly lower rate of wetland loss than the other basins.

Although the 15-minute quadrangle is a convenient mapping scale for a variety of purposes, it commingles (i.e., aliases) the functional processes responsible for coastal wetland loss in Louisiana. In a detailed study of the proximity of canals to patches of high wetland land loss, Leibowitz (1989) concluded that canals were not the primary cause of the loss. In addition, when positive relationships between canals and land loss existed, the relationship decomposed within 5 km of the canal. Our quadrats were 6.4 km on a side and therefore should have captured positive relationships crisply, had they existed.

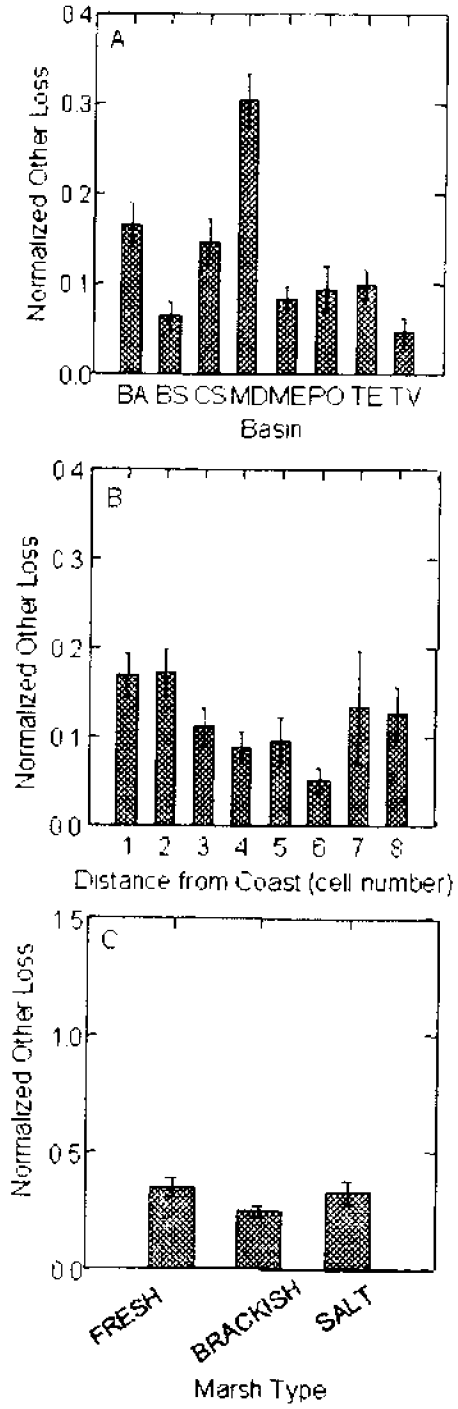

Fig. 5. Differences in wetland loss rates from 1933-1990 among (A) basins, (B) distance from the coast (Cell 1 is nearest the coast), and (C) marsh type. Standard error bars are included. For basins, BA is Barataria, BS is Breton Sound, CS is Calcasieu, MD is Mississippi Delta, ME is Mermentau, PO is Pontchartrain, TE is Terrebonne, and TV is Atchafalaya.

We found that direct land loss (that is, dredged canals) ranged from being not statistically related to other land loss in several basins to accounting for 47% and 68% of the variation in other land loss in the Barataria and Breton Sound basins, respectively. Based on regression analysis, Turner (1997) concluded that canals are responsible for the majority of wetland loss and that when direct loss is zero, other land loss is zero. Our analysis demonstrates that this is not the case. The scale of the 15-minute quadrangles used by Turner leads to statistical relationships between canals and other land loss which is clearly unrelated to canals (i.e., shoreline wave erosion). Penland et al. (1996) also concluded that land loss was due to multiple interactive causes and indicated that about half of the land loss was natural and that the total amount of land loss indirectly due to "submergence" because of human modifications was about 35%. Among the causes of land loss other than canals were wave erosion and tectonic faulting.

In general, we found a lower overall statistical relationship between direct and other land loss than is suggested by the results of Penland et al. (1996). For the combined data set, the R^2 was only 0.092. One reason for this is that the relationship within each basin varies from linear to log-linear to log-log to no relationship to a negative relationship in the Atchafalaya Delta area.

An important detrimental impact of canal spoil banks is that they lead to the reduction of sediment input and poor drainage of marsh soils (Reed 1992; Boumans and Day 1994; Cahoon 1994; Cahoon et al. 1995b; Reed et al. 1997; Swenson and Turner 1987). The progressive waterlogging due to reduced sediment input can interact with existing salinity to produce deleterious effects on vegetation. A number of studies have shown that multiple stresses, such as salinity and waterlogging, have a much more detrimental impact on coastal vegetation than a single stressor (Grime 1979; Mendelssohn and McKee 1988; McKee and Mendelssohn 1989; Grace and Tilman 1990; Shaffer et al. 1992).

We found no clear pattern in land loss rates with distance from the coast, or across different marsh types, suggesting that there is no general pattern of land loss across the salinity gradient. There are documented cases, however, where we believe saltwater intrusion was responsible for land loss. For example, huge marsh losses occurred west of the northern part of Lake Calcasieu during the 1956-1973 mapping interval. These losses occurred after the completion of the Calcasieu Ship Channel in 1941 and followed the passage of Hurricane Audrey in 1957. The hurricane apparently led to massive saltwater intrusion and widespread death of the freshwater *Cladium* marshes which previously occupied the area (Morgan et al. 1958). The remaining marshes are now intermediate to brackish. The construction of the Mississippi River Gulf Outlet led to saltwater intrusion and caused the death of almost all of the *Taxodium* swamps which formerly occurred east of the Mississippi River below New Orleans (Coastal Environments 1972). Some of this area is now open water, but much of the swamp has converted to *Spartina* marsh scattered with ghost cypress trunks.

Turner (1997) concludes that the isolation of most of the deltaic plain from riverine input by flood control levees has not played a significant role in wetland loss. This contradicts a long history of research that demonstrates how the river built and maintains the delta (Fisk et al. 1954; Kolb and Van Lopik 1958; Day et al. 1995, 1997; Roberts 1997). The most obvious example of this are the marshes in the Atchafalaya delta region, a non-leveed coastal bay where subaerial land building has continued since 1973 (Roberts et al. 1980). Our analysis demonstrates that land loss in this area is significantly lower than that of any of the other coastal basins.

Turner suggests that there is a "need for much greater ecological understanding" of wetlands and that there is little appreciation of the role plants play in "dominating the accumulation of sediments through their contribution to soil organic matter below ground." We certainly agree on the need for further study on the functioning of these systems. But it has long been recognized that soil formation and accretion in much of the Mississippi deltaic plain is dominated by organic soil formation from root production and management suggestions have explicitly incorporated this function (Hatton et al.

1983; Templet and Meyer-Arendt 1988; Cahoon 1994; Cahoon et al. 1995a; Day and Templet 1989; Day et al. 1995, 1997; Nyman et al. 1993a,b). Organic soil formation often accounts for 70%-80% of accretion, but addition of mineral sediments results in stimulation of plant production and health. Mineral sediment addition, especially in river water, has several positive impacts on marsh plant communities. The mineral sediments add strength and bulk to the sediments, for example, and they carry nutrients that stimulate productivity and iron which complexes with sulfide phytotoxins. Fresh water also reduces salinity stress.

Based on his analysis, Turner (1997) concluded that land loss rates in the coastal zone will be close to zero by the year 2000. But land loss rates continue to be high, with measured loss rates ranging from 65 to 91 km^2yr^{-1} in the 1980s and the 1990s (Barras et al. 1994). In addition, landscape modeling results (Reyes et al. 1999) and statistical projections based on past land loss rates (J. Suhayda, Dept. Civil Engineering, LSU, personal communication) indicate that land loss rates will continue to be high over the coming decades. For his modeling, Turner used a second-degree polynomial regression to fit wetland loss over time and to extrapolate loss rates from 1991-1995. This type of regression can fit only a symmetric parabola, and extrapolation with polynomial regression in general is cautioned (Zar 1996).

Conclusions

We conclude that wetland loss in the Mississippi Delta is a very complex process and that loss is caused by a dynamic and interacting set of processes. We agree with Turner that canals have been, and continue to be, an important agent in contributing to this land loss. We disagree, however, that canals are responsible for a majority of the land loss and that land loss is zero when canal density is zero. The exclusion of sediments, fresh water, and nutrients of the Mississippi River from much of the coastal zone has eliminated a major land building and maintenance mechanism which historically counteracted many of the processes responsible for land loss and thus is a major factor in coastal land loss in Louisiana. We also agree that hydrologic restoration, specifically with reference to canal spoil banks, should be a necessary component of a holistic, integrated delta restoration plan. This alone, however, will have minimal impact if it is not coupled with reintroduction of river water. Mineral sediments will be generally necessary to rebuild and maintain the coast. It is not likely that removal of spoil banks will result in revegetation on a large scale. Several studies have shown that there is rapid loss of elevation of 10-15 cm when plant death occurs (Nyman et al. 1993a,b; Delaune et al. 1994; Kemp et al. 1999). In addition, soil strength in highly stressed, low elevation wetlands is very low and there is very little elevation gain even with high accretion rates (Cahoon et al. 1995b; Kemp et al. 1999). A number of actions can help reduce land loss, but only riverine input can lead to major creation of new land. This agrees with our fundamental understanding of how deltas function both spatially and temporally (Roberts 1997; Day et al. 1997). If Louisiana is ever to achieve no net loss of its coastal wetlands, we believe that it will be necessary to expand restoration strategies to include major river diversions.

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Landscape Modeling in Coastal Louisiana: Regional Mechanisms of Land Loss

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ABSTRACT: A landscape model, built to examine habitat shifts in coastal Louisiana, examined land loss and yearly shifts of marsh habitats in two watersheds. This model served as a tool for regional analysis of potential management alternatives. The model consisted of several modules linked across spatial and temporal scales. The physical module is a vertically integrated hydrodynamic model that advects and disperses salt and sediment. This was coupled with a process based biological module of above and belowground primary productivity and a module for soil dynamics. The coupling of spatial and temporal scales was a salient issue due to the size of the model. Hydrodynamic flows were calculated in a 100 km² grid and one hour time steps, then averaged and interpolated to daily 1 km² maps for use by the biological and soil modules. Resultant land elevation and habitat characteristics were updated annually and fed back into the hydrodynamic model. Simulations of land loss and spatial variability of marsh types were validated against historical habitat maps with an accuracy higher than 0.85 (out of a maximum of 1.00). Future effects of possible management plans were evaluated for a 30-year projection starting in 1988. Results suggested that river diversions and canal modifications had the largest impacts as opposed to coastal protection plans. Creation of levees or any structure intended to slow water flow resulted in higher flooding conditions and subsequent land loss. The model's strength lies in predicting effects of regional management plans such as water diversions and structural landscape level changes.

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INTRODUCTION

Coastal wetlands across southeastern Louisiana have compacted and are being lost as they converted to open water (Wells 1996). Annual variability in mean sea level (MSL) can be several centimeters per year (Baumann 1980), and can result in increased penetration of salinity into wetlands (Penland et al. 1988). Such changes are believed to underlie the general pattern of displacement of freshwater vegetation by more salinity tolerant communities, and vegetation die-off followed by conversion to open water (Roberts 1997; Wells 1996).

Most wetlands in the Barataria - Terrebonne estuarine complex are losing elevation to MSL at variable rates (Penland and Ramsey 1990). Reed (1995) calculated mean annual land loss rates in Barataria at 20.1 km^2 for 1958 to 1978 and 34.5 km^2 for 1978 to 1988, and for Terrebonne at 24.9 and 31.6 km^2, respectively. Reed (1995) also estimated that indirect land loss due to canal dredging could account for more than 30% in Barataria and 10% in Terrebonne. It appears that local anthropogenic modifications have had different effects in each basin and making it difficult to assess changes on a regional basis.

Understanding these habitat changes is critical to assess the long-term effects of proposed restoration approaches. The objectives of this study were to (1) construct a multiple scale process model for the Barataria and Terrebonne watersheds to understand and predict regional habitat change and (2) assess long-term indirect and cumulative impacts of management alternatives as proposed for wetland restoration.

We have been addressing these issues using spatially articulated landscape models (Costanza et al. 1990; Sklar et al. 1985; White 1991; White et al. 1991). These dynamic spatial interaction models have been called Coastal Ecological Landscape Spatial Simulation (CELSS) by Sklar and Costanza (1991). CELSS models incorporate location-specific algorithms that allow feedback between the local processes and the landscape dynamics, so that both the landscape and the intensity of the processes affecting it change throughout time (Boumans and Sklar 1990). This type of feedback has been tested in aquatic modeling programs like LAPTER (Reyes et al. 1994), and used in terrestrial simulation programs PATCHMOD (Wu and Levin 1994) and ECOLECON (Liu et al. 1994).

STUDY AREA

The Barataria-Terrebonne system is an interdistributary estuarine-wetland system located in southern Louisiana. The Barataria basin portion is located between the natural levees of the Mississippi River and Bayou Lafourche. It is roughly triangular in shape with an area of 6100 km^2. The Terrebonne basin is bordered by Bayou Lafourche on the east and the Atchafalaya River on the west and occupies 5500 km^2 (Fig. 1).

Both basins are dynamic systems undergoing change due to natural and human processes. The Barataria basin has been closed to direct river inflow since 1904. Precipitation provides its main source of freshwater; however, the Mississippi River exerts an indirect influence on salinity in the lower basin by reducing salinity in the nearshore Gulf of Mexico (Perret et al. 1971). The Terrebonne basin is directly influenced by the Atchafalaya River. As a result, the western portion of this basin is one of the few locations in southern Louisiana that has experienced net land gain (Boesch et al. 1994).

The vegetative communities in both basins reflect gradients in elevation and supply of freshwater. Marshes occur as bands of salt, brackish, and fresh vegetation from the Gulf inland (Chabreck and Condrey 1979). Fresh marshes give way to swamps and bottomland hardwoods at higher elevations. These communities have overlapping salinity tolerances when grown in the laboratory, but generally competition leads to distinct zonation in the natural setting (Conner et al. 1987).

METHODS

A landscape habitat prediction model was built for each basin. Each is a dynamic spatial model using variable time and spatial scales. Both models use a finite difference, 2-dimensional and vertically

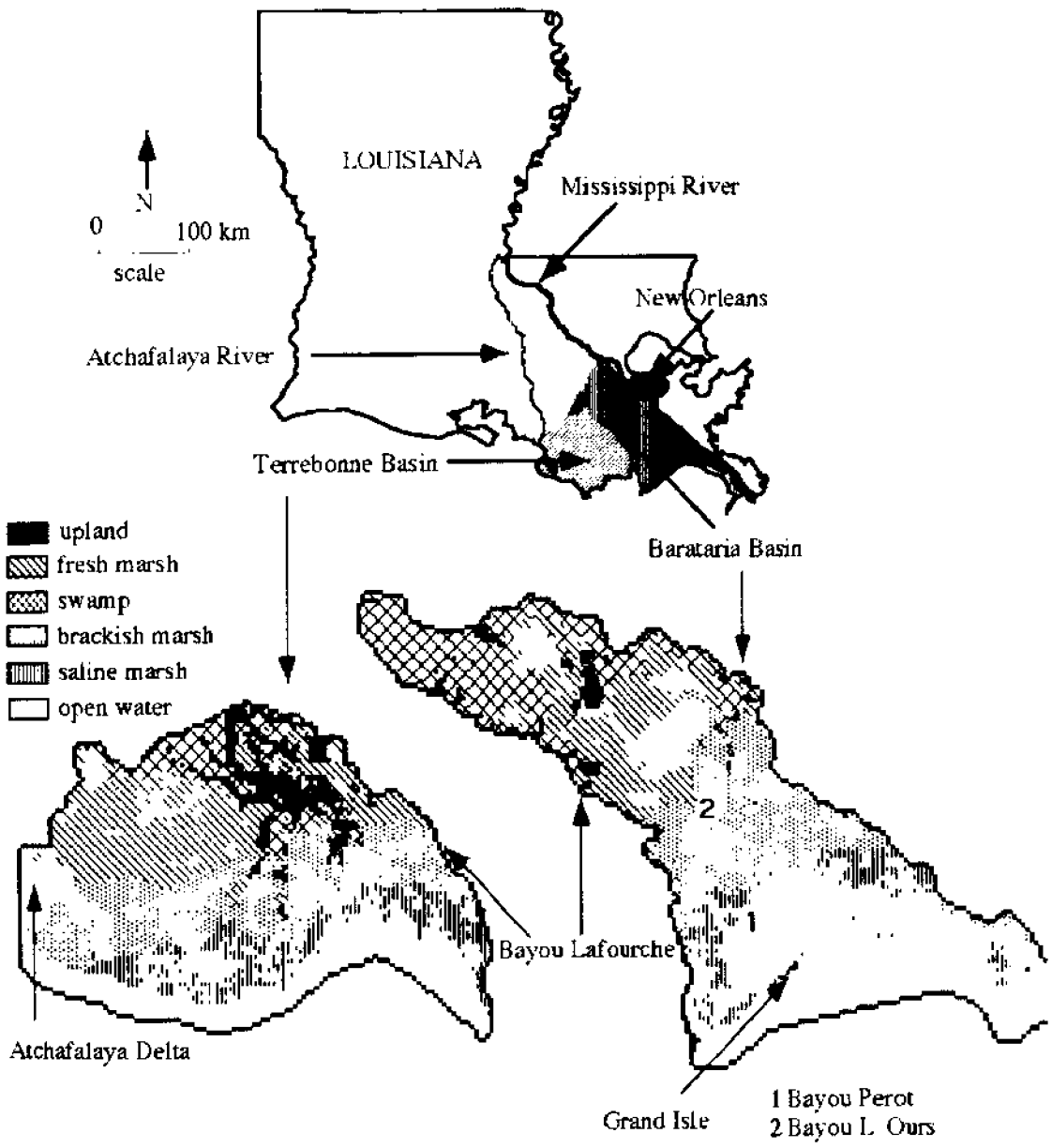

Fig. 1. The State of Louisiana showing location for the Barataria and Terrebonne watersheds. Numbers indicate time series stations.

integrated hydrologic module with a time step (dt) of 1 hour and a spatial cell size of 100 km^2 coupled with a primary productivity module with a dt of 1 day and 1 km^2 cell size. Output from the hydrodynamic and productivity modules are submitted to a soil module (dt of 1 yr., cell size of 1 km^2) and then evaluated by a habitat switching module which redefines the habitat mosaic (cell size of 1 km^2) on a bi-annual basis. The conceptual framework of the four modules is presented in Figure 2. The model was written in FORTRAN and runs on a UNIX platform and specific details for the modules interaction can be found elsewhere (Reyes et al., in review.; White et al. 1997).

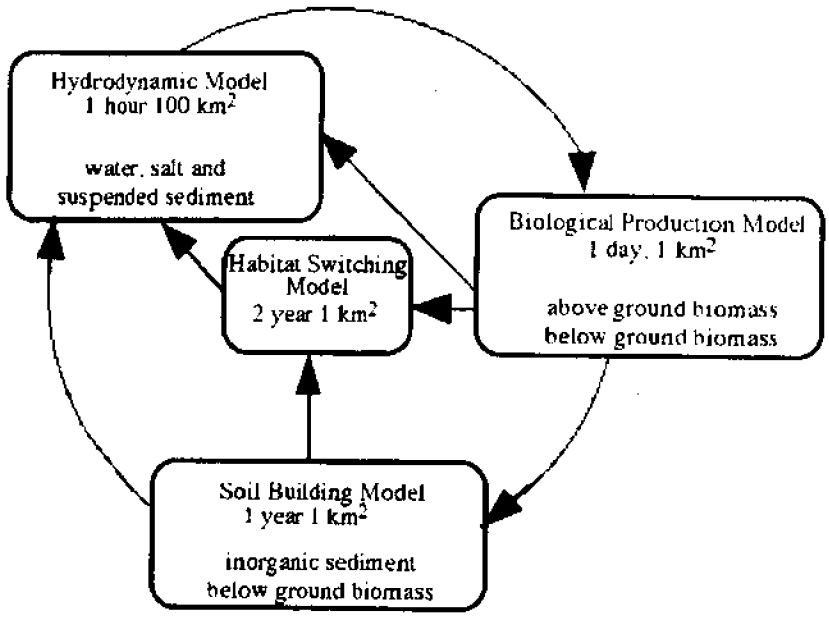

Fig. 2. Flow of calculations for the landscape program modules indicating time and spatial scales.

Hydrodynamics

The hydrodynamic module uses the diffusion wave approximation for shallow water to calculate water movement and sediment transport (Singh and Aravamuthan 1995). This approximation requires that local acceleration, uniform flow, and Coriolis force to be considered negligible. This simplification is necessary because standard hydrodynamic equations require smaller time steps than is practical for long term predictions (Cheng et al. 1993). The effect of friction is accounted for by a Manning coefficient that is the 100 km² average of all 1 km² habitat dependent Manning coefficients included in that 100 km² cell. Inputs include rainfall, pumping outfall, and rivers. Outputs are limited to evaporation and tidal boundary exchanges. Infiltration into groundwater is assumed negligible.

Initial spatial inputs include a 100 km² land elevation map and a suspended sediments field. Resultant hourly values for 100 km² water elevation, water velocity, salinity and suspended sediment distributions are linearly interpolated to yield 1 km² values. Interpolated water and land elevations are used to calculate water depth. Suspended sediment is evaluated hourly, and according to the water velocity, is determined to deposit on the marsh or resuspended. Deposition is calculated as a net daily value, and salinity and duration of flooding are averaged to daily values.

Productivity

The macrophyte module was viewed as a representation of any portion of a given landscape with homogenous characteristics. The biological module runs on a daily time-step, integrating net productivity for the macrophyte community throughout a 1 km² cell. Although structurally the same, the biological module uses different parameters and initial values for each habitat once is spatially distributed. Specifically, two biological components are modeled: non-photosynthetic carbon biomass (roots and stems) and photosynthetic carbon biomass (leaves). Gross production is a function of biomass, maximum gross production rate and a limitation function (Hopkinson et al. 1988). This limitation function includes responses to water level, salinity and temperature. The waterlogging function represents the different tolerances of each habitat (as determined by a representative specie) to flooding conditions (Nyman et al. 1993). Salinity stress is also

determined by habitat type (Pezeshki et al. 1987). The temperature response function varies seasonally. Respiration rates (proxy for metabolic stresses) adjust to these same factors (Burdick et al. 1989; Dai and Wiegert 1996; Hopkinson et al. 1978). Excess fixed carbon is translocated to the non-photosynthetic storage (Howes et al. 1985). The opposite process occurs if respiration losses offset photosynthetic intake.

Soil

The soil module includes three components: inorganic sediments, dead belowground biomass and total below ground organic sediments (dead biomass plus non-photosynthetic biomass from the biological module). Both inorganic and organic components were divided by a constant bulk density to calculate elevation.

Total sediment elevation was calculated by adding the heights of inorganic and organic storage and accounting for percentage pore space. Incorporation of aboveground litter to the belowground storage is assumed negligible (Nyman et al. 1993). The amount of inorganic sediments is conserved and a decomposition rate is applied to belowground organic sediments.

Subsidence accounts for 85-90% of the relative sea level rise (RSLR; estimated at 1.2 cm/yr) within Louisiana coastal marshes (Day and Templet 1989). While subsidence is not explicitly included within the soil module, decomposition of the organic stock partially simulates shallow subsidence. Deep subsidence of the Holocene layer is accounted for by increased rates of eustatic sea-level rise (Penland and Ramsey 1990).

Habitat Switcher

This module keeps track of habitat characteristics and environmental parameters for each cell throughout time. These environmental parameters are summarized and evaluated annually to determine if they are representative of another habitat type. The interaction of primary productivity and environmental conditions defines habitat type (Laurenroth et al. 1993). Once a threshold of change is reached, the model simulates habitat change.

The habitat switcher module has two components: a daily counter and a switcher. The daily counter queries biomass density, salinity and depth of standing water at each 1 km^2 cell. Based on these values, habitat type is determined according to classification criteria (Table 1), and the counter advances by one unit. At regular predetermined intervals, the habitat counters for each cell are submitted to the habitat switcher component. This algorithm evaluates if a habitat change should take place under several conditional rules. If more than half of the time open-water conditions existed then the cell type is assigned as open water (Nyman et al. 1993). If not, the vegetative habitat type with the highest count is assigned. When the habitat type changes, new productivity and respiration rates appropriate for that habitat type are applied in the biological module. The habitat switching decisions are made once every two years in this model version but any time interval could be defined.

Table 1. Habitat type definition by salinity (PSU) and biomass density (kg OM m^{-2}).

Habitat	Minima		Maxima	
	Salt	Density	Salt	Density
Fresh	0	0.92	< 4.5	4.6
Swamp	0	20.3	9.0	45.2
Brackish	4.5	0.4	< 12.5	2.2
Salt	12.5	1.2	40.0	6.0

Forcing Functions

Forcing functions for the landscape model were input as time series. The time series necessary to run both models from 1956 to 1992 were compiled from 15 stations for 9 parameters (wind speed and direction, inorganic suspended sediments, evaporation, rainfall, air temperature, salinity, tides, river discharge).

Daily tide stages were obtained from the National Ocean Service (NOS) at Bayou Rigard, for 1955-1979 and from East Point, Grand Isle from 1980-1988. Daily salinity values at Grand Terre Lab collected by Louisiana Wildlife and Fisheries were

used as the boundary condition for the Barataria basin. Boundary conditions for Terrebonne were set using the salinity distribution reported by Murray and Donley (1994).

Spatial Data Assembly

Habitat maps for the Barataria and Terrebonne basins were provided by USFWS in rasterized 625 m^2 georeferenced maps. These digital maps were derived from aerial photography for 1956 and 1978 and satellite imagery for 1988 and 1990, and aggregated to 1 km^2. In addition to open water and developed upland, each cell was classified as a wetland type characterized by a single dominant species. Salt marshes were characterized by *Spartina alterniflora*, brackish marshes by Spartina patens, fresh marshes include *Panicum hemitomon* and swamps by *Taxodium distichium* (Chabreck 1972; Conner et al. 1987; Tiner 1993; Visser et al. 1996).

CALIBRATION

Recognizing that a landscape model is a complex system with non-linear responses to different environmental stimuli, a calibration strategy was implemented to consider the effects of different scales and impacts of each module on model performance. This strategy allowed us to increase the model complexity as each module was first tested individually and then combined into the landscape model. The data set used for this calibration only included the parameters from 1978 to 1988.

Temporal Calibration

Resulting time series from each module were compared against available data at specific locations. The model was run repeatedly until these time series matched the field data. The hydrodynamic module was tested using the 1994 elevation map (Alawady and Al-Taha 1996), no winds and theoretical tides for 1987 and 1988. Hourly water levels were computed by the model at cells corresponding to the locations of the NOS data and compared for consistency and match. Manning coefficients were then adjusted for each of the habitat types to produce the best match. Salinity was calibrated by adjusting boundary conditions (i.e., salinity initial conditions and diffusion rates). Salinity results matched closely the available data (Murray and Donley 1994) in the lower portions of the basins.

The macrophyte production module was tested using specie specific parameters for each wetland dominant community type. Varying physiological parameters such as, flooding and salinity tolerance within reported ranges (Howes et al. 1986; Nyman et al. 1993; Pezeshki et al. 1987; Visser et al. 1996; Wiseman et al. 1990) produced seasonal trends of productivity and biomass (Conner and Day 1976; Kirby and Gosselink 1976).

Spatial Calibration

The model was run for the 1978 to 1987 decade using the USFWS maps to set initial conditions and final spatial comparison. The objectives were: (1) to simulated ecological processes across the landscape and (2) to verify that all the land loss processes were implicit in the landscape model. This second objective was stated to assure that although when the landscape model does not explicitly incorporate local processes (e.g., canals and spoilbanks), it incorporates whatever regional effects these impacts might have. This calibration method thus compensates for local effects by lowering the regional resilience of the habitats to land loss.

The landscape calibration required a match in habitat distribution. This implied consistency among the land use changes, land loss rate and habitat response to climatic variability. Previous habitat modeling efforts in southern Louisiana (Sklar and Costanza 1991) have utilized a goodness of fit parameter (Ft; Costanza 1989) to evaluate the model performance. This fit index compares two maps using a moving window technique and calculates the degree of accuracy between them in values ranging from 0 (no match) to 1.00 (perfect match). For the calibration purposes, the model was repeatedly run comparing resulting maps from 10-year simulations ending in 1988 to the USFWS 1988 map. This was done until the overall fit improved to 0.85 for both basins.

Initial conditions (salinity, elevation, and Manning coefficients), and biological parameters such as salinity tolerances for each habitat were manipulated during the calibration. The 1978 - 1988 calibrated simulations yielded a fit of 0.89 for Barataria and 0.85 for Terrebonne. There was also agreement for total wetland and water areas for the two watersheds (Ft = 0.96 for Barataria and Ft = 0.94 for Terrebonne). Table 2 presents the resulting fits and habitat areas for each of the runs, starting with the calibration, validation, and management option experiments that will be discussed later in the text.

Another calibration was done by comparing available land loss information. Annual land-loss estimates from 1955/56, 1978 and 1990 generated from the USFWS and USACOE (Dunbar et al. 1992) intervals are given in Table 3 along with those created by the model.

Validation

To validate the model, parameters settings for the base case (1978-1988) were used to simulate 1956-1988 conditions using the USFWS map as initial conditions. The model predicted land loss fluctuations for 1956-78 between 0 and 65 km^2 for Barataria and 85 km^2 for Terrebonne, similar to reported values of 73 km^2 (Gagliano et al. 1981). Habitat and goodness of fit results are presented in Table 2. Goodness of fit values for these simulations were lower than the calibration values. The decrease of accuracy in model predictions can be attributed to the different land loss trends (Table 3) through the 30-years tested.

MANAGEMENT SCENARIOS

Landscape models of this type are one of the few tools that can be used to predict the effects of complex spatial interactions and cumulative, long-term effects of global changes. Simulations, from 1988 to 2018, were performed for a series of management scenarios (Table 2). The first scenario, referred to as Normal Conditions (NC), simulated a future continuation of current trends. Later simulations are evaluated by comparing results against this NC scenario. The following questions can be answered using this comparative approach. What were the effects of the management alternatives? Was it positive or negative with regards to land loss? Was there any associated change in habitat distribution?

Normal Conditions Scenario

The NC scenario consisted of a 30-year simulation for each basin. To run simulations into the future, theoretical time series and boundary conditions needed to be decided. We repeated the original time series in reverse order, because climate tends to be cyclic (Thomson 1995). The forcing functions and boundary conditions are actual data for years 1955-1992 but when the year 1993 was simulated, the climate from 1991 was used, 1994 simulation used climate from 1990, and so on.

The resulting habitat distribution for Barataria (Fig. 3) converted 1,105 km^2 to open water during 1988 to 2018 (Table 2). The largest decline (498 km^2) was for brackish marsh, while only 5 km^2 of swamp were lost. The model identified large portions of the middle and lower brackish marsh that converted to open water, whereas the upper basin, dominated by swamp habitat, remained relatively unchanged.

The brackish marsh sites of Bayou Perot and Bayou L'Ours were subjected to time series analysis (Fig. 4), as these sites underwent a change from brackish marsh to open water. Figure 4a reveals that following 2005, the water level increased and remained high as plant biomass decreased until the habitat at Bayou L'Ours changed in the year 2017. At Bayou Perot (Fig. 4b), this habitat switch occurred in 2015 probably because of rising sea level and increased salinity.

In Terrebonne, 1,204 km^2 of land were converted to open water (of which 660 km^2 were fresh marsh) habitat (Fig. 5). This occurred mostly in a large area of fresh marsh at the northwestern portion of the basin, although some fragmentation of brackish and salt marshes also occurred in the southeast. These changes resulted on a water/land ratio increase from 0.62 to 1.51 by 2018.

Table 2. Summary of scenario results performed in each basin.

	Resulting Habitat Coverage (km^2)					Calibration Fit	
Scenario Name	Fresh Marsh	Swamp	Brackish Marsh	Salt Marsh	Open Water	Land/Water	Habitat
Terrebonne Basin USWFS map	1170	432	828	576	2106		
Base Case (calibration; 1977-87)	1100	516	865	551	2080	0.9433	0.8508
Base Case (validation; 1956-87)	847	657	596	842	2170	0.8664	0.7274
Normal Conditions (1988-2018)	510	428	499	365	3310		
Freshwater Addition	511	428	506	362	3305		0.9829
Hydraulic Restoration	552	429	466	365	3300		0.9829
Barataria Basin USWFS map	755	1022	734	460	2952		
Base Case (calibration; 1977-87)	723	1002	722	634	2854	0.9597	0.8932
Base Case (validation; 1956-87)	1191	795	577	288	3084	0.8648	0.7439
Normal Conditions (1988-2018)	396	1017	236	217	4057		
Freshwater Addition	521	1019	311	236	4057		0.9751
Stopping Saltwater Intrusion	303	1015	226	247	4132		0.9645

Note: Fit values for Base Case scenarios were computed against 1988 USWFS habitat map.
Fit values for management options were computed against the 2018 normal conditions habitat map.

Addition of Freshwater Scenario

This scenario simulated diversion of river water into the Barataria basin. The site for this freshwater diversion was Davis Pond, an area that encompasses 6.6 km^2 of east central Barataria. Inflow was simulated at 301.4 m^3/s (10,000 cfs) from December through March and 28.3 m^3/s (1,000 cfs) for the remainder of the year. A hydraulic connection already exists between Barataria and Terrebonne so the simulated freshwater addition was implemented into Terrebonne by increasing flow

Table 3. Annual loss rates (km^2) for the 1956-78 and 1978-88 periods in Terrebonne and Barataria Basins

Interval	Terrebonne $km^2 yr^{-1}$				Barataria $km^2 yr^{-1}$			
1931-1958	5				7			
1956-1978	25[a]	24.86	21.27	26.73	16[a]	20.09	21.55	25.50
1974-1983	20				22			
1978-1988	15[b]	31.6	33.10	33.89	19[b]	34.5	35.40	27.11
Source:	(Dunbar et al. 1992)	(Reed 1995)	USFWS	Model output	(Dunbar et al. 1992)	(Reed 1995)	USFWS	Model output

Note: [a]1958-74 and [b]1983-1990 intervals.

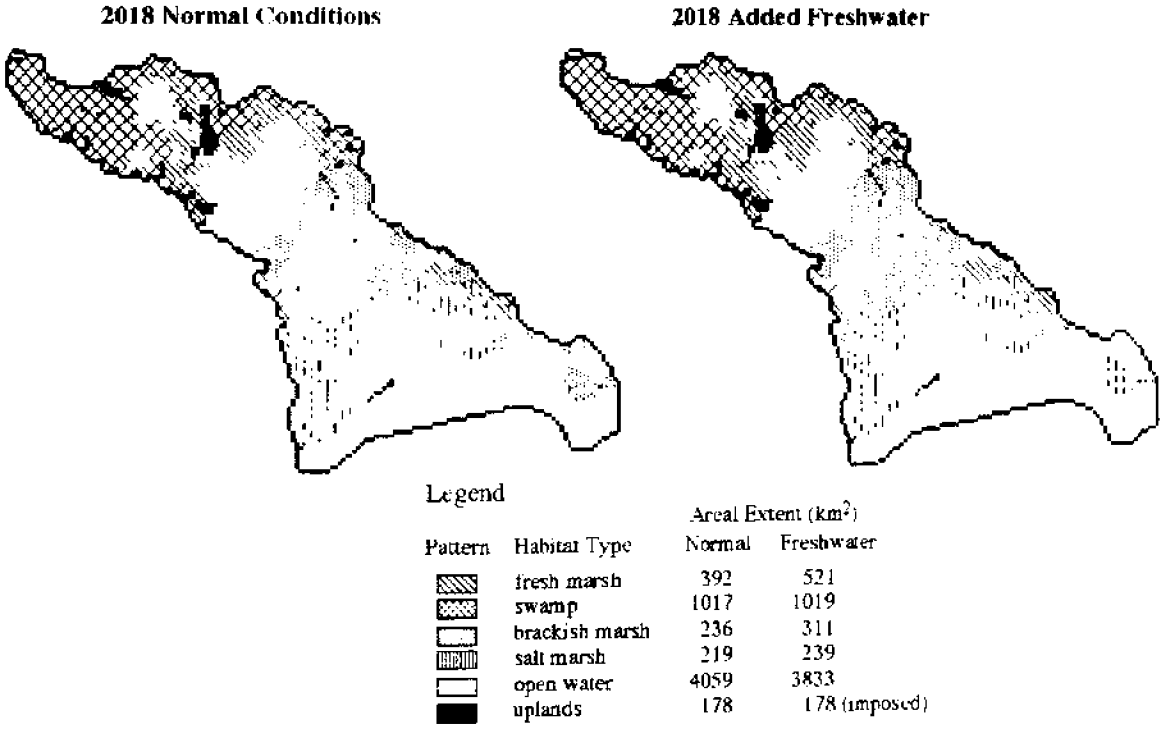

Fig. 3. Resulting habitat distribution of Barataria Added Freshwater scenario compared vs. Normal Conditions scenario.

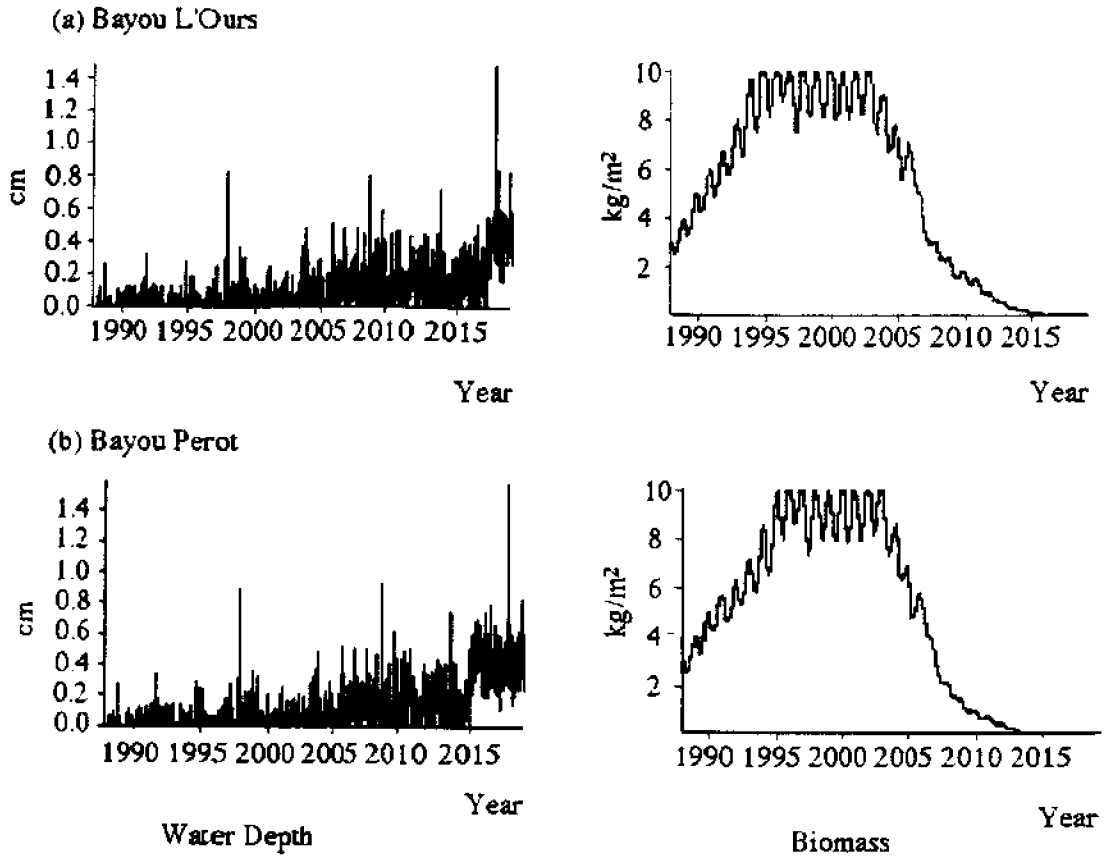

Fig. 4. Time series of water depth and photosynthetic biomass for (a) Bayou L'Ours and (b) Bayou Perot sample locations for Normal Conditions scenario.

rates there by 84.9 m^3/s (3,000 cfs) during peak discharge.

Results from this scenario showed that land loss in Barataria was reduced by 113 km^2 compared to NC (Fig. 3). The preservation of fresh marsh was largely responsible for this difference. Habitat distribution effects were minor in Terrebonne and resulted in almost no difference from the base case scenario (Table 2).

Restoration of Internal Hydraulics Scenario

The simulation of restored internal hydraulics was done only to the Terrebonne basin. The modifications implemented in this scenario in Terrebonne include: (1) Restoration of natural levees at 15 sites that had been destroyed and (2) Improvement of the internal water flow by redirecting flow from the west portion of the basin farther into the east. For this simulation, Manning coefficients were increased locally as a proxy for the modifications (Fig. 6a). We assumed that increasing water friction in the selected area would result in increased flooding and redirect flow towards the internal portions of the basin.

The resulting habitat distribution in Terrebonne showed about 42 km^2 more of fresh marsh and 33 km^2 less of brackish marsh compared to the NC (Fig. 5). This was due to a freshening of the midwest portion of the basin, and fresh marsh preservation (Fig. 6b). The scenario had negligible effects upon the distribution and amount of swamp and salt marsh habitats (Table 2).

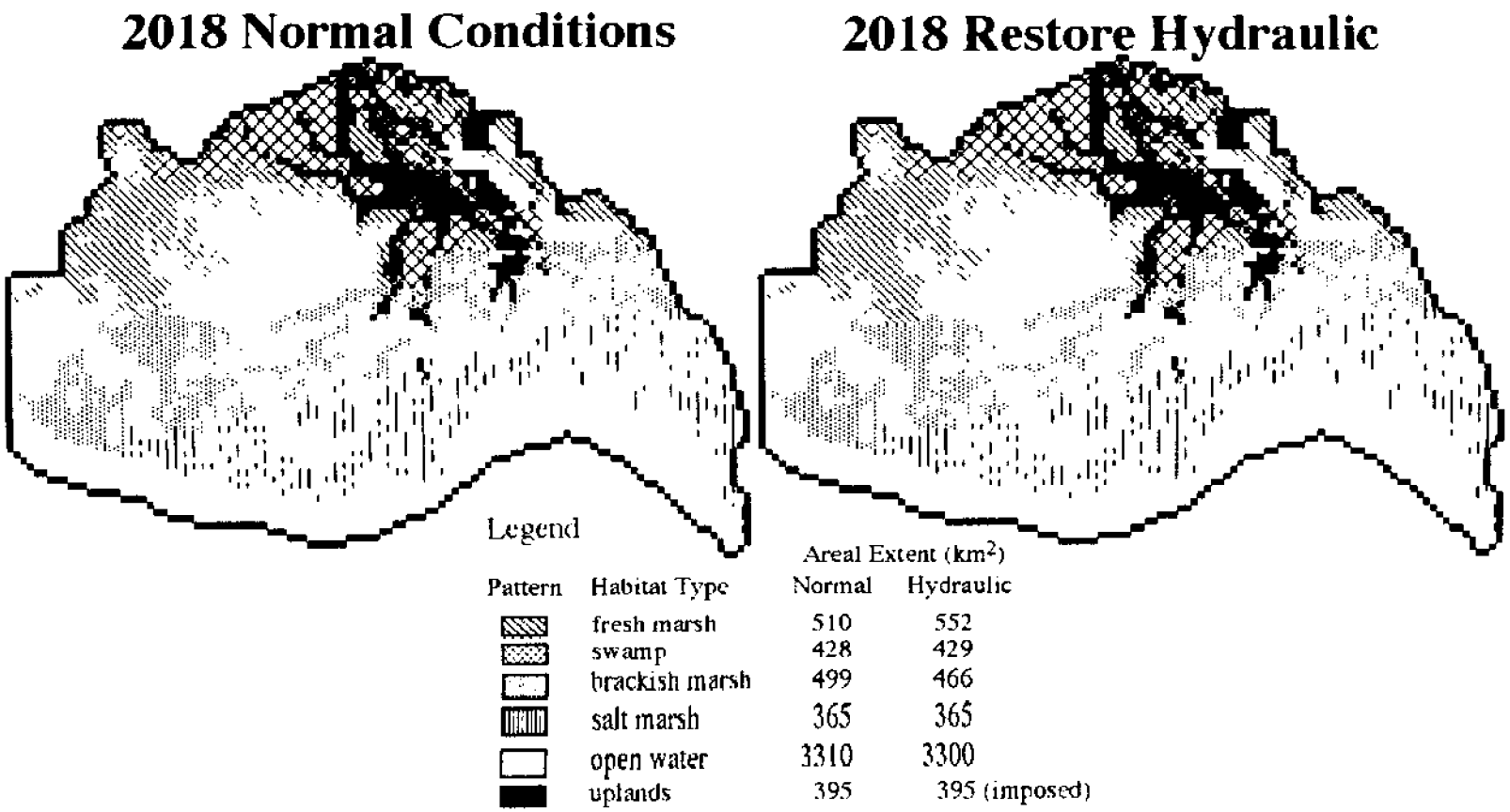

Fig. 5. Resulting habitat distribution of Terrebonne Restore Hydraulic scenario compared vs. Normal Conditions scenario

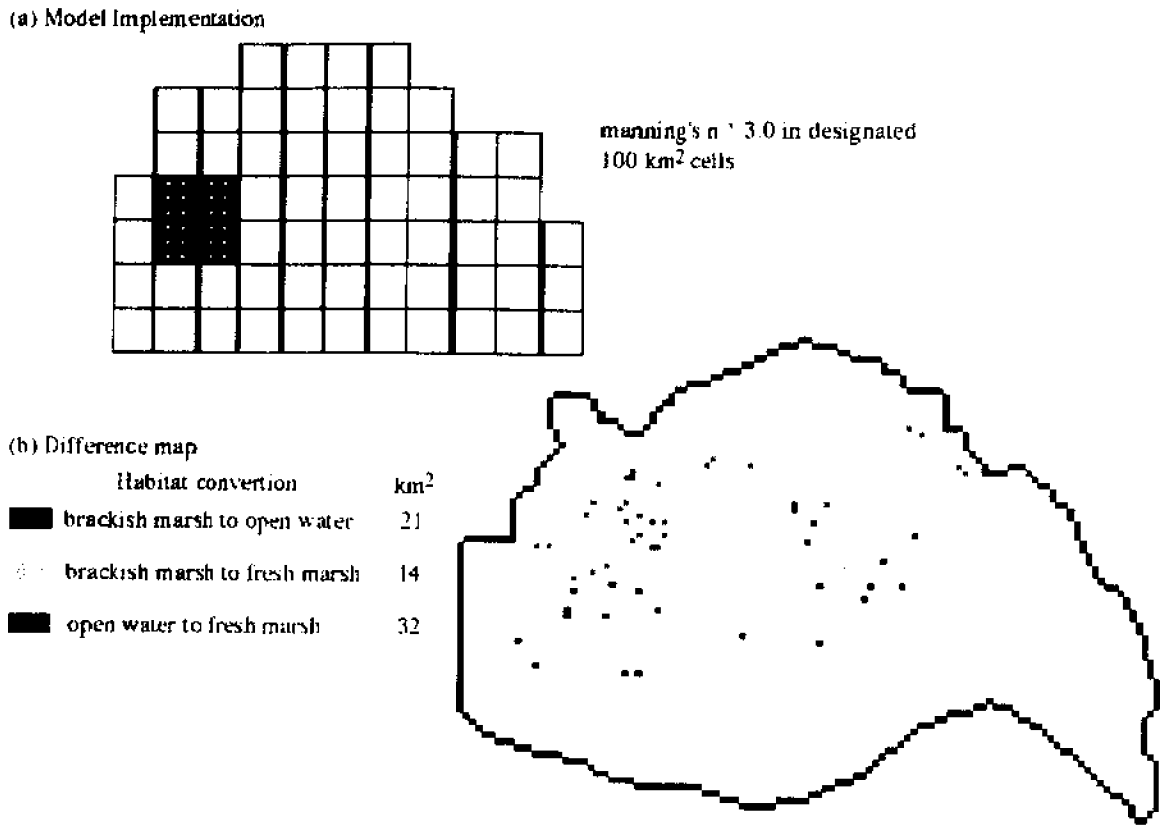

Fig. 6. (a) Manipulations imposed within model to perform Terrebonne Restore Hydraulic Conditions scenario. (b). Difference habitat map for Terrebonne Restore Hydraulic Conditions scenario vs. Normal Conditions.

Stopping Saltwater Intrusion Scenario

In this scenario, a barrier to saltwater intrusion into the upper basin was simulated for Barataria. This levee crosses the width of the Barataria basin with only one break in the center portion that allows flow from Bayou Perot and Bayou Rigoletts to Little Lake. The results showed an increase in open water (Table 2) compared to the NC scenario. About 110 km^2 of fresh marsh was lost, 10 km^2 of brackish marsh were lost for a total open water gain of 88 km^2 (Fig. 7). There was preservation of an additional 35 km^2 of salt marsh. The greatest amount of land loss took place in fresh marshes located in the northwest. This was due to increased flooding duration north of the simulated levee (Fig. 7).

DISCUSSION

Others have found that the principal regional factors driving long-term trends in land-loss and habitat change in coastal Louisiana are (1) sea level rise and subsidence, (2) changes in the introduction of freshwater and sediments from the Mississippi and Atchafalaya Rivers, and (3) modifications to internal hydrology (Baumann et al. 1986; Boumans and Day 1993; Coleman 1988; Day et al. 1997; Day and Templet 1989; Gagliano et al. 1981; Reed 1995; Salinas et al. 1986; Turner 1997). Our landscape models are driven by these same dominant regional processes and were, therefore, sensitive to factors that affect how land and water surfaces evolve interactively through time. This means that the models are less sensitive to local factors such as canal dredging, or natural ones like nutria eat outs or fires.

When only the western portion of the Terrebonne basin was modeled in the original CELSS study (Sklar et al. 1991), a Ft = 0.882 for the calibration run from 1956-1978 and Ft = 0.79 for the validation run of 1978-1983 were obtained.

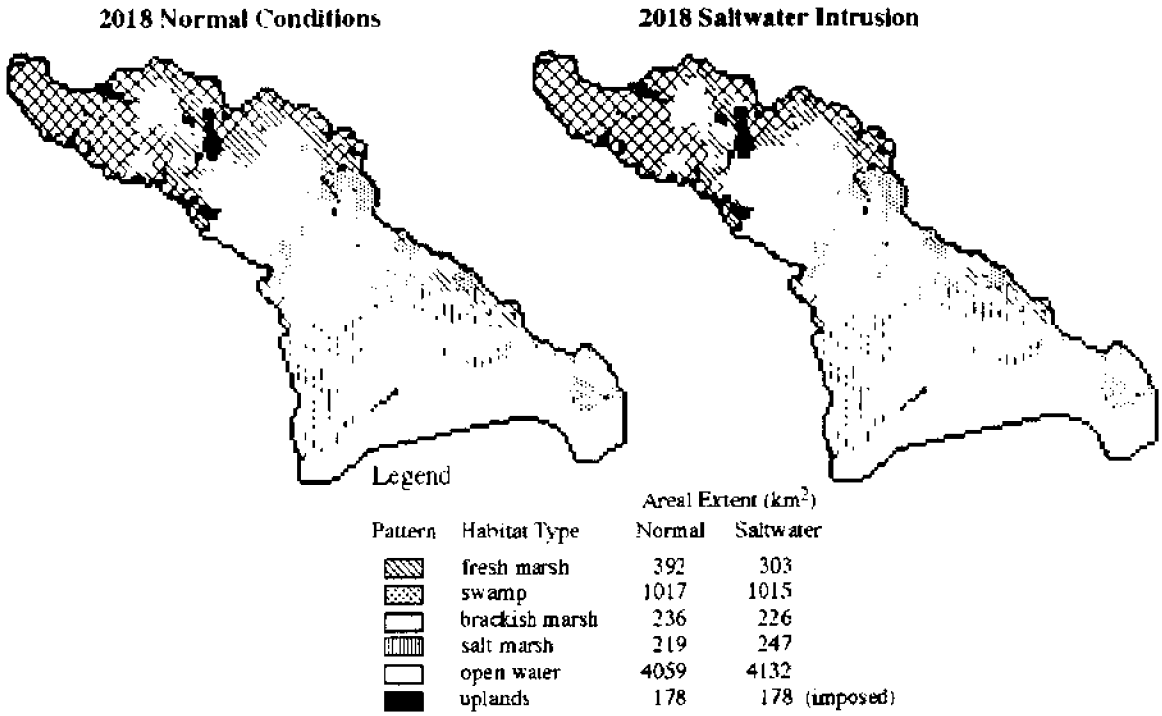

Fig. 7. Resulting habitat distribution of Barataria Saltwater Intrusion scenario compared vs. Normal Conditions scenario.

The results for both present models which included the area modeled by Costanza and his colleagues are similar (Table 2). These results lead us to believe that the present model satisfactorily represents the Barataria and Terrebonne systems at a large scale and provide with an adequate spatial and temporal resolution for the model management predictions.

The assumptions of the landscape models, particularly with respect to the spatial resolution of the water routing (100 km²), limited comparisons between adjacent areas, and made assessment of the long-term potential of small projects (less than 1 km²) virtually impossible. The landscape models should be used as reconnaissance tools to provide an initial estimate of the type and level of hydrologic change necessary to achieve a desired response. Plant and soil processes at less than 1 km² scale and hydrologic processes at less than 100 km² that have not been modeled may produce localized changes.

The biological module and habitat switching algorithms focused on those factors that directly or predictably influenced land elevation and habitat type. These mechanisms had their major impacts on the marsh communities, thus setting a trend of land loss in these areas. This was due in part to differences in initial elevation, salinity, and water levels between upper and lower basins.

CONCLUSIONS

Two regional models that combine hydrodynamic and biological processes at different time and space scales are presented. These mechanistic models included feedback among four different modules (water, soil, plant and habitat switching). A 10-year calibration showed that the processes modeled are sufficient to explain up to 85% of the habitat distribution and 95% of the land/water changes.

Scenario results demonstrated the importance of increasing water flow into both basins. As these areas are subject to limited freshwater inflows, the rate of land loss only increases.

The advantage of using these landscape models is that they allow one to evaluate natural processes with a regional perspective and to investigate cause and effect of the management options at any location.

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Salt Marsh Plant Colonization, Growth, and Dominance on Large Mudflats Created Using Dredged Sediments

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ABSTRACT: Large mudflats (up to ë200 ha) created from sediments dredged from the Calcasieu Ship Channel are being colonized by marsh halophytes. We compared colonization and growth by dominant marsh vegetation at edge and interior locations of created sites ranging in age from 6 mo to 14 yr and natural reference marshes. Unvegetated area declined rapidly with age of created marshes (interior locations: 99.7% in the 6 mo old marsh, 42.8% in the 3 yr old marsh, and 0.1% in the 14 yr old marsh). This change occured primarily as a result of rapid colonization and vegetative growth by smooth cordgrass, *Spartina alterniflora*, in all sites. The first year's colonization was mainly by rafting of mats of roots and rhizomes, but vertical growth of buried plants was also important in some marsh edge locations. Seedlings contributed to marsh grass recruitment in the second and third years in the two youngest sites. *Spartina alterniflora* dominated or co-dominated in our 1996 survey: a) 14.4% (edge) - 2.7% (interior) of site III created in 1996 and surveyed about 6 months after de-watering, b) 82.7% (edge) - 27.3% (interior) of site II created in 1993, and c) 100% (edge) - 46.8% (interior) of site I created in 1982, d) 100% (edge) - 92.2% (interior) of reference marsh 1, and e) 99% (edge) and 100% (interior) of reference marsh 2. Other marsh grass species dominated or co-dominated less area. Dominance by *S. patens* ranged from 0 - 15.8% (created marshes) and 0 - 5.8% (reference marshes) depending on location and site. Dominance by *Distichlis spicata* ranged from 0 - 60.3% (created marshes) and 0 - 2.0% (reference marshes) depending on location and site. The annual succulent *Salicornia bigelovii* exhibited thick growth between patches of grass in many portions of the interior of site II. *Spartina alterniflora* was shorter and had greater ramet densities in site I, the oldest created marsh, relative to the other marshes. Plant height and biomass was greatest in site II. There was no significant difference in plant height and biomass due to location within marshes (edge vs interior). The numbers of flowering stems of *S. alterniflora* were not different in site II and the natural references marsh 1. However, there was no flowering in site I (the 14 yr old created marsh).The cause of these differences among populations could be either environmental or genetic. *Spartina alterniflora* cover reduced below- canopy air temperatures during summer, but had no significant effect in other seasons. Sediment organic matter generally increased with the age of the created marsh.

Key words and phrases: salt marsh creation and restoration, marsh plants, dredge sediments, *Spartina alterniflora, Spartina patens, Distichlis spicata, Salicornia bigelovii*

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INTRODUCTION

Restoration of marsh structure and function is a critical component of the efforts to offset wetland losses in coastal Louisiana. One restoration method involves the creation of marsh habitat in open water areas by using sediments dredged for either that purpose or as a by-product of navigation channel maintenance. Marsh creation has sometimes included planting one or more species on the dredged sediments. However, in many instances, large bare mud flats are created under the assumptions that plants will colonize "naturally," that the ensuing mix of dominant species is determined mainly by elevation and hydrology, and that these "marshes" will have all or most ecological functions typically associated with marshes of that group of species. These assumptions have not been fully tested, especially in large area sites.

Environmental gradients, such as tidal inundation, salinity, and soil anoxia, have been shown to affect tidal marsh plant growth and survival (Ponnamperuma 1972, Mendelssohn et al. 1981, King et al. 1982, Nixon 1982, Mitsch and Gosselink 1986, and many others). Interspecific competition between dominant salt marsh grasses along the gradients establishes the typical zonation patterns noted in high and low intertidal marsh settings (Bertness and Ellison 1987, Bertness 1991). In low marsh, flooded soils are often anoxic and the ability of *Spartina alterniflora* Loisel to oxygenate its roots and rhizophere is a key factor allowing this species to become dominant (Gleason 1980, Mendelssohn et al. 1981). High marsh soils are sometimes more oxygenated and consequently are often less stressful in terms of this physical factor. *Spartina alterniflora* can colonize and grow under high marsh conditions. However, it will be excluded when *Spartina patens* (Ait.) Muhl., a strong competitor for space, is also in the site (Bertness 1991).

The southwestern portion of the Calcasieu estuary near West Cove is strongly influenced by oceanic water entering through the Calcasieu Ship Channel. Consequently, marshes on the east side of highway 27, where our study took place, are dominated by halophytes such as the marsh grasses noted above. Studies of colonization, growth, and species interactions are required to assess the rate of marsh development and patterns of succession. We studied the colonization and vegetative growth of the dominant grass species *Spartina alterniflora*. We also characterized the percent cover of all abundant grass species (*S. alterniflora*, *S. patens*, and *Distichlis spicata* (L.) Greene) in 3 created marshes of different ages and 2 reference marshes in order to assess patterns of succession and dominance. This effort was part of a larger collaborative suite of studies that also included analyses of the genetic diversity of *S. alterniflora* and the levels of metal and organic contaminants in sediment and biota.

METHODS

Site Descriptions

Three large flats (>40-200 ha) of dredged sediments were created by the U.S. Army Corps of Engineers in 1982, 1993, and 1996 at the Sabine National Wildlife Refuge in southwestern Louisiana during maintenance dredging of the Calcasieu Ship Channel. All sites used in this study were on the east side of state highway 27, south of the city limits of Hackberry, LA and north of the refuge headquarters. The flats were created by pumping a slurry of dredged sediments and water into areas that had historically been marsh, but which were open water with occasional marsh islands at the time of restoration. All of the created marsh sites have containment berms on one or two sides and are contiguous with natural salt marsh or open water on other sides. Marshes were open to flooding from either the Hog Island Gully canal (site II and reference marshes 1 and 2), old Hog Island Gully and West Cove (site III), or the Calcasieu Ship Channel and Long Bayou (site I). Our study was initiated prior to construction of site III in 1996. Thus, this site was sampled for certain variables (percent cover, clone size, and sediment organic content) but was not included in the quantitative analyses of growth, biomass, etc. Natural marshes adjacent to the flats were used as reference sites. Reference marsh 1 was located on the north side of the Hog Island Gully canal to the west of the site II created marsh. This reference marsh was sampled

for cover, stem density, height, and flowering whenever created marshes I and II were sampled. We established a second reference marsh following creation of the 1996 site. This reference marsh 2 was located on the south side of the Hog Island Gully canal immediately to the west of site III.

Quantitative analyses of biomass and plant size and environmental conditions

In April 1996 we established stations for study at two locations ("edge" near open water or adjacent marsh and "interior" further from open water) in two created marshes (site I created in 1982 and site II created in 1993) and a reference marsh (reference 1, adjacent to site II). In site I and reference 1, edge locations were within 2 m of open water and interior plots were 50 m inland. Using 50 m from open water for interior plots sufficed because these marshes were only approximately 150 - 200 m wide and thus interior plots actually represented marsh interior. However, this was not the case in the much-larger site II. This site was continuous with existing natural marsh on its western margin. As "edge" locations in this large (200 ha) site, we picked plots of vegetation that were clearly growing on dredged sediments and were closest to the bordering natural marsh. For the "interior" plots, we chose a location nearer marsh center about 400 m from the edge which appeared representative of the typical interior of the site. Thus, for all study marshes, interior locations were situated between about 1/4 and 1/2 of the distance from open water to the most distant point across the marsh.

Three permanent plots were established and marked with PVC poles at each location. In site II, where *S. alterniflora* often existed as spatially separated circular patches, plots were located in separate patches of the grass. The patch radius at study initiation in April 1996 ranged from 3.06 to 7.49 m. Continuous swards of grass existed rather than separate small patches, in site I and reference marshes. In these sites, plots at a location (edge or interior) were situated 50 m apart.

We sampled each plot quarterly over 1 year. Ten quadrats (25 x 25 cm) were located by haphazard toss at each sampling. In each quadrat we recorded numbers of stems, length of every stem as a measure of height, numbers of flowering stems, and collected all above ground material for biomass determination (dry mass after drying at 70° C). Also, we recorded water depth and salinity at each plot. Air temperature was measured above and below the *S. alterniflora* canopy.

Biomass, height, flowering and stem density data were analyzed by repeated-measures or two-way ANOVA as appropriate using Systat. Repeated-measures analysis was used for temporal data when the assumptions of the test were met. Differencs in air temperature above and below canopy was tested using a paired t-test.

Percent Cover Determinations

We ran a minimum of 3 replicate transects (4 m wide) across each marsh in all created and reference marshes. We measured the distances of open ground (or water) and all vegetated patches falling on the transect line. The dominant species, or co-dominant species mixture (assessed by visual estimation as approximately equal spatial coverage), was recorded for each section of vegetation. The percent cover was calculated from the means of the replicate transects in each marsh.

Sediment Organic Matter

As one metric of long-term effects of vegetation on soil structure, we conducted a one-time sampling of the study plots for sediment organic content. Litter was removed from surface (upper 5 cm) soil samples and the sediment was dried to constant temperature at 70° C. Dry samples were usually very hard and were crushed and ground prior to determining organic content via loss on ignition (500° C for 4 hours).

RESULTS

Environmental Conditions

Elevations in the marshes range from 0 to +0.7 m with 0 defined as elevation at marsh-open water interface (C. Norman, unpublished data). In site II, elevations at the edge plots ranged from 0.15 to 0.24

m and from 0.31 to 0.46 m at interior plots. Because of logistical constraints, elevations were not gathered at individual study plots in the other marshes. While the sites were relatively flat, there were topographic highs in sites II and III which were probably associated with the placement of discharge pipe openings during sediment deposition.

Water depths varied seasonally, among marshes, and between locations within marshes. The driest time was the first collection (spring 1996) with overall mean depths to ground water (below surface values are indicated by negative sign) in created sites of -66.3 ± 2.62 cm Site II and -18.8 ± 2.11 cm in Site I. In reference marsh 1 there was standing water at this collection (0.4 ± 2.20 cm). In the winter of 1997, all marshes had standing saline water (Site II 5.6 ± 1.91 cm, Site I 6.0 ± 1.89 cm, Reference marsh 1 8.9 ± 1.90 cm, and Reference marsh 2 18.8 ± 2.10 cm). Water elevation data were not collected in the youngest restored marsh (site III) because it was not de-watered enough to allow walking access for sampling until late into the study.

Historic data provided by the Sabine National Wildlife Refuge show considerable within-and-among-year fluctuations in salinities in the Hog Island Gully Canal adjacent to restored sites II and III and both reference marshes (Fig. 1). From 1991 - 1997, the mean ± 1 S.E. salinity was 10.7 ± 0.51 mg l^{-1} n = 142, and the range was 0.7 to 23.4 mg l^{-1}. The salinities of ground water and surface waters in our study ranged from ≈0 to 32.0 ± 8.0 mg l^{-1} (Table 1), and generally tracked patterns of change of salinities in the canal. Salinity values in site I (the 14 yr old created marsh) were typically a little higher than in other sites. This site is nearest the Calcasieu Ship Channel.

At higher elevations in two of the created marshes (sites II and III) surface sediments, especially in unvegetated areas, were often hard, dry, and cracked when there was no standing water or recent rainfall. Under these conditions, surface sediment salinities were substantially higher than those of the ground water (often 20-60 cm below sediment surface). Sediment salinities from site II,

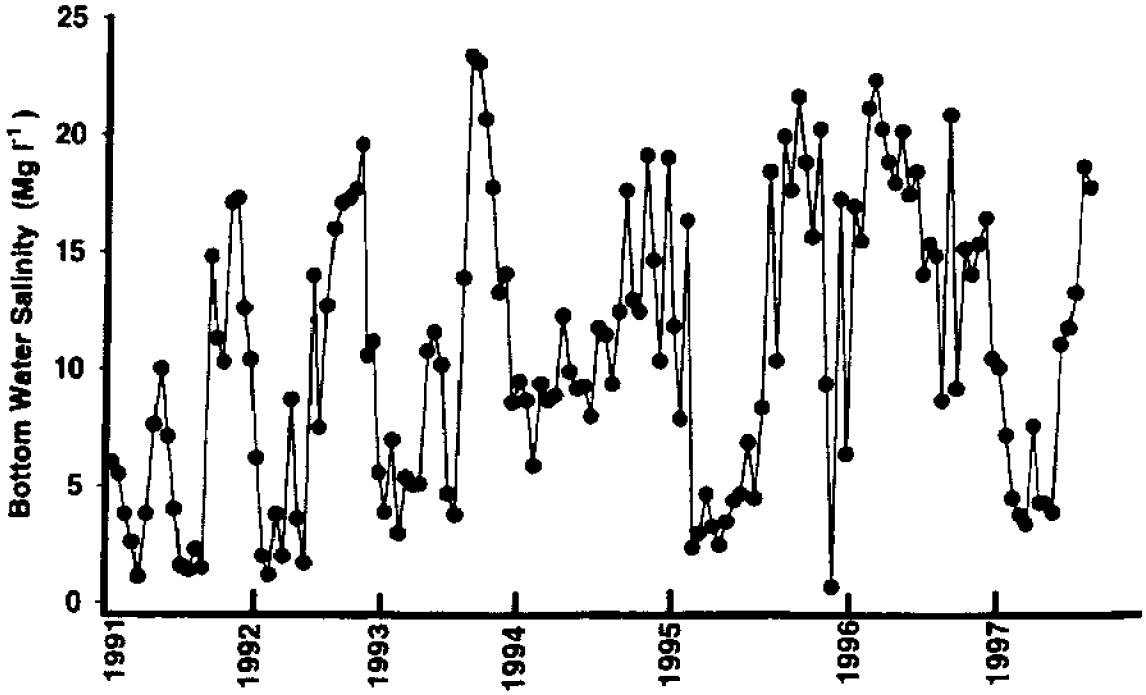

Fig. 1. Historic bottom salinities from 1991 - 1997 in the Hog Island Gully Canal are shown. On the x-axis, the year label is placed at the start of a particular year. Data collection varied but points were gathered approximately bimonthly. Data were provided by R. Walters of the Sabine National Wildlife Refuge (U.S. Fish and Wildlife Service).

Table 1. Quarterly measures of mean (± 1 S.E.) salinities in mg l^{-1} of standing or ground water are shown for each study site (n=6). ND indicates data not collected. Zero values indicate fresh water within the limits of measurement.

	Mean Salinity Mg l^{-1} (1 S.E.)		
	Site II (3 yr old)	Site I (14 yr old)	Reference Marsh 1
Apr 96	22.0 (4.0)	32.0 (8.0)	ND
Jul 96	21.7 (2.5)	20.0 (0.0)	16.3 (0.4)
Nov 96	19.3 (3.1)	21.8 (1.8)	17.2 (1.3)
Mar 97	2.2 (1.1)	0	0

measured in July 1996 after mixing equal volumes of distilled water and dry sediment and allowing 5 hr to come to equilibrium, were 50.2 ± 4.1 Mg l^{-1} (n = 5). Sediment from reference sites, site I, and near-open-water areas in site II that was treated in the same manner, had much lower salinities that were similar to those of ground water.

The % organic matter differed among sites (one-way ANOVA, P = 0.0005, R^2 = 0.876). Values from site I (created 1982) were not significantly different from the reference 1 marsh (12.1 and 9.5% respectively). The two youngest marshes (sites II and III) had less sediment organic matter (4.7 - 7.3%) than the reference marsh. Reference marsh 2 had the highest sediment organic content (19.3%).

Air temperatures above and below *S. alterniflora* canopy were significantly different only during the summer (t-test, P = 0.0005). The overall mean temperature was 3.9° C cooler canopy in summer.

Field Observations of Initial Colonization

Field observations of site III, which was created and became de-watered during our investigations, indicate that grass colonization occurred within a few months of sediment de-watering and as a result of two main mechanisms: 1) in the low intertidal zone, where the layer of dredged sediments was not thick, buried *S. alterniflora* colonized by vegetative growth and 2) over much of the site numerous small (generally < 0.5 m^2) patches of *S. alterniflora* with intact roots and rhizomes rafted in with tides and storms. *Spartina patens* colonized by rafting and *D. spicata* apparently by seed over those early months following site de-watering. However, relative to *S. alterniflora*, these latter two species did not colonize extensively in terms of numbers of patches and total area covered.

Percent Cover

The unvegetated area in created marshes decreased with marsh age and increased with distance from open water (Table 2). Data from sites II and III (created in 1993 and 1996, respectively) show that colonization was more rapid near the edge of the marsh compared to the interior of the marsh. Field observations in site III suggest that a substantial proportion of this colonization near the edge was from vertical growth of buried *S. alterniflora*. Both reference marshes had nearly 100% vegetative cover (Table 2). *Spartina alterniflora* was the dominant or co-dominant species in all sites (Table 2). At edge locations (within 2 m of open water) in created sites, the area covered by *S. alterniflora* alone or as a co-dominant increased with age of the site from 14.4% (site III, created 1996), to 82.7% (site II, created 1993), and 100% (site I, created 1982). At interior locations (> 50 m from marsh edge) in the created sites, area with *S. alterniflora* as sole-or-co-dominant also increased with site age from 2.7% (site III, created 1996), to 27.3% (site II created 1993), and 46.8% (site I, created 1982) . Edge locations of the 2 reference marshes had 99 - 100% cover by *S. alterniflora* (Table 2). Dominance by *S. alterniflora* in interior locations was typically less than at marsh edge, but showed the same increasing trend with increasing age of created marsh: 2.7% site III, 27.3% site II, and 46.8% site I. Reference marshes had 92.2 - 100% dominace by *S. alterniflora* at interior locations.

In interior locations of the oldest created marsh (site I) the area dominated or co-dominated by the high marsh grass *D. spicata* was about the same as

Table 2. Percent cover by the major habitat types in created and natural reference marshes. Values are means of n=3 transects per location in each marsh and as such may not total to 100%. Mixed species groupings indicate numerical dominance by the species included. ** indicates open water at high tide in this marsh. Note that *Salicornia bigelovii* is an annual and in the winter, this portion of the habitat is also bare ground (with standing dead plants). Mixed dominance was recorded when species covered approximately equal proportions of a patch. Ages of created sites at time of sampling was site III (0.5 yr), site II (3.0 yr), and site I (14 yr). In each marsh, "edge" denotes the area from water's edge to 50 m inland; and, "interior" denotes from that point on to the approximate center of the marsh.

	Created Marshes						Reference Marshes			
	Site III		Site II		Site I		Ref 1		Ref 2	
	edge	interior	edge	interior	edge	interior	edge	interior	edge	interior
Bare Ground	85.1	97.0	2.4	42.8	0	0	0	0	0.1**	0
Salicornia bigelovii	0	0	2.5	27.5	0	0	0	0	0	0
Spartina alterniflora	14.4	2.4	68.3	27.3	92.0	22.0	100	92.2	71.0	98.4
Spartina patens	0.1	0.2	4.0	1.7	0	12.0	0	5.8	0	0
Distichlis spicata	0.3	0.4	8.6	0.7	0	28.8	0	2.0	0	0
Iva frutescens	0	0	0	0	0	0.2	0	0	0	0
S. altern.+S. patens	0	0	10.5	0	0	0	0	0	9.2	0
S. altern.+D. spicata	0	0.3	3.9	0	8.0	9.5	0	0	18.8	2.6
S. altern.+B. frutescens	0	0	0	0	0	1.5	0	0	0	0
S. patens+Iva frutescens	0	0	0	0	0	2.5	0	0	0	0
D. spicata+I. frutescens	0	0	0	0	0	7.6	0	0	0	0
D. spicata+B. frutescens	0	0	0	0	0	0.6	0	0	0	0
S. patens+I. fru.+B. fru.	0	0	0	0	0	1.3	0	0	0	0
S. alt.+ D. spic.+I. fru. +B. fru.	0	0	0	0	0	13.8	0	0	0	0

that dominated by *S. alterniflora* (Table 2). This species covered much less area in other created marshes, 0 - 12.5% depending on marsh and location within marsh and 0 - 18.8% in reference marshes (Table 2). In created site I, where it was most abundant, the area dominated by the third most abundant grass species, *S. patens*, was about 3x less than the greatest degree of cover by *D. spicata*. In reference marshes *S. patens* dominated 0 - 9.2% of site area (Table 2). Patches dominated by mixtures of species increased in frequency of occurrence and total site area covered with increasing age of created marsh, but were less prevalent in both reference marshes (Table 2).

The annual *Salicornia bigelovii* dominated ungrassed areas and displayed very robust growth during spring and summer in the interior of site II (Table 2). *Salicornia bigelovii* individuals growing within grassed patches were much smaller and occurred in low densities (field observations). It was possible to find occasional *S. bigelovii* in the interior of the older site I (created 1982) mixed in with the grasses but plants did not attain the high density and large stature seen in site II. In site III (created 1996) there was very little colonization by *S. bigelovii* the first year (1996-1997 data) and no part of the site had any degree of "dominance" in terms of cover by this species (Table 2). However, our

field observations the following year (summer of 1998) indicate substantial cover by *S. bigelovii* in parts of site III resulting from extensive colonization by seeds/seedlings in obvious natural low elevation drift zones where seeds came in on the tide and in apparent seedling shadows around the few individuals that did colonize the first year. Further field observations in the spring of 1999 indicate a very high density of *S. bigelovii* now in site III. Thus, the pattern of extensive cover of non-grassed portions of interior areas by *S. bigelovii* recorded in site II (created 1993) is being repeated in site III (created 1996).

Densities of *S. alterniflora*

Spartina alterniflora stem densities were 1.5 - 2 x greater in the oldest created marsh (site I), than in site II, or in the reference marsh 1 (Fig. 2). Repeated-measures ANOVA showed that differences among marshes was significant (P = 0.0005). There were no significant main effect differences between edge and interior locations within marshes (P=0.693), although the significant interaction (P = 0.022) indicated that densities varied differently among locations in the different marshes (Fig. 2).

Stem densities also varied significantly seasonally (P = 0.0005). Both created sites I and II showed continual increases in density over the course of one year, while densities in reference marsh 1 increased, and then decreased, at both edge and interior locations. This resulted in a significant time x site interaction (P = 0.0005).

Heights of *S. alterniflora*

Stem length (height) was significantly different (P = 0.0005) among marshes (Fig. 3). Site I, the oldest created site (created 1982), was dominated by short-form *S. alterniflora*, while site II (created 1993) and reference marsh 1 were populated by much taller *S. alterniflora*. There were no significant differences (P = 0.330) in height at edge and interior locations within marshes (Fig. 3). Height varied significantly over season during the study year (P = 0.0005) and differences among marshes in the pattern of change over time is

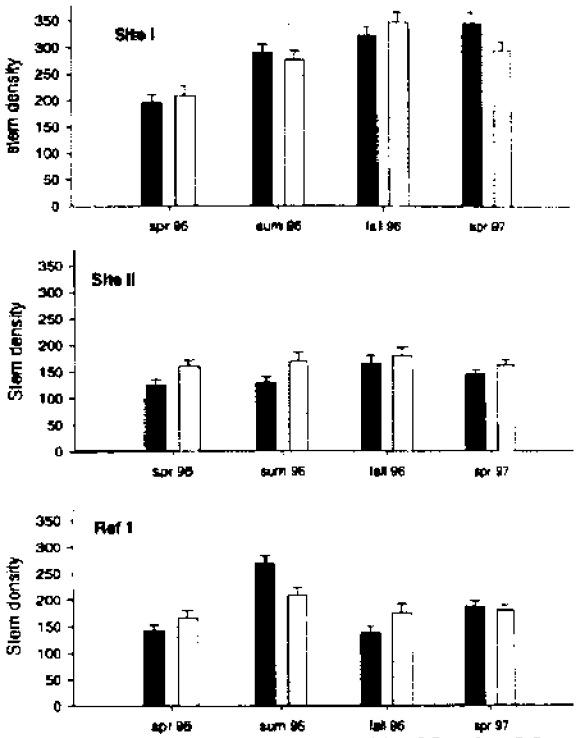

Figure 2. Ramet densities (stems/[25 x 25 cm]) of *S. alterniflora* are presented for restored marshes Sites I and II and the natural reference marsh by marsh edge and marsh interior locations. The x-axis is sampling times (Apr. 96 - Mar. 97). Values are means ± 1 standard error.

indicated by a time x site interaction (P = 0.0005). The mean heights in site I were always less than 40 cm and did not vary substantially over the year (Fig. 3). Heights in reference marsh 1 ranged from about 60 - 80 cm and both edge and interior locations showed general decreases in height over the year. Heights in site II ranged from about 55 - 100 cm and fluctuated dramatically over the year (Fig. 3). The tallest plants were at the interior location of site II and were situated far from open water. The mean values exceeded 100 cm and some individual stems along the outer portion of circular clones of the grass were 2 m long (unpublished data).

Above Ground Biomass of *S. alterniflora*

Above ground biomass is generally dependent on both stem density and height. In our study sites, differences in plant height tended to determine the differences in biomass. The above ground biomass

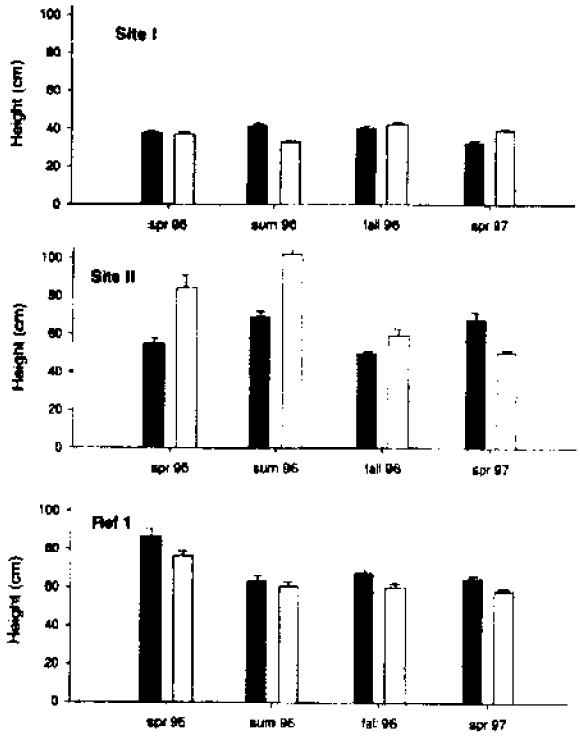

Fig. 3. Mean heights ±1 S.E. of *Spartina alterniflora* stems for each study marsh and location within marsh. Solid bars are edge and open bars interior locations in each marsh.

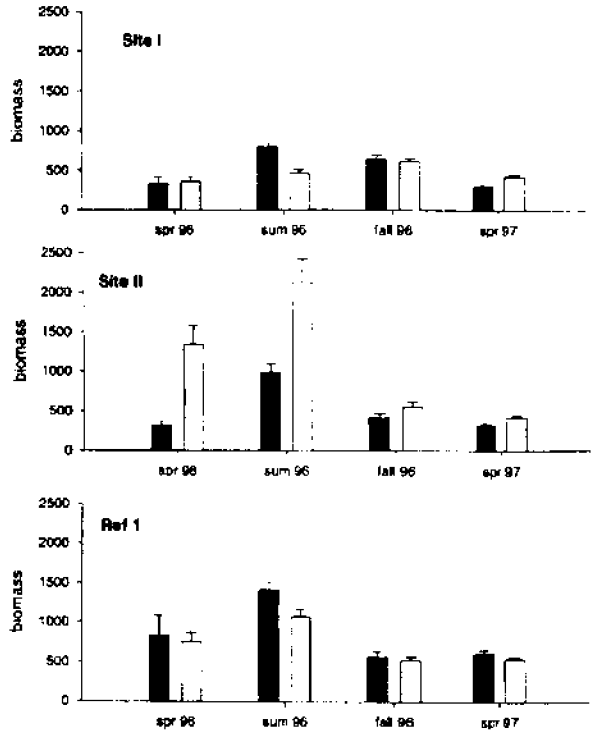

Fig. 4. Mean ±1S.E. aboveground biomass of *Spartina alterniflora* stems for each study marsh and location within marsh. Solid bars are edge and open bars interior locations in each marsh.

was lowest in site I, where the shorter growth form of *S. alterniflora* was prevalent (Fig. 4). The greatest biomass occurred in the interior location of site II, far from marsh edge, where robust clonal growth resulted in very long stems and leaves. Biomass at interior locations in site II was >2x that at edge locations during the spring and summer 1996 collections (Fig. 4). Biomass peaked in the summer at most sites (Fig 4). In July, when greatest biomass occurred in all but one marsh location, two-way anova indicated a significant difference among marshes ($P = 0.0005$).

Flowering Stems

Flowering occurred in late summer in site II and the reference marsh (Fig. 5). Seeds were evident at both sites by fall and seedlings noted in site II the following spring. No flowering occurred in site I during the study. There was no difference in density of flowering stems between site II and reference marsh 1, but both of these marshes had significantly more flowers than site I because there was no flowering in that site (two-way ANOVA, site main effect $P = 0.001$). There was no difference in flowering density between inner vs. edge locations within a marsh ($P = 0.302$) nor a significant marsh x location interaction ($P = 0.459$).

DISCUSSION

Colonization, Dominance, and Succession

Salt or brackish marshes can develop relatively rapidly on large dredge material flats. *Spartina alterniflora* dominated all study marshes and our observations suggest that the first year's colonization was by two modes. When the restored sites were first created and still covered with water, numerous small floating segments of eroding natural marsh with intact roots and rhizomes functioned as dispersing propagules and became established after floating into the sites (field observations in the 1996 created marsh site III). Subsequent vegetative

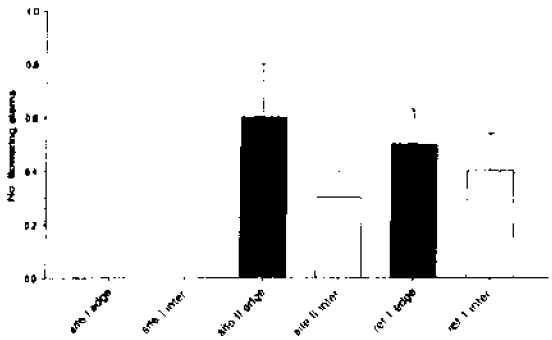

Fig. 5. Mean (1 S.E.) number of flowering stems are shown for the early Fall 1996 collection. Values are depicted by different marsh (site I, site II, and ref I) and location within marsh. Solid bars are edge locations and open bars are marsh interior locations. There was no flowering in site I.

growth resulted in a high percent cover by this species in the marshes created in 1982 (site I) and 1993 (site II). For the interior portions of the created sites, this mechanism probably accounted for most of the first year colonization by *S. alterniflora*. Deposited sediment covered portions of the existing marsh along site edges that were continuous with existing natural marsh and numerous patches of *S. alterniflora* occurred via vertical growth from buried roots and rhizomes (site III, field observations).

Sexual reproduction may have contributed to colonization of the site in later years. We observed a large seed set by *S. alterniflora* in the fall of 1996 in site II (created in 1993) and a number of seedlings the following spring. It is possible that some patches of this species also became established through seedlings the previous year since this site was 3.5 years old when we first made field observations. In site III, there were seedling "shadows" around many clonal patches in the spring of 1998 about 2 years after site creation (field observations).

Rapid vegetative growth of *S. alterniflora* resulted in its dominance in all sites regardless of the elevation and distance to marsh edge (Table 2). *Spartina alterniflora* was dominant or co-dominant over 2.7% (interior location) and 14.4% (edge location) in site III when it surveyed at 6 months after site creation. *Spartina alterniflora* in site II (surveyed 3 years after site creation) was the single dominant or co-dominant over 27.3 % (interior location) and 82.7% (edge location) of the marsh (Table 2).This species dominated or co-dominated 46.8% of marsh area at interior locations, and occurred in near mono-specific stands at edge locations in site I (surveyed 14 years after site creation). In the two reference marshes, *S. alterniflora* was dominant or co-dominant over a minimum of 92.2% of site area regardless of nearness to open water (Table 2).

Despite dominance by *S. alterniflora*, the dry conditions for portions of the year in the interior portions of sites II and III appeared conducive for the establishment of high marsh species or transitional species. In site III (surveyed for percent cover when 0.5 years of age), occasional *Salicornia bigelovii* existed; and, in site II (at 3 years of age) there was a dense spring and summer (1996) cover by this succulent halophyte over much of the non-grassed areas in the interior of the developing marsh (Table 2). This species is an annual and its colonization was by seed. In the youngest site (site III, created 1996), *S. bigelovii* was absent in the summer of 1996 which is consistent with the fact that the site was not yet de-watered when seeds would have been establishing. In the spring-summer of 1997 we observed widely scattered individuals of *S. bigelovii* in site III. In 1998 and 1999 there has been increasing dominance by this species over most of the non-grassed area of this site (field observations). This species grew well in bare areas in sites II and III, but was sparse and much smaller when it occurred within grassed patches. We conjecture that the annual *S. bigelovii* leads a rather ephemeral existence in these created marshes, with its dominance increasingly limited over the years by spread of the perennial grasses.

Typical high marsh grasses (*S. patens* and *D. spicata*) only accounted for at most 14.5% and 8.6% area cover respectively in site II and <0.5% in site III. Where these species have colonized, the patches appear to be healthy and growing well (field observations). However, there has been little colonization. We hypothesize that this may be a result of a paucity of rafting propagules, or viable seed, or both. Because *S. alterniflora* generally dominates near water's edge in marshes in this

portion of the Sabine National Wildlife Refuge, this is the species most likely to break free and raft into a new location. The other grass species are further removed from the water and often *S. alterniflora* occurs between them and open water. This result would tend to reduce the frequency of ramets of these species breaking free, washing out of a marsh, and floating to new sites. Although we observed all 3 grass species flowering, we have no data on the abundance of viable seed. We have observed *S. alterniflora* seedlings in both sites II and III, but not seedlings of *D. spicata* and *S. patens*. If seed set or seedling survival is low, this too could limit colonization of these high marsh grasses. These observations suggest that further studies of seed set, viability, floatation, and colonization of the perennial grasses are needed to resolve questions regarding colonization by these species.

While the physical conditions of the interior of sites II and III and studies of competitive outcomes (Bertness 1991) suggest that *S. patens* and/or *D. spicata* should eventually come to be the dominate perrenial grasses in interior portions of sites II and III, the rapidity of colonization, degree of dominance, and robustness of growth of *S. alterniflora* and the slowness of colonization by *S. patens* and *D. spicata* in these locations indicates that this process may take many years. Interestingly, dominance by mixed-species assemblages increased with age of created marshes, and was lower in the presumably much older reference marshes (Table 2). The preponderance of patches dominated by mixtures of species in site I suggests that created marshes of intermediate age (this site was 14 years old when surveyed) may be fertile locations for future studies of species-species interactions.

Comparison of Marsh Structure and Function

The two created marshes (sites I and II) studied for such vegetation factors as size, growth, biomass and reproduction were both dominated by *S. alterniflora*. However, the growth forms of this species were very different in the two marshes. The older site I was dominated by shorter *S. alterniflora* with more densely packed ramets that did not flower during the study year. The younger site II was more similar to the reference marshes in that plants were significantly larger (height and biomass), lower in ramet density, and flowered during the study. Whether this is some function of sediment or other physical factors or is a difference in dominant genotype(s) is not yet known. We did observe, but did not quantify, more frequently water-logged soils and apparent reddish iron deposits in site I where short-form *S. alterniflora* dominated. Long term studies of these sediments, during years of compaction and/or other structural and chemical changes, and their effects on vegetation may be needed to shed light on this question. Also, transplant experiments among sites and locations could address questions of growth form.

Full assessment of marsh function involves measurements of such diverse items as sediment biogeochemistry and nutrients, productivity, food webs, population biology and genetic viability, and succession patterns. Such an undertaking was not within the scope of this study. However, in addition to biological data on colonization and growth of the dominant plant species, we did evaluate a few physical parameters that are affected by plant community development. These variables were: air temperature above-and-below-canopy as an indicator of how plant cover can affect micro-environmental conditions; and, sediment organic matter as a metric of long-term changes partially caused by plant growth and decay.

The sediment organic matter content of the oldest restored marsh (site I) was not significantly different from that in the reference marsh 1, suggesting long-term effects of marsh grass growth and decomposition on soil structure. Sediment from the younger restored sites (sites II and III) had about 2/3 as much organic material (6%) as in the reference marsh 1 (9.5%). Values for organic matter in surface sediments of the reference marsh 2 (19.3%) were similar to results from a marsh terracing study located <0.5 km to the south (LA Department of Natural Resources Coastal Restoration Division 1993).

Air temperature is one metric of effects of vegetation on surface microenvironmental conditions which could affect growing conditions for other plants or habitat quality for small animals.

Cover by *S. alterniflora* significantly reduced air temperatures just above ground (or water) surface during the summer, but had no measurable effect in the winter. Cooler conditions below grass canopy during the summer may reduce evaporation of soil moisture. This possibility is supported by our qualitative field observations that soil surfaces in grassed patches tended to be less "dry and cracked" during the summer compared to areas immediately outside the grass patch. Other direct or indirect effects of reduced below-canopy temperatures could occur on seed/seedling viability, surface soil salinities, invertebrate survival and foraging behavior, etc. In studies of restored marsh function, more emphasis should be placed on ascertaining the effects of dominant vegetative cover on the physical factors such as sediment organic matter and microclimate factors affecting conditions for other plant species and marsh fauna.

While our study has shown that marsh vegetation develops on these dredge sediment flats, questions remain about differences in *S. alterniflora* size and flowering among marshes, patterns of succession, effects of (and effects on) sediment parameters, long-term sediment compaction rates, faunal use, levels of export of plant production, and the genetic variability in the dominant species. In addition, further evaluation of the amount and value of marsh edge vs. interior in terms of productivity and animal habitat is needed for large areas such as the sites at Sabine National Wildlife Refuge. Further work is needed before declaring these marshes complete "successes" in terms of the restoration or creation of fully-functioning salt marsh habitat.

ACKNOWLEDGMENTS

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Low-Cost Wetland Restoration and Creation Projects for Coastal Louisiana

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ABSTRACT: Wetland restoration and creation efforts are subject to different economies of scale, including scales of ecological proportions. An inverse economy of scale apparently operates in the various small and large wetland restoration projects possible or implemented in coastal Louisiana. The \$ ha^{-1} gained (average \$12.8 thousand ha^{-1} created+restored) is inversely related to project cost, for projects ranging from \$0.5 to \$100 million. There is a 15-fold increase in \$ ha^{-1} gained as project size increases by a factor of ten. There appear to be generic economies of scale inherent to similar environmental management approaches that represent a compromise of at least three attributes: attempts to control ecosystem behavior (predictability and use), ecosystem complexity, and incomplete ecosystem knowledge. Small-scale projects are described that are very cost effective, including terracing, backfilling, restoration of abandoned agricultural lands (built in wetlands circa 1915), small crevasses and spoil bank management. The smallest projects (\$20,000 each) create land at slow rates (@ 5 ha yr^{-1}) and tend to be very cost effective (\$20 to \$500 ha^{-1}). There appear to be more than enough sites to apply these small projects at a rate that would equal the anticipated restoration and creation rates resulting from the current CWPPRA program.

Keywords: wetland creation, restoration, economy of scale, river diversion, Louisiana, coastal zone.

Introduction

Effective wetland restoration efforts in coastal Louisiana should be not only long-lasting (decades), but also economically efficient. Economists have long recognized that there is an economy of scale operating on agricultural lands. An Englishman and Frenchman, almost simultaneously (Sir James Steuart and Turgot 1767; in Schumpeter 1954) observed that as additional agricultural land was newly cultivated, that the yields per area eventually decreased, presumably because the new farm land was of more marginal value. Turgot, in particular, recognized an interval of increasing returns before the interval of decreasing returns. Subsequent observations have shown these general observations of the patterns in yield and agricultural land cultivated to extend to other endeavors in government, scientific research, and even household economies. Becoming aware of these economies of scale would seem to be a prudent aspect of both financial and political management in an era of limited resources for wetland restoration.

There are a variety of wetland creation, rehabilitation and restoration methods possible in coastal Louisiana whose scale varies widely. Most of the Coastal Wetland Planing, Protection and Restoration Act (CWPPRA) projects are individually large. For example, the 1995 review of 4 years of projects shows that the average project created or restored 203 ha at an average cost of \$2.6 million, or \$12.8 thousand ha^{-1} for the average of individual projects. The average cost/area of all projects was \$28.6 thousand ha^{-1} of restored or created land (Anon 1995). There are also several types of smaller projects whose costs and results

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are either well-documented or that have substantial estimates to justify their implementation. These latter projects are not common in south Louisiana, mostly for what seem (to me) to be for sociological or political reasons. They are relatively smaller projects than those funded by CWPPRA because of their individual cost or impact area. These smaller projects are briefly reviewed here and then compared to our experience with larger projects in terms of the economy of scale.

Examples of Low-Cost Projects

Small Crevasses (based on a summary in Boyer et al. 1997 and Turner and Boyer 1997)

Constructing "artificial" crevasses, or cuts in the natural levee of the Mississippi River, has been attempted by the US Fish and Wildlife Service in the Delta National Wildlife Refuge (DNWR) to build land within the Refuge (Fig. 1). The newly constructed crevasses create emergent wetlands after two years of subaerial growth at about 4-5 ha yr^{-1} (Fig. 2). The present total cost is <$200 ha^{-1} after six years and will decline to about $54 ha^{-1} as additional land builds. There is also documentation of the growth rate of larger natural crevasses, and of the costs and anticipated land gains (restoration or creation) from currently funded CWPPRA projects (Anon 1995). The range of the individual

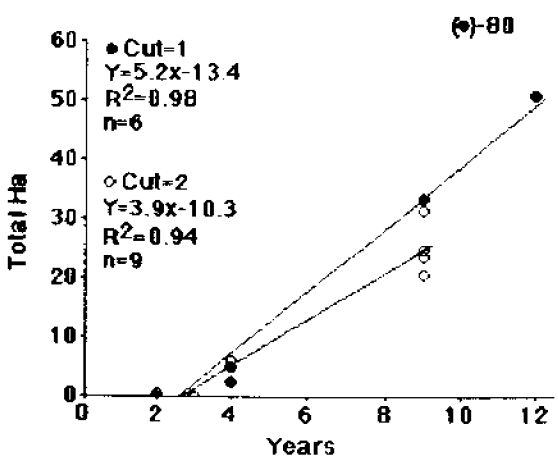

Fig. 2. The growth of land in newly created crevasses in the Mississippi River delta (from Boyer et al. 1997).

projects and water diversions range from $12 thousand to $76 million, the percentage river diverted varies from 0.06% to 19%, and the flow is directed through a channel from 30 to 900 m wide. The proposed or actual land gain rates from these crevasses range from 5 to more than 600 ha yr^{-1}.

Land gain increases with project size or the amount of water diverted, but not as fast as project costs increase. There is, therefore, an inverse relationship between the cost of land gained and the size of the crevasse (Fig. 3). Note that updated cost estimates are included for the West Bank, Davis Pond and Caenarvon Diversion that were not available at the time of the paper by Turner and Boyer (1997). Neither large nor small crevasses, of course, can be built everywhere.

Agricultural Impoundment Restoration (based on Turner and Neill 1984 and Trepagnier et al. 1995)

Impoundments occur throughout south Louisiana wetlands (Fig. 1). Some were built as wildlife refuges and others were formed by interlocking canal spoil banks. The oldest are probably those built to "reclaim" land for agriculture. These impoundments are visible reminders of the era, following passage of the Swamp Land Acts of the last century, when publicly owned swamplands were sold to individuals in transactions which often "lacked the characteristics of pristine honesty" (Harrison and Kollmorgen 1947). Some areas were promoted as having a very high potential for profit through agriculture.

Ten failed or abandoned coastal agricultural impoundments (circa 1915) were examined by Trepagnier et al. (1995) to determine recent wetland restoration rates in former agricultural impoundments that had failed. Aerial photography from between 1978 and 1988 was used to determine the percent wetland area and levee length and changes between mapping intervals. One site gained wetland between 1978 and 1988, and four sites gained wetland between 1985 and 1988. Wetland area in the remaining sites were either stable or declining during the study period. The average wetland

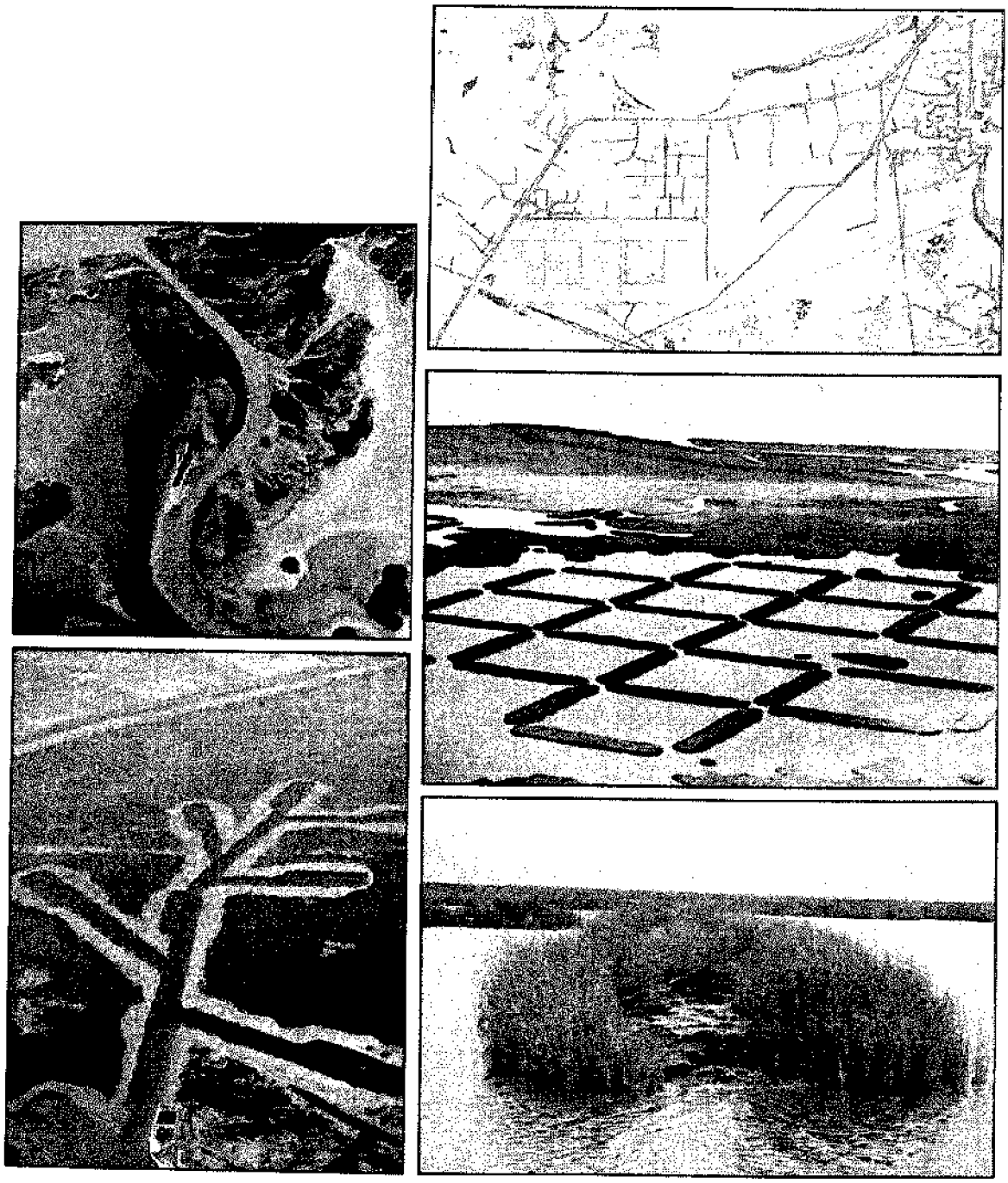

Fig. 1. Photographs of various small restoration projects or potential sites in south Louisiana. Top left: A recently-built crevasse (1990). Bottom Left: A network of spoil banks resulting from dredged canals. Top right: the square in the center is the Delta Farms open water area, which formed from a collapsed levee during a hurricane in 1960. The area immediately to the north was drained and farmed in 1915, but was freshwater marsh on 1930s aerial photos. Middle right: an aerial photograph of the Sabine terracing project. Bottom right: a ground view of a recently created terrace (+5 months) in the Sabine terracing project.

change rates for all areas ranged from -4.8 to + 2.54% yr^{-1} from 1978 to 1988. Although part of this variability in rates is due to mapping interpretations, hydrologic factors unique to each site were also important. The results from a multiple regression model indicated that restoration is inversely related to impoundment size and directly related to levee reduction. Trepagnier et al. (1995) used this statistical model to estimate the costs of removing the levees and the potential land gains. Levee removal will enhance wetland restoration rates at a very favorable cost (<$1 ha^{-1}) and be sustainable with little additional management. The recovery rate, however, was estimated to be about 1% yr^{-1} if 10% of the levee were removed. A rather clear example of the possibilities are shown in the area north of the open water body known as Delta Farms, in the Barataria Basin. The open water formed from a former agricultural impoundment whose levees collapsed in 1960. The area on the northern border was drained and farmed in 1915, but was abandoned sometime thereafter, perhaps after a hurricane. The land manager, C. Breaux, recalls dragging a tractor out of the marsh when he was a child (pers. comm.). Aerial photographs show that it was mostly wetland by the 1930s before the Intracoastal Waterway was built, and it remains that way today.

Terracing (based on information in CWPPRA Planning documents)

Terracing is a wetland restoration practice in south Louisiana that involves piling material on a shallow and submerged surface to form an exposed surface of dredged material. This new surface is not like a typical spoil bank dredged for navigation, which is continuous, in two parallel lines, and rises above normal high tides. These terraces are more like a series of disconnected ridges arranged perpendicularly to each other, whose surface is flooded by high tides (Fig. 1). The Natural Resources Conservation Service has constructed several of these in the Chenier Plain, and with some success. The purpose of the raised banks, or terraces, is to dampen the wave energy to allow plants to take root and suspended matter to fall out in greater amounts, and to protect the adjacent shoreline. Vegetation forms initially on the exposed spoil material. Later the vegetation extends into the water, but at presently undetermined rates. The cost/ha gained of this type of project is in the neighborhood of $3,000 to $8,000/ha after 5-10 years.

Spoil Bank Management (based on material in Turner et al. 1994a)

There is virtually no area of the Louisiana coast that is without a dredged canal or channel nearby. These canals are built with various dredging methods to facilitate navigation, belowground mineral recovery, pipeline construction, and trapping. Most canals, however, were constructed to service the oil and gas industry, especially to float in drilling equipment. They are occasionally re-dredged (or 'swept') when they fill in. In an aerial view of the marsh, canals appear as straight lines with a parallel man-made levee on either side (Fig. 1). Some canals are isolated, and others are found in dense networks. The linear features of canals are in sharp contrast to the anatomising natural channels that form a dendritic drainage pattern with naturally low levees.

The material removed to create the canal is deposited nearby in a continuous line of dredged spoil material. This man-made levee, called a spoil bank, usually has different vegetation (e.g., shrubs) from the surrounding wetland. Spoil bank levees have an average lifetime of less than 50 years (Monte 1978). Gas right-of-way pipelines may not have spoil banks because they are often filled in with the previously dredged material immediately after the pipeline is laid (backfilling); these 'backfilled' canals frequently re-vegetate enough to make the new plant cover indistinguishable from the surrounding marsh. The aggregate length of these canals and spoil banks is in the range of 9 to 11,000 km and 18 to 22,000 km, respectively.

The cumulative impact of constructing many small individual canals and levees is to impound marshes, often unintentionally, and cause land loss. Spoil banks change the flow of water into and out of the marsh (Swenson and Turner 1987), cause open water ponds to form nearby (Turner and Rao 1990),

and are spatially and temporally related to the conversion of wetland habitats to open water (e.g., Bass and Turner 1997; Turner 1997).

Indirect impacts result from the: (1) longer wetland drying cycles, even in semi-impounded wetlands, as a consequence of altered water movements into and out of the wetland. The lengthened drying periods promote soil oxidation and subsequent soil shrinkage; (2) flooding events that may lengthen behind spoil banks, presumably as a consequence of water being trapped behind the spoil bank once water enters overland during very high tides. When wetland flooding increases enough to seriously waterlog soils and change soil chemistry, plants may become stressed to the point where growth reduction or even die-back occurs (e.g., Mendelssohn et al. 1981; McKee and Mendelssohn 1989); (3) lower sedimentation rates behind spoil banks in any wetland type, because of the reduced frequency and depth of tidal inundation; (4) In addition, the spoil banks consolidate the underlying soils. Subsurface water movement is thus decreased, both because of the reduced cross-sectional area and the reduced permeability of material beneath the levee (Turner 1987).

There is no systematic management plan to restore spoil banks to wetlands by removing them. We used a hydrologic model and field data to estimate that removing about 10% of the spoil bank would restore hydrologic flows *if the spoil removal were strategically placed*. The restoration cost ha^{-1} for projects 1 to 10,000 ha ranges from $1,000 to $1, respectively. The economy of scale is due both to the high mobilization costs for small projects, and, to the perimeter:area relationships affecting hydrologic flows.

Backfilling Canals (from Turner and Neill 1984, 1994; Neill and Turner 1987)

The return of former spoil bank material back into the canal is referred to as "backfilling". The intended result of backfilling is to reestablish the natural drainage network of the wetland and to offer increased habitat to habitat-limited wetland flora and fauna. Wetland conservation with improved shallow water habitat is, therefore, the general goal of backfilling. Canal backfilling is generally intended to directly achieve four objectives:

1) To fill in the canal,
2) To reestablish emergent marsh vegetation in the canal,
3) To eliminate the spoil bank, and,
4) To reestablish emergent marsh vegetation on the spoil bank.

Backfilling began to be required in 1979 as a condition for issuing a permit to dredge some canals. Canals may be backfilled after the drilling site is abandoned (on-site mitigation), or, another canal may be backfilled (off-site mitigation) to obtain a permit for dredging a new canal or other activity. The spoil bank is leveled as near to marsh elevation as possible, and the spoil bank material that is moved is used to fill in the existing canal. The fill is placed evenly over the bottom of the canal.

While the backfilled canal and adjacent berm we studied was not restored to a completely natural condition, its status was intermediate in many respects to that at a natural bayou and an unfilled canal (Turner et al. 1994b).

The reduction in elevation of the spoil bank at the backfilled canal resulted in greater water movement in and out of the marsh, reduced the intensity of soil waterlogging in both the streamside and inland marshes, and promoted the growth of marsh grasses, rather than upland vegetation, in the shallow canal zone parallel to the spoil bank (called a 'berm'). In addition, the partial filling of the canal prevented the development of reducing conditions in the bottom sediments and sulfide accumulation, promoted the establishment of submerged aquatic vegetation, and created productive, shallow water benthic habitat. Backfilling resulted in both a shallower canal and a more natural adjacent marsh and thereby partially reversed the changes wrought in the marsh system by canal construction. In a broader sense, backfilling can affect land-water conversion by several routes: 1) direct gain, 2) reduced erosion through changes in water regime, and 3) increased accumulation of land-building

sediments in the marsh through increased overland water flow and increased belowground plant productivity.

Backfilling costs were estimated assuming an average backfilled canal by Neill and Turner (1987). Backfilling restores wetlands at a cost of \$1,200 to \$3,400 ha^{-1}, depending on whether only the direct, or also the indirect impacts, respectively, are included. The cost of dredging the original canal is about \$25,000 ha^{-1}.

Comparisons with Large Projects (from Boyer et al. 1997, Turner and Boyer 1997)

The cost per area gained in restoration has been compared for crevasses and all projects by Turner and Boyer (1997). In general, the economic efficiency of planned restoration projects decreases as project size increases. The inverse relationship is not trivial, but covers several orders of magnitude. The range is from less than \$100 ha^{-1} to more than \$10,000 ha^{-1} (Fig. 3). A 1000-fold increase in project cost results in a 100-fold increase in the cost of creating each ha for river diversions. A 10-fold increase in project cost results in a 15-fold increase in cost (\$ ha^{-1}) for all the coastal restoration/creation projects started in the first 4 years of CWPPRA (Fig. 4, left side). It might seem that wetland area is an imperfect measure of secondary project benefits (e.g., erosion protection, percent plant cover, access for marine organisms). For this reason, the various CWPPRA agencies developed a Wetland Value Assessment (WVA) to estimate ecosystem changes for CWPPRA projects in terms of "habitat units". The cost habitat $unit^{-1}$ also demonstrates an inverse economy of scale (Fig. 4, right side).

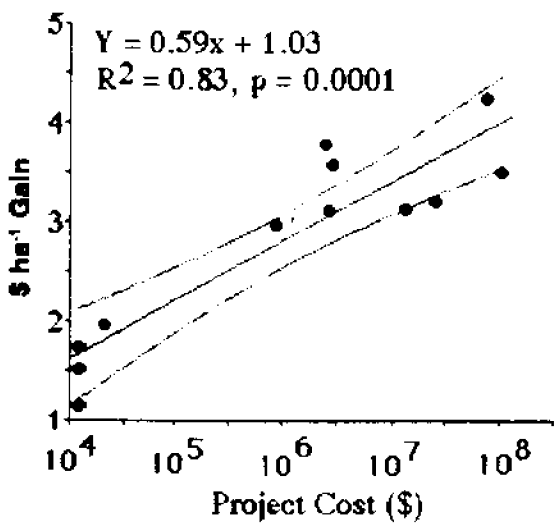

Fig. 3. The relationship between river diversion size and cost. A linear regression of the log-10 transformed data is shown with the 95% confidence limits. Data were calculated on an annual basis assuming a 50 year lifetime. A 8 Jan. 1998 Louisiana Department of Natural Resources Project Status Report was used to update the West Bank diversion costs. The Davis Pond and Caenarvon diversion are included at the 1998 cost.

The small projects described here (small crevasses, backfilling, spoil bank management, agricultural impoundment restoration and terracing) have much lower costs than the average CWPPRA projects. In other words, smaller projects create and restore coastal land much more effectively than the larger projects.

The most cost-effective land building projects are the smaller ones. River management becomes increasingly expensive, and dramatically so, as the project size increases. Bureaucratic oversight becomes more demanding. It might be possible to reduce the cost of the larger projects with experience and further review. However, the scale effects appear to be too large and consistent to be the result of mis-judgment, inexperience, or math errors. Turner and Boyer (1997) suggested that the observed inverse economies of scale for coastal Louisiana wetland restoration projects are the consequence of at least three attributes: attempts to control ecosystem behavior (predictability and use), ecosystem complexity, and incomplete ecosystem knowledge. The uncertainty embedded in these projects is why monitoring is included (up to 20% of project costs) and why engineers sometimes seem to "overbuild". The large size of some of these projects is unique in the history of coastal Louisiana. We should expect uncertainty.

Although many smaller projects are more cost effective than a few large ones, small projects cannot be built everywhere, anymore than large projects can be built everywhere. Individual project types

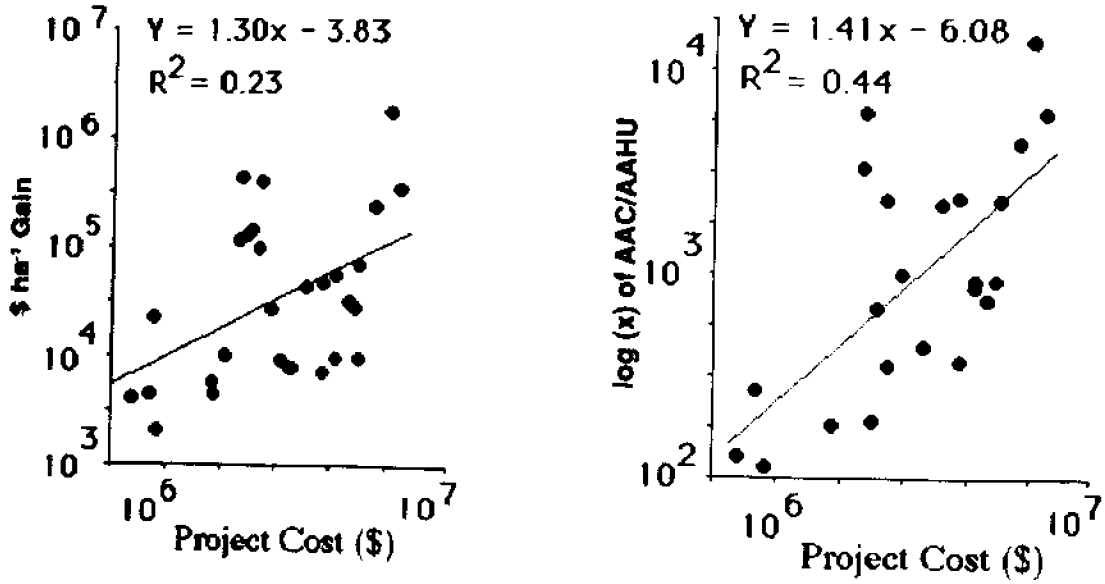

Fig. 4. The relationship between the cost per ha land created or restored and the project cost for all CWPPRA projects in south Louisiana, including diversions. Left: $ ha^{-1} cost vs. project cost. Right: Average Annual Cost per Average Annual Habitat Unit (AAC/AAHU) vs. project cost. The data are from Anon (1995).

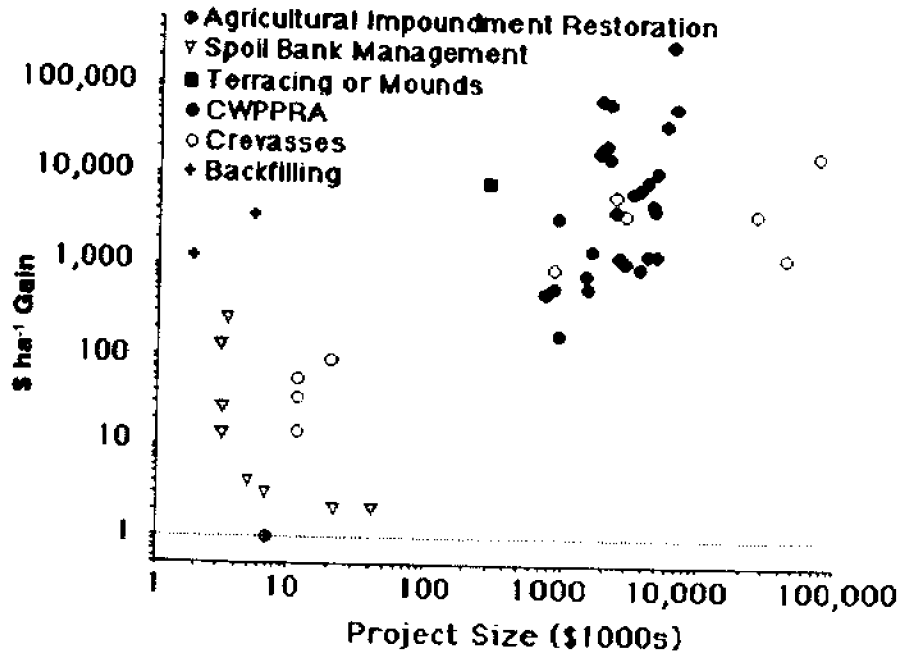

Fig. 5. The relationship between various small and large restoration projects in coastal Louisiana (implemented and un-implemented project costs are averaged). The relationship between restoration/creation project size and the cost per ha wetland restored or created. The literature sources are from: CWPPRA (Coastal Wetland Planning, Protection and Restoration Act), Anon 1995; agricultural impoundment restoration sites: Trepagnier et al., 1995; spoil bank management: Turner et al, 1994b; crevasses or river diversions: Turner and Boyer 1997; backfilling canals: Turner et al., 1994a; terracing or mounds: Anon 1995.

have inherent requirements that dictate, to some extent, where they can be implemented. These constraints may require adjustments in management efforts to achieve optimum uses of the available funding and ecological resources. It is worth noting, however, that the vast majority of CWPPRA projects are large projects. The first 4 years of the CWPPRA program had only 1 of 63 projects that cost less than $500,000.

It is worth asking if there is enough area to use these smaller projects. Only preliminary numbers are available, but they suggest that small projects are a viable option for restoration efforts. The CWPPRA program has projects in place to attempt to create and restore wetlands at a rate of 479 ha yr^{-1} (Anon 1995). A small crevasse builds at 4.76 ha yr^{-1} (Boyer et al., 1997), so it would take only 100 small crevasses, at $48/ha, to create what costs the CWPPRA program $28,000 $h^{-1}a$. There are a few hundred locations available to build crevasses (Boyer and Turner 1995), but land owner permission is required—something that may not be forthcoming. The open water in abandoned agricultural impoundments could be recovered at 10 ha/yr. (Trepagnier et al., 1995), at a cost of about $1 ha^{-1}. There are at least 43 impoundments, perhaps 20 of which could be restored with permission. The average size is 2,352 ha. The potential restoration area is, assuming 50% recovery, 23,520 ha, at 50 years, or 470 ha yr^{-1}, equivalent to the CWPPRA land gain at <0.01% of the per ha cost. The greatest opportunity to implement low cost restoration projects has to do with restoring oil and gas recovery canals and reversing their negative impacts on wetland hydrology. There are tens of thousands of spoil banks and canals that could be backfilled, equaling 80,426 ha in 1978 (Bauman and Turner 1990). There is a much larger area of impoundments and semi-impoundments that could be managed, an area that perhaps exceeds 500,000 ha. It would be necessary to convert only 0.15 % yr^{-1} of the canals, spoil banks, semi-impoundments and impoundments to wetlands each year (at less than $1000 ha^{-1}), to match the objectives of the projected results of current CWPPRA projects.

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WETLAND MANAGEMENT

Effects of Weirs on the Depth and Duration of Flooding in a Louisiana Marsh

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ABSTRACT: This analysis examines the effects of two marsh management weirs on the depth and duration of marsh flooding, and utilizes the hourly water-level data at four of the real-time Data Collection Platforms (DCP) for the period of September 1, 1993 to August 1, 1996. The continuous water-level data on the inside and outside of the water control structures at Mill Creek and Bayou LaCarpe indicate that the weirs significantly altered the number and duration of flooding events on the surrounding marsh (Kolmogorov-Smirnov critical values of 0.14 and 0.10 respectively). This resulted in fewer short duration (<24 hr) flooding events and an increase in long duration (>2 wk) flooding events. Subsequently, the marsh was inundated to a greater depth and for longer durations with the presence of water control structures. At Mill Creek, where the weir was the only link to the surrounding hydrology, these effects are more significant than at Bayou LaCarpe where the water is able to indirectly bypass the weir. Subsidence, complex drainage patterns, improper project design, and inadequate structure manipulations are the likely reasons for the failure of the project at controlling hydrology.

Introduction

Louisiana possesses 41% of the total coastal wetlands in the contiguous U.S. and these wetlands are in a severe state of degradation due to natural and anthropogenic causes (Turner 1990). Mass harvesting of cypress timber beginning in the early 1900s and dredging of oil and gas access canals beginning in the 1940s dramatically changed the landscape of coastal Louisiana (Myers et al. 1995; Reed and Rozas 1995). Perhaps more important than the direct loss of coastal wetlands from canal construction is the resulting change in hydrology and subsequent changes in marsh creation processes (Cahoon 1994).

Marsh management in Louisiana began in 1954 in the form of manipulated impoundments in an attempt to curb the effects of these natural and anthropogenic changes (Chabreck 1960, 1962). The primary goal of these early marsh management ventures was to encourage the growth of plant species deemed "desirable" for waterfowl and fur-bearing animals and to create "optimal" marsh/water interspersion for waterfowl by controlling water level and salinity (Chabreck 1960). Structural weirs were first used in the early 1940s to gain access to trapping areas by maintaining minimal water levels for the safe passage of boats. The use of weirs as a marsh management tool did not begin until a decade later (Chabreck and Hoffpauir 1962).

Those early passive methods were replaced with active techniques as the goals of marsh management changed. Present marsh management techniques in Louisiana include the use of levees,

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pumps, and a variety of structure types in combination to control water levels and salinities. In addition to advances in the construction and operation of weirs, flapgates, culverts, and other marsh management structures, the basic goals have shifted toward a prevention of the conversion of marsh into shallow open-water areas. Active marsh management indicates that water control structures are operated to moderate water level variability, reduce saltwater introduction, and seasonally change the volume of water in the management area for the benefit of both vegetation and wildlife. The results from previous studies indicate that this type of management can be successful in enhancing plant growth when proper drawdown is achieved, and in increasing waterfowl and wildlife numbers (Hess et al. 1989). However, marsh accretion rates have been reported to be lower in managed marshes than in comparable unmanaged reference marshes (Cahoon 1994). Achieving the goals set forth for an active marsh management project involves a series a trade-offs in which managers may not be able to capitalize on maximum sediment and freshwater availability in lieu of considerations for plant growth and wildlife behavior patterns (Cahoon and Groat 1990). Conflicting opinions on the impacts of active marsh management necessitate our continued effort to examine the available monitoring data and evaluate the effectiveness of these projects at meeting their management goals (Cowan et al. 1988).

We report the results of approximately 3 years of hourly water level and salinity monitoring in the Falgout Canal Protection project. The goal of this study was to determine the effectiveness of a marsh management project at meeting its target water levels and salinity. Specifically, we are looking at the effects of two weirs on the depth and duration of marsh flooding. In lieu of any pre-project monitoring or a suitable reference area, continuous data on the inside and outside of two water control structures are utilized to determine whether the Falgout Canal Protection project has met its target water levels in the first three years of operation.

The Falgout Canal Protection project is an active marsh management project bordered by Falgout Canal to the south, the Houma Navigation Canal (HNC) to the east, the natural ridge of Bayou du Large to the west, and a natural ridge to the north (Fig. 1). This marsh management plan is located in the deltaic plain of Louisiana where wetland loss rates are estimated at 65.6 km^2/yr (Dunbar et al. 1992). The project area, which encompasses 3,004 ha (7,423 acres), consists of cypress/tupelo swamp (living and dead), fresh/intermediate marsh, brackish marsh, and open water. Anthropogenic changes, such as the construction of navigation canals and their respective levees, have altered the hydrology of the area. These changes allow for greater tidal influence and the introduction of saline waters into freshwater wetland areas (Wang 1988). Salinities in the area currently range from 0 to 21 ppt. These anthropogenic changes to hydrology coupled with the high rates of local subsidence (0.80 cm/yr to 1.18 cm/yr in Terrebonne parish), contribute to the loss of marsh in the area (Penland et al. 1988).

In 1956, 95% of the project area was emergent vegetation according to a soil survey map of Terrebonne Parish (U.S. Department of Agriculture 1956). The dominant vegetation at that time was *Alternanthera philoxeroides* (Martius) Grisebach (alligatorweed), *Cephalanthus occidentalis* L. (buttonbush), *Myrica cerifera* L. (waxmyrtle), *Pontederia cordata* L. (pickerelweed) and *Taxodium distichum* (L.) Richard (bald cypress) (USDA 1956). Analysis of near-vertical, color-infrared aerial photography (1:12,000) from 1989 indicated 60 to 70% of the area had been converted to open water and that 70% of the existing cypress swamp was dead or dying.

The objectives of the Falgout Canal Protection project are to preserve and enhance 3,004 ha (7,423 acres) of marsh and cypress/tupelo swamp, reduce saltwater intrusion, and improve wildlife habitat. The project consists of three elements: (1) levee construction and maintenance, (2) seven water control structures (e.g., variable crest weirs), and (3) a pumping station (Fig. 1). Construction of the project began July 2, 1992 and ended on April 2, 1993. Levee construction occurred along 5,730 m (18,800 ft) of the northern bank of Falgout Canal (the southern project boundary) and repairs were made to 10,516 m (34,500 ft) of spoil bank along

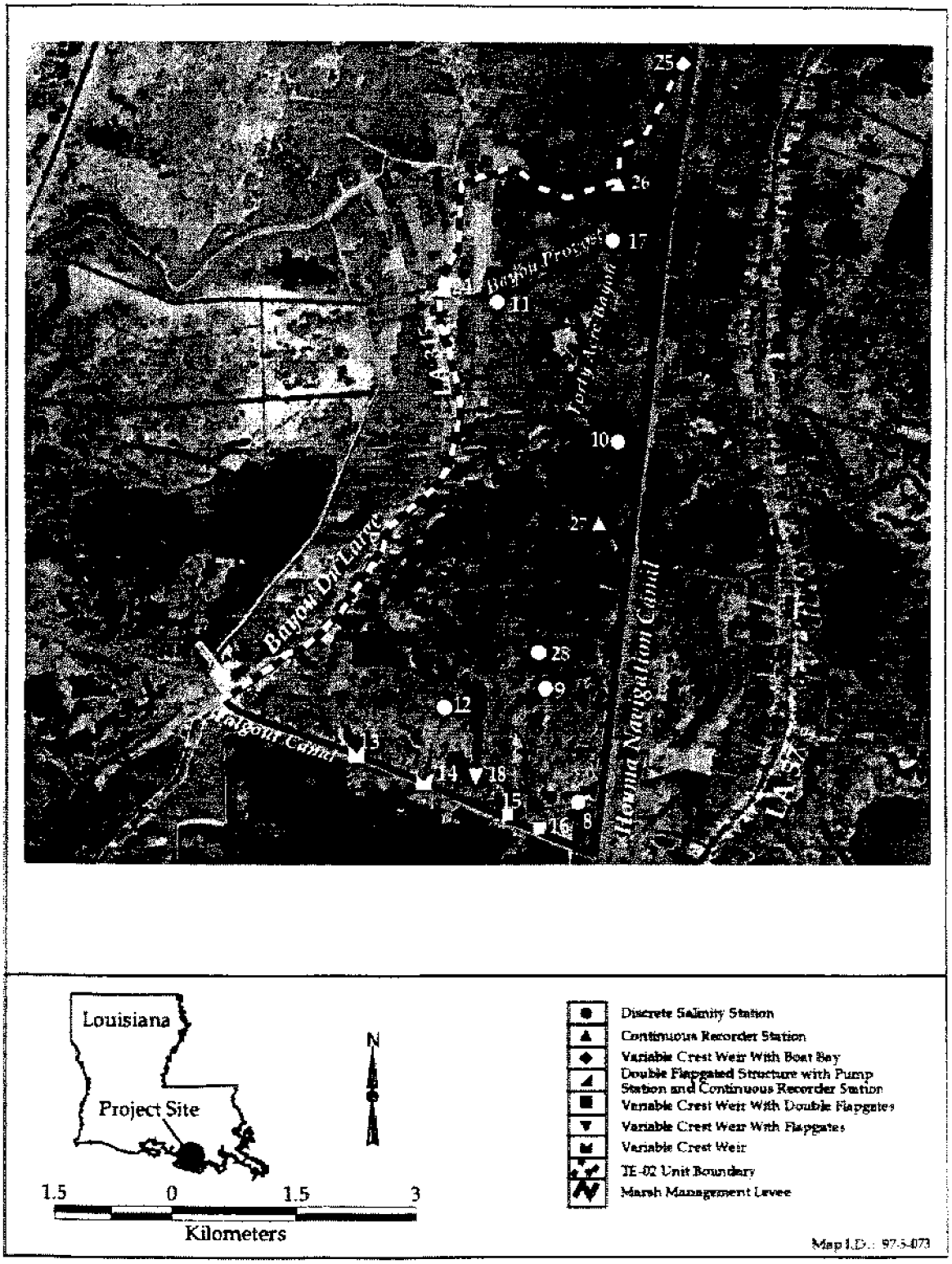


Fig. 1. Falgout Canal Protection (TE-02) project location and features.

the western bank of the HNC (the eastern project boundary). Five water control structures are located along Falgout Canal (stations 13, 14, 18, 15, and 16), one water control structure is located at the intersection of Forty Acre Bayou and the HNC (station 25), and one water control structure is located at the intersection of Bayou Provost and Bayou du Large (station 24) (Fig. 1). The pumping station is located at the structure in Bayou du Large (station 24). The pump is designed to draw fresh water from Bayou du Large into the project area via Bayou Provost. However, it has never been activated and the Louisiana Department of Natural Resources, Coastal Restoration Division (LDNR/CRD) personnel have observed storm water being pumped into Bayou Provost from adjacent agricultural fields on at least two occasions. Operation of these water control structures contributes to the project objectives and its specific goals.

The seven water control structures of the Falgout Canal Protection project are scheduled for operation in two phases. Phase I consists of a drawdown period in which water levels are to be drastically reduced to enhance revegetation of emergent marsh. Phase II, or the maintenance phase, is designed to control the flow of water through the project, keeping water off the marsh surface during the growing season, and strategically flooding the marsh at other times of the year. The project was designed to be in Phase I for the first year of operation, and at subsequent times if deemed necessary; otherwise, the project shall be in Phase II. When the project is in Phase I (drawdown), operational changes are scheduled at four times during the year (16 January, 1 March, 16 June and 2 November). When the project is in Phase II (maintenance), operational changes are scheduled at three times during the year (16 January, 1 March and 2 November).

Methods

The two structures of interest in this analysis are located at stations 18 (Mill Creek) and 25 (Bayou LaCarpe) (Fig. 1). These are the only two weirs that have continuous Data Collection Platforms on both the inside and outside of the project. Water level and salinity data collected at these stations were used to determine the effectiveness of the two weirs at meeting the project goals of maintaining water levels below the marsh surface during the growing season, achieving prescribed water levels set forth in the management plan, and reducing salinities inside the project area. Target water levels were based on marsh surface elevation, and range from 0.3 m (1.0 ft) below to 0.05 m (0.167 ft) above the marsh level. For the majority of the year, including the entire growing season, water levels are intended to be below the marsh surface. There are no set target salinities, but there are several safety provisions in the operational schedule to close various structures when salinities greater than 1.0, 3.0 and 5.0 ppt respectively are detected outside of the project area (Coastal Use Permit P850732).

The structure at Mill Creek is a variable crest weir with two 1.83 m (6 ft) bays with a one-way flapgate on each bay (Fig. 2a). The flapgates are designed to facilitate water flow out of the project when they are in the operating position. The only time that these flapgates are to be operating is from March 1 to June 15 of a Phase I (drawdown) year. The structure at Bayou LaCarpe is a variable crest weir with eight 1.52 m (5 ft) bays and one 3.66 m (12 ft) boat bay (Fig. 2b).

The four Data Collection Platforms (DCPs) utilized in this analysis are located inside and outside the project area (on either side of the respective water control structures) at stations 18 and 25 (hereafter, Mill Creek and Bayou LaCarpe, respectively) (Fig. 1). The DCPs record water depth, specific conductance and temperature hourly with Handar 436A incremental shaft incoders and Hydrolab H20s. The U. S. Geological Survey (USGS), under contract to LDNR/CRD, services the DCP units every 4 to 6 weeks, and compensates the data for biofouling when the margin of error is greater than 5% when compared to a newly calibrated hand-held instrument. When compensation is necessary, a linear shift is applied to the data; however, no shifts in the data were needed for the period of this study. The data can be accessed through a direct satellite link or through a modem link to the USGS data base. Data were collected from these stations from September 1, 1993 to August 1, 1996.

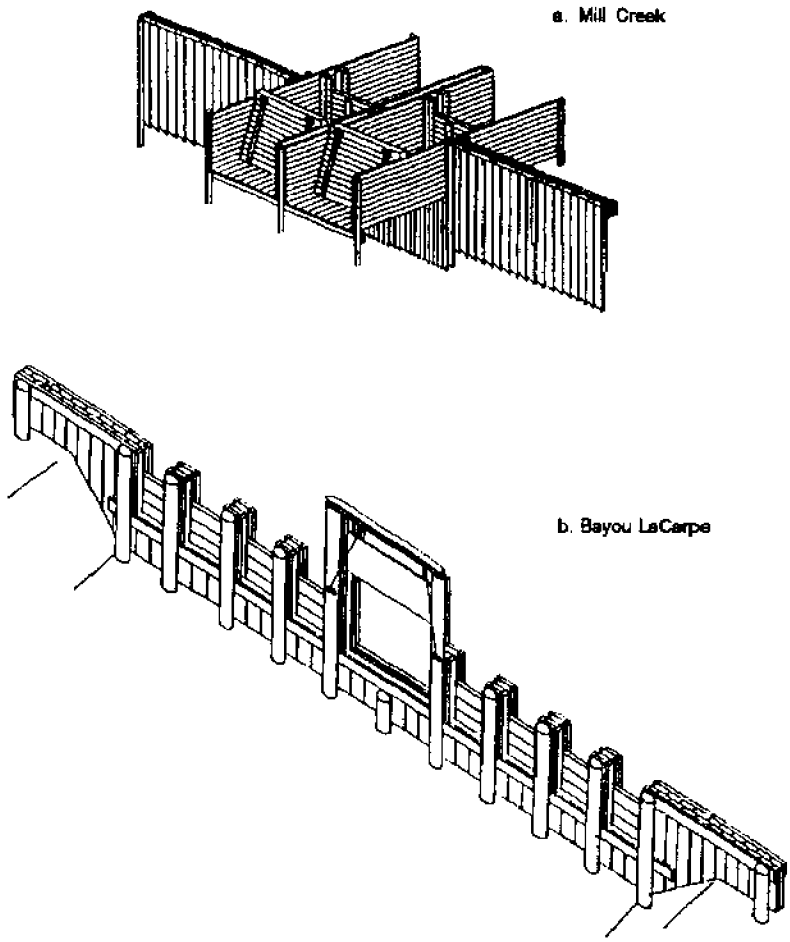


Fig. 2. Weir design at (a) Mill Creek and (b) Bayou LaCarpe.

Fourteen staff gauges, surveyed to National Geodetic Vertical Datum (NGVD), were installed at the seven structures in June 1995. Additionally, marsh elevations in the southern, middle, and northern portions of the project were surveyed to NGVD in June 1995.

Results

Operational changes at the structures have occurred on seven occasions since project completion (Table 1). Six additional operational changes were scheduled, but there is no documentation for any of these changes and it is therefore uncertain as to whether or not they were actually implemented. The project was in Phase I (drawdown) from March 1994 to June 1994 and from March 1996 to April 1996. Otherwise, the project was in Phase II (maintenance) (Table 1).

Mean marsh elevation is 0.341 m (1.12 ft, SD=0.10 ft) NGVD in the northern zone (near station 26, Bayou LaCarpe), 0.411 m (1.35 ft, SD=0.15 ft) NGVD in the middle zone (near station 27), and 0.539 m (1.77 ft, SD=0.12 ft) NGVD in the southern zone (near station 18, Mill Creek) (see Fig. 1 for station locations). Hourly water level at Mill Creek was ≤ mean marsh elevation 29.7% of the 1994 growing season (from March through October) and 40.2% of the 1995 growing season. Hourly water level at Mill creek was ≤ mean marsh elevation 39.1% of the growing season for the entire data set (September 1993 through August 1996) and was ≤ the upper limit of the 95% confidence interval

Table 1. Operational changes executed for Falgout Canal Protection (TE-02) project structures at stations 18 and 25. Height of stoplogs in the bays are expressed in centimeters NGVD.

Date	Station 18[a]	Station 25[b]	Phase
2-Apr-93 (construction end)	-15 (-0.5 ft) flaps raised	-31 (-1.0 ft) bay 4, -6	Phase II (maintenance)
8-Mar-94	-61 (-2.0 ft) flaps operating	+46 (+1.5 ft)	Phase I (drawdown)
4-Jun-94	-15 (-0.5 ft) flaps raised	-31 (-1.0 ft)	Phase II (maintenance)
17-Oct-94	+31 (+1.0 ft) flaps raised	-15 (-0.5 ft)	Phase II (maintenance)
21-Nov-95	-15 (-0.5 ft) flaps raised	-15 (-0.5 ft)	Phase II (maintenance)
24-Mar-96	-61 (-2.0 ft) flaps operating	+61 (+2.0 ft)	Phase I (drawdown)
13-Apr-96	-15 (-0.5 ft) flaps raised	-31 (-1.0 ft)	Phase II (maintenance)
25-May-96	-15 (-0.5 ft) flaps raised	+61 (+2.0 ft)	Special Request (partial drawdown)

[a] = 2 bays, 2 one-way flapgates
[b] = 8 bays, 1 boat bay

of the mean marsh elevation 40.7% of that period. Hourly water level at Bayou LaCarpe was ≤ mean marsh elevation 21.1% of the 1994 growing season (from March through October) and 10.8% of the 1995 growing season. Hourly water level at Mill Creek was ≤ mean marsh elevation 25.5% of the growing season for the entire data set (September 1993 through August 1996) and was ≤ the upper limit of the 95% confidence interval of the mean marsh elevation 27.6% of that period.

For the entire data set, the marsh surface was flooded 57.1% of the time inside and 58.3% of the time outside the weir at Mill Creek. At Bayou LaCarpe, the marsh surface was flooded 54.8% of the sampling period inside the weir and 55.0% of the time outside of the weir. However, the distribution of the duration of flooding events was significantly different between the inside and outside of the structures at both stations (Kolmogorov-Smirnov critical values of 0.14 and 0.10, respectively) (Fig. 3). Fewer short (<24 hr) flooding events and more long (>2 wk) flooding events occurred inside the project area. Despite no difference in the percentage of time marshes were flooded inside and outside the managed area, the presence of weirs facilitated longer and deeper flooding events.

Mean water level inside the project area was approximately 0.15 m (0.5 ft) higher than outside, but this difference was not statistically significant (p=0.32) due to the high variance of water levels outside the structures. There was no significant difference in water levels between stations 18 and 25 (p=0.12). Water levels did vary significantly between the inside and outside of both weirs at any given month and year (side*station*year*month

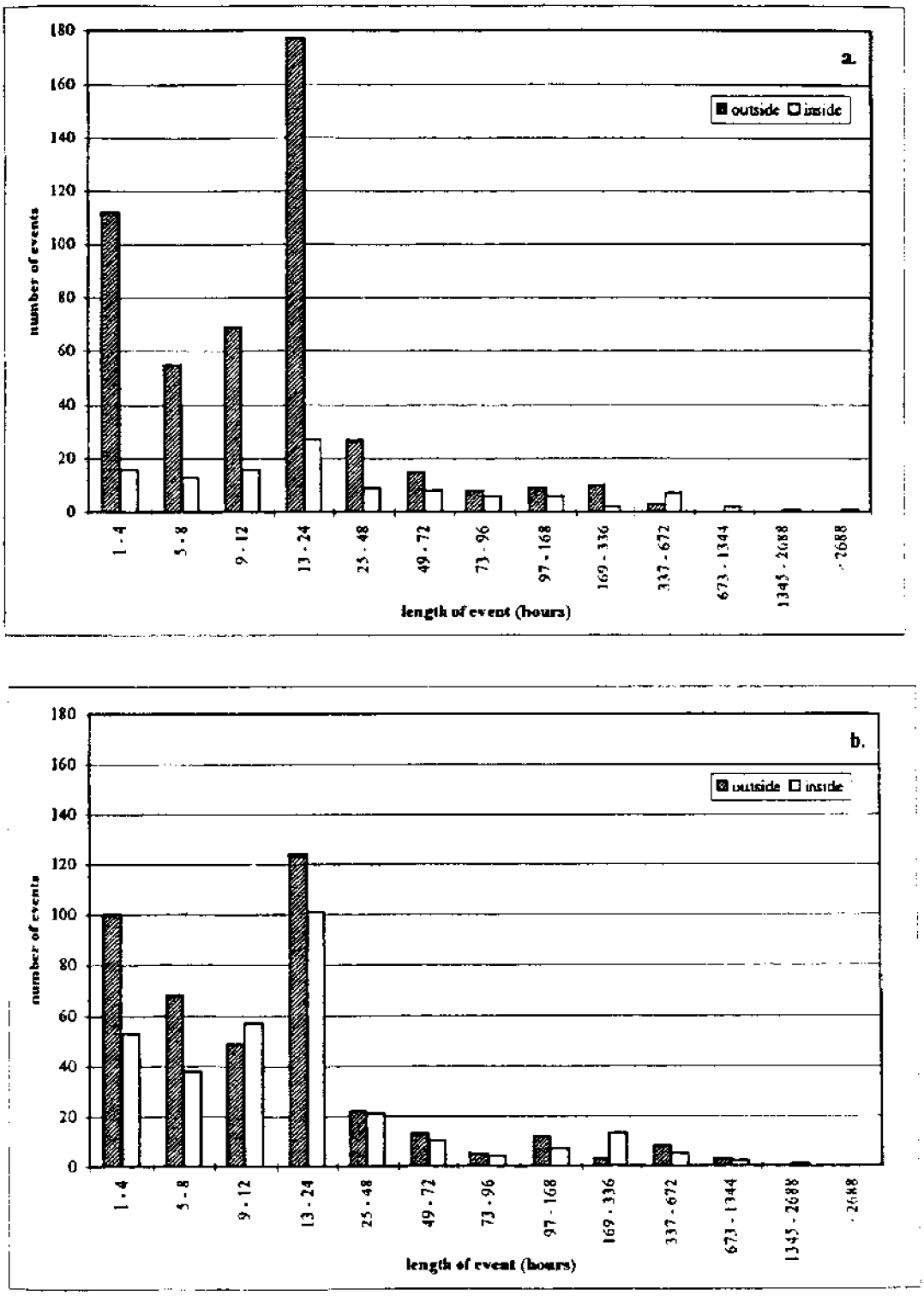


Fig. 3. Duration of flooding events inside and outside the structures at (a) Mill Creek (station 18) and (b) Bayou LaCarpe (station 25).

interaction significant [$p<0.01$]). The structure at station 18 appears to dampen water level variability more than the structure at station 25 (Fig. 3).

The project was in Phase I (drawdown) for only three months (8 March - 4 June) in 1994, and less than one month (24 March - 13 April) in 1996. The marsh management plan calls for a target water level of 6 to 12 inches (approximately 15 to 30 cm) below mean marsh elevation during the March to June drawdown period. During the 1994 March to June drawdown period, water levels actually ranged from 15 to 30 cm above mean marsh elevation, thus failing to meet the target water level. Although water levels were lowered (0 to 20 cm relative to outside water levels), they were not lowered to the extent called for in the marsh management plan (relative to marsh elevation) for the brief period in which the project was in Phase I (drawdown) (Table 1). An increase in the duration of flooding events generally corresponded to an increase in the depth of flooding although this pattern was not exhibited on the inside at Mill Creek (Figs. 4 and 5).

Salinities in the project were significantly lower inside than outside (0.09 ppt difference) ($p<0.01$), and salinities at Mill Creek were significantly higher than those at Bayou LaCarpe (1.35 ppt difference) ($p<0.01$). The difference in salinities between inside and outside varied between stations (station*side interaction significant [$p<0.05$]). Hourly salinities were higher inside than outside the project 58.9% of the period reported for Mill Creek and 34.0% of the period reported for Bayou LaCarpe.

Discussion

The Falgout Canal Protection project has not met the goal of maintaining water levels below marsh elevation during the growing season. The structures do not provide adequate control of water levels with the present operational scheme. Water levels were above mean marsh elevation a large percentage of time during the growing season, as drawdowns were insufficient to dewater the project area. Although the number of short flooding events in the managed area was decreased, the number of long flooding events (>2 wk) was increased relative to the unmanaged area (Fig. 3). An increase in marsh submergence has been shown to be more detrimental than somewhat elevated salinities to *Sagittaria lancifolia* L., a dominant marsh plant in the project area (Webb and Mendelssohn 1996).

Undesireable water levels within the project area could have resulted from improper operation of the structures, especially in emergency high salinity/high water events when management personnel could not respond immediately because of more pressing needs to protect inhabited areas. However, even when operations were made as scheduled, the water levels were not reduced to levels outlined in the management plan.

These conclusions are based on continuous data collected directly in the vicinity of two weirs. The lack of any substantial data on conditions that existed in the project area prior to construction as well as the lack of any monitoring in a nearby reference area, make comparisons between preexisting and current conditions difficult. Future restoration efforts should place more of an emphasis on preconstruction monitoring and simultaneous monitoring of surrounding wetlands.

Observations of landowners and LDNR/CRD personnel indicate that water in the northern portion of the project area flows from station 24 toward the east and north to station 25 (Fig. 1). This drainage pattern is confirmed by the lower marsh elevations in the northern portion of the project, and by an ongoing hydrologic survey (Arthur Long, personal communication). The entire northern hydrologic unit does not appear to be included within the project area, as there is water exchange between the HNC and the project area through breaches in the project boundary, and flows bypass the structure at Bayou LaCarpe. The northern portion of the project sustained widespread damage from airboats, marsh buggies, and chain saws during seismic work performed in January 1996. Much of this damage completely destroyed the vegetation and significantly changed the hydrology of the area when new channels were created and levees were breached. A lack of hydrologic control in the vicinity of Bayou LaCarpe may account for the less pronounced difference in the depth and duration of

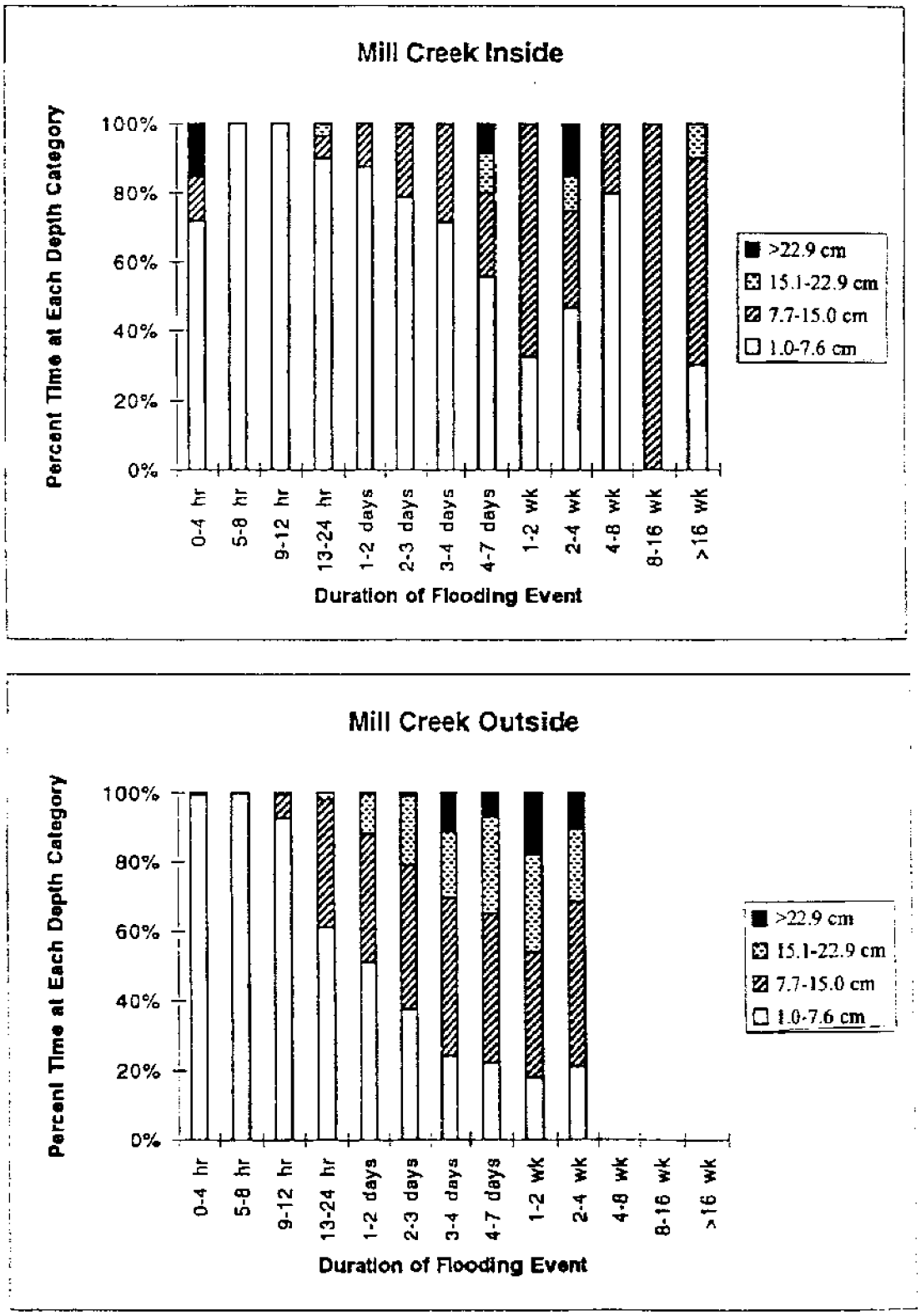
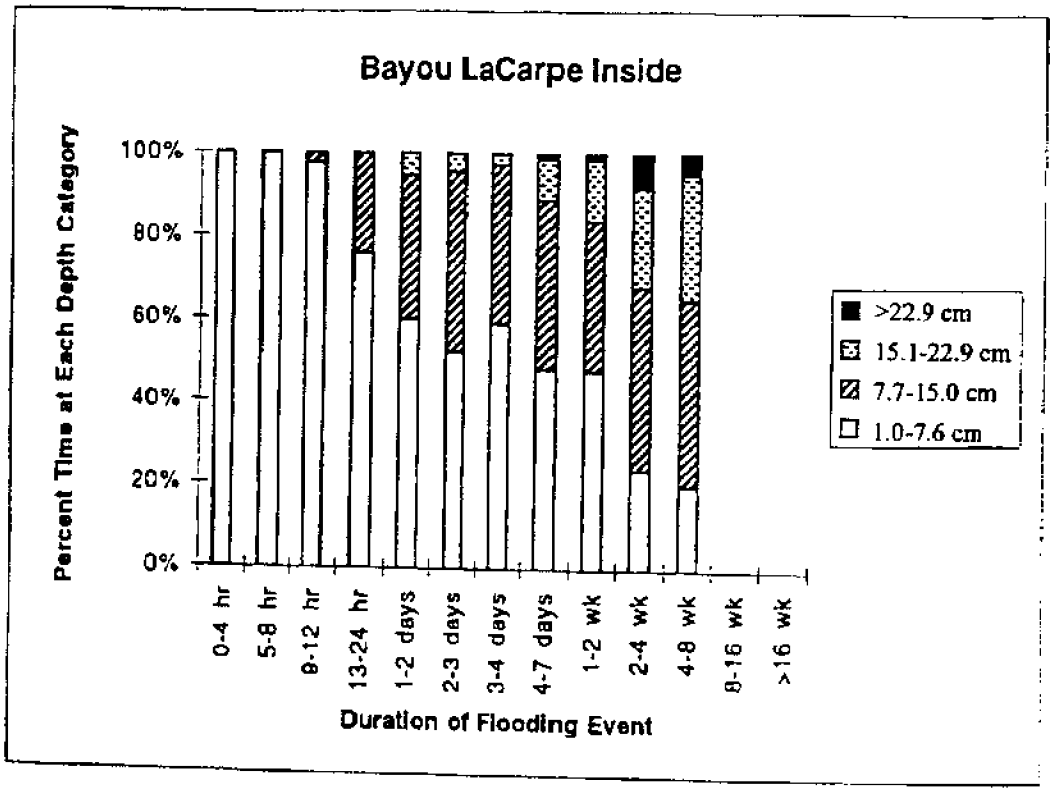


Fig. 4. Proportional depth of flooding for each duration of flooding event at Mill Creek inside and outside.



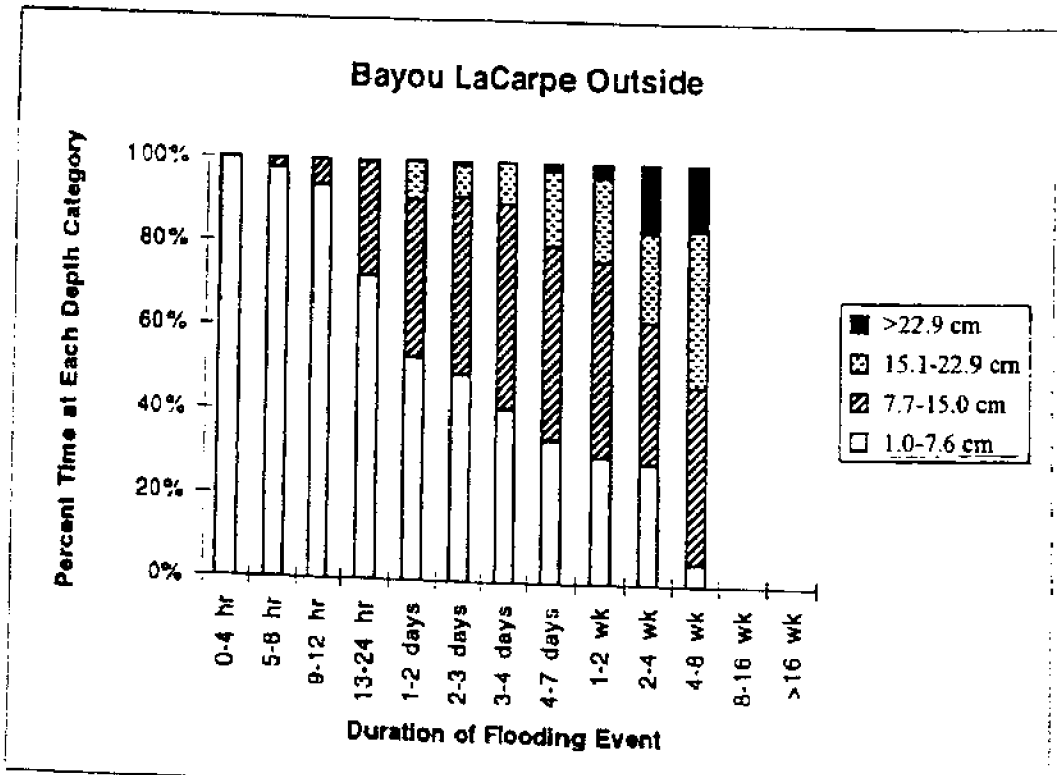


Fig. 5. Proportional depth of flooding for each duration of flooding event at Bayou LaCarpe inside and outside.

flooding events between the inside and outside of the project area (Figs. 3 and 5) as compared to Mill Creek where the weir is not bypassed (Figs. 3 and 4).

The water level pattern at the Falgout Canal Protection project is consistent with the findings of previous studies of managed marshes. Structural marsh management often reduces the coupling of the managed marsh with the surrounding hydrology (Boumans and Day 1994). A reduction of water level fluctuations in managed marshes can also indicate a reduction in the net exchange of nutrients and sediments and lead to a decrease in marsh accretion (Cahoon and Turner 1989; Boumans and Day 1994; Cahoon 1994). Furthermore, the greater accretion deficit and lower marsh elevation relative to sea level observed in managed marshes may explain why target water levels are seldom if ever reached. In contrast, some weir-managed marshes in the Chenier Plain region of Louisiana have not shown additional marsh loss or changes in accretion (Nyman et al. 1990a; Foret 1997). This different response to management by marshes in the Chenier Plain may be attributed to differences in substrate and local hydrology, both of which influence the effectiveness of weirs. However, Cahoon (1990) showed that managed marshes in both the Chenier Plain and Mississippi River Delta regions of Louisiana have significantly lower vertical accretion than nearby unmanaged marshes.

The exclusion of tidal energy from the system may also contribute to degradation of the project area (Fig. 3). Seasonally flooded cypress swamps show higher productivity than swamps that are continuously flooded, drained, or those with a slow rate of water flow (Mitsch and Ewel 1979; Conner and Day 1982). In addition, impoundment and levee construction have been linked to lower net primary productivity in wetland trees (Megonigal et al. 1997). Furthermore, restoring the natural hydrology of wetlands by removing levees has been shown to have a positive influence (Trepagnier et al. 1995).

An increase in the mean water level over the marsh, such as has occurred at the Falgout Canal Protection project, causes a significant decrease in plant aboveground biomass due to reduced soil redox (McKee and Mendelssohn 1989). The combination of increased submergence and increased salinity caused a greater decrease in plant growth when these factors occurred together (Webb and Mendelssohn 1996). Organic matter makes up a large portion of marsh soils, so a decrease in plant production can also cause a decrease in soil organic matter content (Nyman et al. 1990b). Low primary productivity can create a feedback loop in which a reduction in organic matter inputs to the marsh soil leads to inadequate marsh accretion, which leads to still lower primary productivity through a further increase in flooding stress (Nyman et al. 1993). These potential changes in marsh elevation were not considered in the management design of the Falgout Canal project.

Marsh management at Falgout Canal using weirs and flapgates is not meeting target water level goals. Under this management regime, the project area experiences fewer short flooding events and more long flooding events even though the absolute time the managed marsh is flooded is not significantly altered relative to areas outside the project. Although the weirs did statistically significantly decrease the salinity in the project area, the absolute difference of 0.09 ppt over three years may not be biologically significant. This minimal reduction in mean salinity inside the project area likely does not offset the concomitant negative impact of the increase in the depth and duration of flooding events over the surrounding marsh.

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Effect of 3-D Seismic Vehicular Traffic on Marsh Soil Elevations in a Southwest Louisiana Coastal Marsh

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ABSTRACT: Louisiana's Coastal Zone is an environmentally sensitive area that is considered one of the world's transition zones which is an area between high land and the Gulf of Mexico. Seismic activity has increased in Louisiana wetlands during the 1990's and both industry and governmental agencies are striving to minimize impacts associated with this development activity. Geco-Prakla conducted a 3-D seismic survey on Rockefeller Refuge from June 18, 1996 to July 1, 1997. We evaluated the effect of vehicular traffic, which included marsh buggies and airboats, on marsh soil elevations. Thirteen, 30.5 m transect lines were established perpendicular to vehicular traffic routes. Six transects were located in Unit 14, which is an intermediate marsh semi-impoundment, and seven transects were established in a brackish Tidal Marsh. Forty elevation points, which included disturbed and undisturbed marsh, were surveyed along each transect line to determine if vehicular traffic caused marsh soil compaction along traffic routes. Disturbed soil elevations were not different from undisturbed soil elevations for all transects, Unit 14, and the Tidal Marsh. Marsh buggy and airboat traffic did not appear to cause significant marsh soil compaction during this study.

Introduction

Louisiana's Coastal Zone is an environmentally sensitive area that is considered one of the world's transition zones which is an area between high land and the Gulf of Mexico (Hulme 1996). The systematic supervision of seismograph exploration in Louisiana to protect natural resources of the state was established in 1939 by the Department of Conservation, State of Louisiana. Program control was transferred to the Department of Wildlife and Fisheries in 1944 and continues today (Department of Wildlife and Fisheries 1946).

The Louisiana Department of Wildlife and Fisheries Fur and Refuge Division manages over 202,500 wetland hectares in coastal Louisiana. Thirteen 3-D seismic projects have been completed on these lands. Experience gained from these projects has provided the basis for development of regulations to minimize impacts associated with seismic activities. Geco-Prakla conducted a 3-D seismic survey on the 34,020 hectare Rockefeller Refuge, Cameron and Vermilion Parishes, Louisiana from June 18, 1996 to July 1, 1997 (Fig. 1).

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The area is considered "one of the most important wildlife areas in the United States" (Joanen et al.1969), therefore Louisiana Department of Wildlife and Fisheries personnel developed a special permit to minimize environmental impacts associated with the project. The seismic prospect encompassed 167 square miles and used a slanted source line pattern with a 1980 foot receiver line spacing " to reach the survey's geological goal and minimize marsh traffic" (Hulme 1996).

Concern over the possible deleterious effects of off-road vehicles to wetland ecosystems in the Florida Everglades began to develop in the 1970's. Duever et al. (1981) documented the effects of airboats, all-terrain cycles, and tracked vehicles on South Florida wetland habitats. Nidecker et al. (1993) developed a monitoring program to assess the impacts of 3-D seismic activity on wildlife and wetlands at Sabine National Wildlife Refuge, Cameron Parish, Louisiana. Nidecker et al. (1993) found that soil compaction from marsh buggy travel varied from 0.3 inches to 2.0 inches at eight sampling stations. One damp marsh area in the project area experienced soil compaction of six inches ; but "determining the impact on the soil matrix from seismic activity is a complex issue and one that was beyond the scope of this study" (Nidecker et al. 1993). We designed this study to quantify the effects of airboats and marsh buggies used for seismic exploration on marsh soil elevations within a semi-impounded and tidal marsh at Rockefeller Refuge.

Study Location

We conducted this study at Rockefeller Refuge in Unit 14 and a tidal marsh south of Unit 14 in Vermilion Parish, Louisiana (Fig. 1). Unit 14 is a 972 hectare semi-impoundment with both forced drainage and gravity flow water control capabilities. The area is characterized as intermediate marsh with major vegetative species being wiregrass (*Spartina patens*), bulltongue (*Sagittaria lancifolia*), and bullwhip (*Scirpus californicus*) (Chabreck and Linscombe 1988). The area south of Unit 14 is characterized as a brackish tidal marsh with major vegetative species being wiregrass and saltgrass (*Distichlis spicata*) (Chabreck and Linscombe 1988).

Unit 14 soils are classified as Banker series and consist of poorly drained, slowly permeable soils that formed in fluid clayey coastal alluvium and organic sediments in brackish marsh (United States Department of Agriculture 1996). Scatlake soils, which occur in the tidal marsh south of Unit 14, are similar to the Banker series, but are more saline than Banker soils (United States Department of Agriculture 1996).

Materials and Methods

Unit 14 and the Tidal Marsh south of Unit 14 were selected for this study to compare the effects of vehicular traffic on marsh soil elevations within two different marsh types. The traffic route along receiver line number 586, running north/south thru Unit 14 and the Tidal Marsh south of Unit 14, was chosen for this study. Geco-Prakla requested permission to conduct an experimental equipment test on the line after conventional 3-D seismic data was collected, and a complete traffic history of the line was available. Two additional traffic routes, running east/west within Unit 14 and the Tidal Marsh were also established for the study (Fig. 1). Vehicles used during the seismic survey and experimental test included a 4.0 m by 8.2 m marsh buggy mounted with a vibra-ram and numerous car engine powered airboats. Marsh soil elevation data were collected from February 21, 1997 to March 7, 1997 after all vehicular traffic activity was completed. Thirteen, 30.5 m transect lines running perpendicular to traffic routes were established. Six lines were located in Unit 14, while seven lines were located in the Tidal Marsh (Fig. 1). Transect line 1 in Unit 14 was not used for the study due to technical difficulties encountered during data collection. Transect lines extended 15.2 m on each side of a permanent center pole. Elevations at fifteen sampling points with 30.5 cm spacings and five sampling points with 305.0 cm spacings were recorded on each side of the center pole (Fig. 2). Cross-sectional vehicular traffic disturbance distances were determined visually on each side of the center pole by extending a tape measure from the center pole along the transect line to a point where compressed vegetation ended and undisturbed vegetation began.

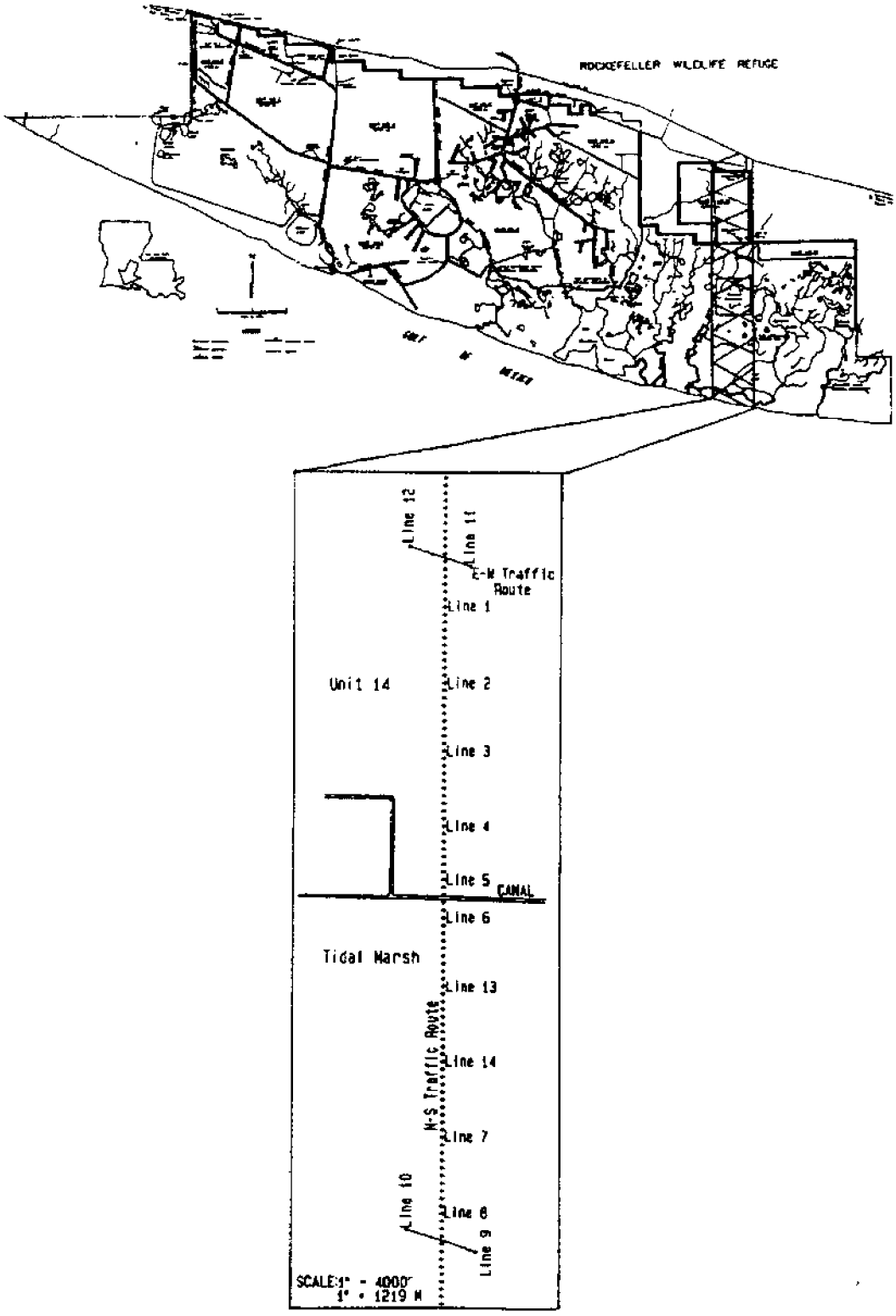

Fig. 1. Location of Rockefeller Refuge, Cameron and Vermilion Parishes, Louisiana and locations of Unit 14, the Tidal Marsh, vehicular traffic routes, and transect lines.

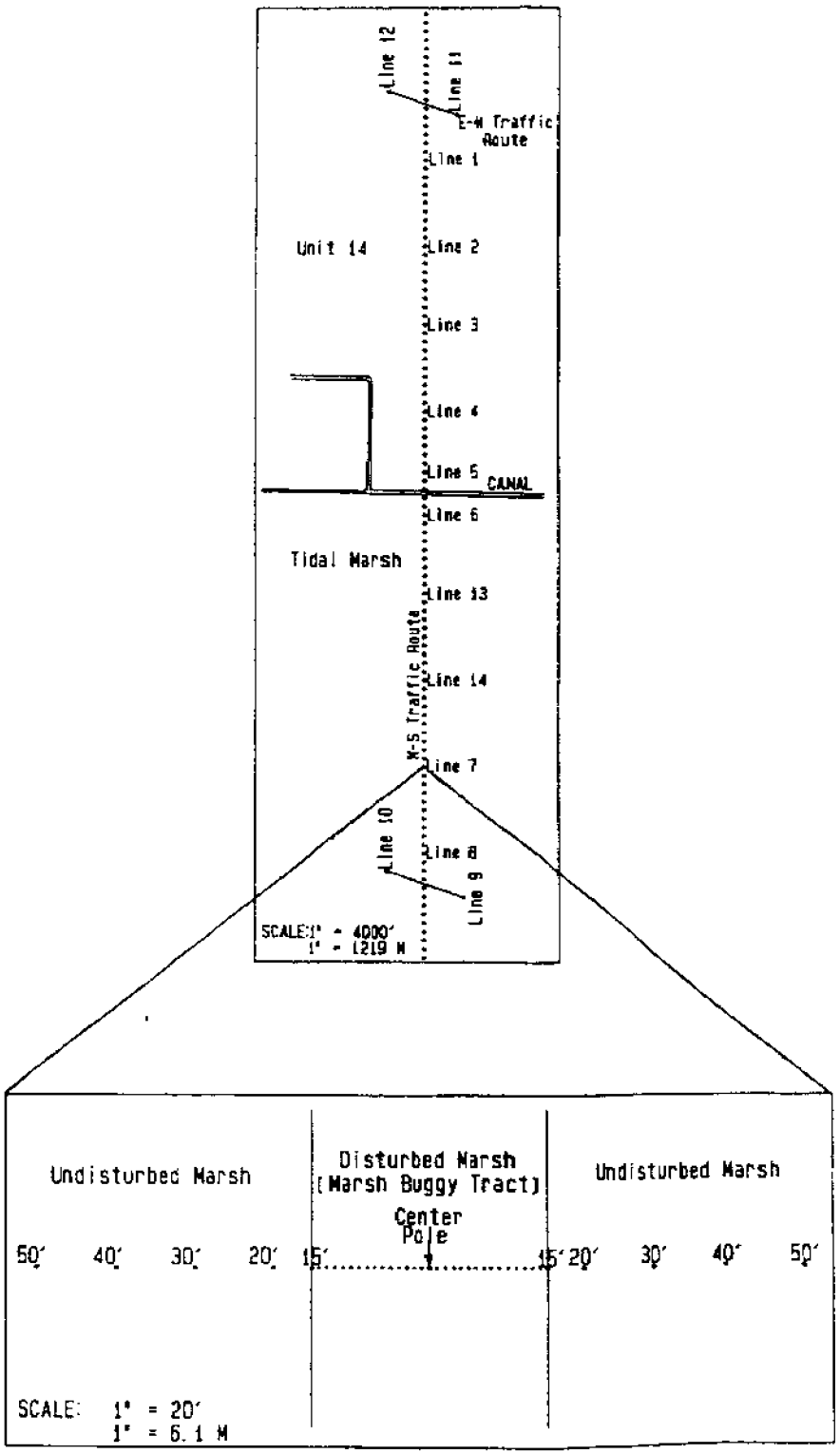

Fig. 2. Location and cross-sectional view of transect lines used to study the effects of 3-D seismic vehicular traffic on marsh soil elevations, Rockefeller Refuge, Cameron and Vermilion Parishes, Louisiana.

Water level data was collected between the first and tenth of each month by Rockefeller Refuge personnel. Staff gauges are located in Unit 14 and the Tidal Marsh. Average marsh level (mud line) equals a reading of 1.0 (30.5 cm) on staff gauges.

An EAGL-I electronic level mounted on a flathead tripod and CR-16-I leveling rod equipped with a LS-4 detector was used to collect elevation data at sampling points. The first reading on each transect line was taken at the center pole. The leveling rod was inserted through marsh grass to the mud line. Grass and vegetative debris was cleared by hand, and the soil surface was also identified by hand. Care was taken to minimize soil disturbance at sampling points. The LS-4 detector was moved along the leveling rod producing an audio signal and liquid crystal display beam when the elevation value was reached on the leveling rod.

Results

Differences in marsh soil elevations between disturbed and undisturbed marsh (subplot effect) and transect location (whole-plot effect) Unit 14 and Tidal Marsh were evaluated using a split-plot analysis of variance mixed model in Proc Mixed of SAS (1990). Least squares means were compared at an alpha = 0.05 level.

Vehicular traffic activity was variable along traffic routes (Table 1). The width of disturbed areas along each traffic route was variable (Table 2) and ranged from 5.5 m at transect line 8 to 14.9 m at transect line 2. The average disturbed area for all transect lines was 10.3 m.

Water levels within Unit 14 ranged from greater than 30.5 cm below marsh level in June, 1996 to 59.1 cm above marsh level in November, 1996. Water levels within the Tidal Marsh varied from 4.9 cm above marsh level in February, 1997 to 64.6 cm above marsh level in October, 1996 (Table 3.).

There was no interaction between location and disturbance (F=2.15, 11df, P=0.1706). Disturbed soil elevations were not different from undisturbed soil elevations for all transects (F=0.63, 11df, P=0.4430). There was no difference in soil elevations between Unit 14 and the Tidal Marsh (F=0.01, 11df, P=0.9357).

Mean soil elevations were 1.18 cm, 0.61 cm, and 1.75 cm below the center pole soil elevation reading for disturbed sampling points and 0.66 cm, 1.05 cm, and 0.27 cm below the center pole soil elevation reading for undisturbed sampling points for all transects, Unit 14, and the Tidal Marsh, respectively (Table 4).

Table 1. Number of airboat and marsh buggy passes documented during the Geco-Prakla, 3-D seismic survey at Rockefeller Refuge, Cameron and Vermilion Parishes, Louisiana, 1996-1997.

		Location			
		Unit 14		Tidal Marsh	
Line	Year	Airboat	Marsh Buggy	Airboat	Marsh Buggy
Receiver 586	1996	5	3	12	1
	1997	1	1	1	1
Total		6	4	13	2
East/West	1997	6	4	13	2

Table 2. Disturbance widths of traffic routes recorded from transect lines established in Unit 14 and the Tidal Marsh following the Geko-Prakla 3-D seismic survey at Rockefeller Refuge, Cameron and Vermilion Parishes, Louisiana, 1996-1997.

Location	Transect	Width (m)	Mean (m)
Unit 14	2	14.94	
	3	12.80	
	4	10.67	12.55
	5	12.50	
	11	12.19	
	12	12.19	
Tidal Marsh	6	5.79	
	13	12.19	
	14	10.36	8.06
	7	6.10	
	8	5.49	
	9	7.62	
	10	8.84	
		Overall Mean = 10.3	

Table 3. Water Levels from Unit 14 and the Tidal Marsh during the Geko-Prakla 3-D seismic survey at Rockefeller Refuge, Cameron and Vermilion Parishes, Louisiana, 1996-1997.

Month	Year	Location	
		Unit 14	Tidal Marsh
		Water levels (cm)[a]	
June	1996	Dry[b]	50.0
July	1996	Dry[b]	45.1
August	1996	Dry[b]	51.8
September	1996	50.6	50.6
October	1996	69.5	95.1
November	1996	89.6	62.8
December	1996	84.7	52.4
January	1997	68.3	39.6
February	1997	61.6	35.4

[a] 30.5 cm = Marsh Level
[b] >30.5 cm = Below Marsh Level

Discussion

Vehicular traffic activity varied along traffic routes due to varied marsh water levels during the 3-D seismic survey. Dry marsh conditions within Unit 14 caused marsh buggy use to increase, while water levels above average marsh level in the Tidal Marsh allowed increased airboat use. Louisiana Department of Wildlife and Fisheries 3-D seismic survey guidelines are flexible, but encourage limited marsh buggy and airboat passes to minimize environmental impacts. Seismic companies are encouraged to limit marsh buggy passes to one and airboat passes to 10 on traffic routes. This goal was not achievable during this seismic survey due to summer drought conditions and the necessity to perform an additional equipment test after the seismic survey was completed in the study area.

Mean soil elevation readings were lower than center pole soil elevation readings for all transects. Vehicular traffic on both sides of the center pole may have caused a slightly elevated ridge at the center pole, similar to ridges formed by vehicular traffic along test lanes in various South Florida marsh types (Duever et al.1981).

The greatest mean elevation value (1.75 cm) occurred in the disturbed area within the tidal marsh. Although no significant difference occurred between disturbed and undisturbed soil elevations within the Tidal marsh we observed signs of visual vegetative damage in disturbed areas of transects 13 and 14. Vegetative damage assessment was outside the scope of this study. Chabreck (1994) evaluated 25 sites traversed by marsh buggies and/or airboats during the 3-D seismic program conducted by Plains Resources, Inc. at Sabine National Wildlife Refuge, Hackberry, Louisiana and found that four of 21 sites traversed by marsh buggies contained some evidence of rutting. Three sites contained one inch ruts and marsh damage was considered light. One

Table 4. Least squares means of soil elevation (cm) in disturbed and undisturbed marsh studied in Unit 14 and Tidal Marsh following the Geco-Prakla 3-D seismid survey at Rockefeller Refuge, Cameron and Vermilion Parishes, Louisiana, 1996-1997.

	Condition				
Sites	Disturbed	Undisturbed	F	df	P
Unit 14	-0.608	-1.046	0.19	11	0.6722
Tidal Marsh	-1.745	-0.270	3.16	11	0.1033
All Transects	-1.177	-0.658	0.63	11	0.443

site contained four inch ruts and marsh damage was considered moderate. Chabreck (1994) did not statistically evaluate differences between soil elevations in disturbed versus undisturbed marsh, but stated "ruts that are one inch deep should not restrict plant growth." Chabreck (1994) also found that three of four rutted sites showed signs of plant regrowth. The fourth site had been burned and no plant regrowth was noted in the marsh buggy trail or adjacent undisturbed site.

Marsh buggy and airboat traffic did not appear to cause significant soil compaction during this study. Results presented by Duever et al. (1981) were similar. Airboats and track vehicles evaluated in various South Florida wetland habitats did not cause soil compaction. Rutting from vehicular traffic was considered the most severe soil impact, but rut depths decreased following the tests and were shallow or almost undetectable one year after treatments.

Summary

Geco-Prakla conducted a 3-D seismic survey on Rockefeller Refuge from June 18, 1996 to July 1, 1997. We evaluated the effect of vehicular traffic, which included airboats and marsh buggies, on marsh soil elevations. Vehicular disturbance was evaluated along 13, 30.5 m transect lines running perpendicular to traffic routes. An electronic level and leveling rod equipped with a detector were used to collect elevation data. Disturbed soil elevations were not different from undisturbed soil elevations for all transects, Unit 14, and the Tidal Marsh. Marsh buggy and airboat traffic did not appear to cause significant marsh soil compaction during this study.

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Effects of 3-D Seismic on Vegetation of a Southwest Louisiana Coastal Marsh

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ABSTRACT: Integrating oil exploration and development with wetland protection and restoration is an important facet of public and private land management in coastal Louisiana. The use of 3-D seismic in the coastal zone is relatively new, and is currently the method of choice for obtaining high quality geophysical data for locating oil and gas deposits. A better understanding of the environmental impacts associated with such activities is necessary to minimize and mitigate for any impacts. We studied vegetation response to traffic associated with a 3-D seismic operation at Rockefeller Refuge in Cameron Parish, Louisiana. The 3-D operation, which included 91 miles of lines within 9,197 acres of the refuge, occurred 2 January 1995, through 13 April 1995. Marsh buggy drills were used to place subsurface explosives, and airboats were used extensively to survey the lines, place and service recorders, service explosives, and monitor and clean all lines. We sampled vegetation before, one year after, and two years after the seismic activity on paired treatment and control sites in an effort to document effects on live stem densities, open water, and vegetative species changes. Sampling occurred on portions of two receiver lines and two shot lines within a management unit of intermediate coastal marsh. Results indicate no effect of seismic activity on total emergent vegetation, marshhay cordgrass (*Spartina patens*), or open water. However short term vegetative species changes occured via an increase in *Cyperus oderatus* the first year following the seismic activity.

Introduction

Coastal marsh managers strive to maintain a balance between resource utilization and protection, especially when their management activities are partially financed by resource development. The compatibility of oil exploration and utilization with wetland protection and restoration is an integral part of resource management plans on wildlife refuges and private marshlands in coastal Louisiana. Therefore, it is important to document impacts of oil exploration on marshes in an effort to keep negative impacts to a minimum.

Three-dimensional seismic surveys are currently the method of choice for obtaining geophysical data to locate subsurface oil and gas

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deposits (Ensminger et al. 1997). In coastal marshes this requires considerable vehicular traffic by airboats and marsh buggies in areas that normally experience little or no such traffic. Previous accounts of similar activities have met with mixed results. Ensminger et al. (1997) reported minimal detrimental impacts from a 3-D survey of Marsh Island State Wildlife Refuge (SWR) due to environmentally sensitive project planning, equipment, and methods, but impacts were not quantified. Chabreck (1994) assessed a seismic program at Sabine National Wildlife Refuge and found little long term impacts. Emergent vegetation and submerged aquatic vegetation (SAV) showed some initial evidence of impacts, but almost all study plots showed evidence of regrowth, even during a "non-growing season" (Chabreck 1994). Only sixteen percent of marsh buggy-traversed areas showed any damage, all of which was assumed to be recoverable after one growing season (Chabreck 1994). Three-D seismic work at the Mississippi River delta resulted in some evidence of vegetative species changes, although no detectable increase in open water areas resulted (M. Windham, unpubl. data). Polasek (1997) studied the effect of pipeline-excavating equipment (marsh buggies and airboats) on vegetation of a coastal marsh in the Texas Chenier Plain. Sixty-seven percent of his emergent vegetation plots failed to revegetate to 85% original coverage by the following year (Polasek 1997). The level of vehicular traffic that produced these results was not reported, thus comparability to 3-D seismic activity is unknown.

On January 2, 1995, Mobil Exploration and Producing U.S., Inc. commenced a seismic operation in southwest Louisiana that included 3,713 ha (9,197 ac) of Rockefeller SWR. Field work on the refuge was completed on April 13, 1995. This presented an opportunity to evaluate impacts of such an activity when implemented under close supervision of LDWF seismic section, refuge staff, and other Fur & Refuge Division staff responsible for oversight of mineral activities. The objectives of this study are to determine whether or not vehicular marsh traffic associated with 3-D seismic work causes changes in live stem densities of vegetation, open water areas, or vegetative species composition.

Study Area & Methods

Rockefeller SWR consists of approximately 30,400 hectares of coastal marsh in Cameron and Vermilion Parishes of Louisiana (Wicker 1983). Marsh types range from saltmarsh along the 42.6 km of coastline, to freshmarsh impoundments near the northern boundary along the Grand Chenier ridge (Wicker 1983). This study was conducted in Unit 6, a 21,700ha semi-impoundment of mostly intermediate marsh dominated by marshhay cordgrass (*Spartina patens*). Unregulated fresh-water inflow from the north into Unit 6 occurs via the Superior Canal that hydrologically connects the Mermentau Basin to the refuge. Water control along the southern levee of the unit consists of two sets of large radial-arm gates and two sets of aluminum flapgated structures with variable crest risers. A 40 ft radial arm lock system at the northwest of the unit is also used to control water and boat passage. Management of the unit is for stable water levels that produce high quality alligator habitat, and waterfowl food in the form of SAV. Prescribed burning is also used to prevent excessive fuel build-up and promote the growth of earlier successional plant species.

The seismic survey utilized a checkerboard pattern of receiver and source lines at 805.2-m intervals. Marsh buggy drills were used to place subsurface explosives, and single-engine airboats were used to survey the lines, place and service recorders, service explosives, and monitor and clean all lines. Two-pound charges were placed at a depth of 6.1 m at intervals of 50.3 m on source lines. Marsh buggies were only used on shot lines, and were limited to 1 pass except where special permission was granted by the refuge supervisor. Estimates of vehicular traffic were obtained from daily activity logs of each airboat and marsh buggy. Therefore we obtained conservative estimates of the number of passes by each vehicle type, for each line. Coordinated environmentally sensitive planning between all involved parties occurred via pre-project meetings, weekly meetings, and other coordination as deemed necessary.

Sampling occurred on portions of two receiver lines and two shot lines that crossed the unit. The

four lines - 2 north-south receiver lines and 2 east-west shot lines - were chosen because they were approximately equally distributed lattitudinally and longitudinally within the unit.

We sampled vegetation with the line-intercept method described by Chabreck et al. (1960). Two hundred twenty sampling stations were evenly spaced along receiver and shot lines at 76.2-m intervals with the following exceptions: 1) half-mile line sections which crossed Superior Canal were not sampled because visibility prohibited determining line location, and 2) stations which occurred totally in open water were moved to the nearest point along the line with emergent vegetation, from which the next 76.2-m interval was measured. Live vegetation by species and open water was estimated to the nearest 0.1-ft (3.0 cm) at each station. We made visual estimates of the portion of a 5-foot (1.52-m) line comprised of live vegetation by species and open water, with the remainder comprised of dead stems, bare ground, and space. Open water was defined as an area void of emergent vegetation, but can include SAV. For each sampling station on a receiver or shot line, a corresponding control station was established approximately 30 m from the line in a similar habitat type. Wooden stake markers (5 cm*5 cm*3 m) were placed 3.66 m due south (source lines) or west (receiver lines) from sampling stations.

Initial data were collected 10-18 November, 1994, 6-7 weeks prior to the seismic activity. Thereafter, data were collected by the same observer annually for two years 13-21 November. All data were collected and analyzed in English units, so results are reported as such.

Data were logratio transformed (Aitchison 1986) to better accomodate analysis as a continuous proportion. Values of 0 or 5.0 were perturbed slightly to 0.01 and 4.99, respectively, to accomodate the transformation. Values for total emergent vegetation, marshhay cordgrass, *Cyperus oderatus*, and open water were each analyzed as a response variable in a separate Before/After/Control/Impact (BACI) model (Green 1979, Stewart-Oaten et al. 1986) using PROC MIXED in SAS (SAS Institute, Inc. 1997). Models included effects of treatment (seismic activity or control), line (1-4), time (before, one year after, or two years after), and all interactions. The treatment-by-time (BACI) interaction indicates the effect of an impact. All means are reported as least-squares means. We performed pairwise comparisons using Tukey's procedure (Steel et al. 1997). Means are detransformed for reporting. An alpha of 0.05 was used to determine significance.

When P-values for the BACI interaction were marginal (0.05 - 0.10), we also evaluated those data with a logistic analysis in PROC GENMOD in which the response is a binomial count with 50 possible values (0.1-5.0 in increments of 0.1). Wald statistics and associated probabilities were then used to determine significance.

As part of routine refuge management activities, the study area was prescribe burned in the fall of 1994. Most of the study area was burned 20-28 November 1994, after annual vegetative sampling. A small southeast portion of the study area was burned 14 October 1994, five weeks prior to annual vegetative sampling

Results

Seventy-nine pairs of sampling stations were used in analyses. The remainder were discarded for one or more of the following reasons: 1) seismic activity did not actually impact the treatment sample station, 2) we were unable to relocate markers after prescribed burns, or 3) they were unintentionally impacted by subsequent seismic activities.

For total emergent vegetation (TEV), the BACI interaction was non-significant ($F = 1.07$, $df = 2, 520$, $P = 0.3243$) (Table 1). Treatment, time, and the time-by-line interaction were significant effects (Table 1). The trend of higher control values, both before and after the treatment, is evident in the least-squares means (Table 2).

For marshhay cordgrass, the BACI interaction was non-significant ($F = 1.90$, $df = 2, 520$, $P = 0.1510$) (Table 3). Treatment and the time-by-line interaction were significant effects (Table 3). As with TEV, the trend of higher control values for all years is evident (Table 4).

Table 1. Results of BACI model for effect of 3-D seismic activity on total emergent vegetation in Unit 6 of Rockefeller State Wildlife Refuge, Cameron Parish, Louisiana.

Source	NDF	DDF	F	Prob > F
Treatment	1	520	13.73	0.0002
Time	2	520	5.17	0.0060
Treatment*Time (BACI)	2	520	1.07	0.3428
Line	3	104	0.32	0.8097
Treatment*Line	3	520	1.82	0.1433
Time*Line	6	520	4.29	0.0003
Treatment*Time*Line	6	520	0.49	0.8195

For *Cyperus oderatus*, the BACI interaction was significant ($F = 4.69$, $df = 2, 520$, $P = 0.0096$) (Table 5). *Cyperus oderatus* values were higher at treatment stations the first year following the treatment than at any other combination of time and treatment (Table 6). Treatment, time, line, and the time-by-line interaction were also significant ($P \leq 0.0216$).

For open water, the BACI interaction was marginally significant ($F = 2.76$, $df = 2, 416$, $P = 0.0647$) (Table 7). Logistic analysis indicated a non-significant Wald statistic for the BACI interaction ($X^2 = 3.3454$, $df = 2$, $P = 0.1877$). Tukey's pairwise comparisons of the least-squares means revealed the value for the treatment stations the first year after the treatment was higher than the value for the control stations that year, but values the second year following the treatment did not differ (Table 8). The treatment effect and the time-by-line interaction were also significant (Table 7).

The east and west receiver lines were subjected to 118 and 114 airboat passes, respectively. Servicing recorders accounted for 80-81% of passes on receiver lines. The north and south source lines were subjected to 45 and 30 airboat passes, respectively, in addition to a marsh buggy pass. A second marsh buggy pass occurred on a short (10%) section of the north shot line.

Table 2. Least-squares means[1] of total emergent vegetation values[2] by treatment[3] and time[4] in Unit 6 of Rockefeller State Wildlife Refuge, Cameron Parish, Louisiana.

Time	Treatment	Control
Before	1.7881 AB	1.9990 A
After 1 year	1.7097 AB	2.3357 A
After 2 years	1.2795 B	1.8302 AB

[1]Means with the same letter are not statistically different (Tukey's procedure)

[2]Expressed in feet, out of a 5 foot sampling line

[3]Subjected to 3-D seismic traffic or control

[4]Before, one year after, or two years after 3-D seismic activity

Discussion

Lack of significant BACI interactions indicates that the seismic activity had no effect on TEV, marshhay cordgrass, or open water (Tables 1, 3, and 7). Although tracks were evident immediately after the seismic operation, the first post-seismic

Table 3. Results of BACI model for effect of 3-D seismic activity on marshhay cordgrass (*Spartina patens*) in Unit 6 of Rockefeller State Wildlife Refuge, Cameron Parish, Louisiana.

Source	NDF	DDF	F	Prob > F
Treatment	1	520	21.43	0.0001
Time	2	520	1.32	0.2674
Treatment*Time (BACI)	2	520	1.90	0.1510
Line	3	104	0.60	0.6145
Treatment*Line	3	520	1.19	0.3111
Time*Line	6	520	5.32	0.0001
Treatment*Time*Line	6	520	0.78	0.5883

Table 4. Least-squares means[1] of marshhay cordgrass (*Spartina patens*) values[2] by treatment[3] and time[4] in Unit 6 of Rockefeller State Wildlife Refuge, Cameron Parish, Louisiana.

Time	Treatment	Control
Before	1.2385 AB	1.4502 A
After 1 year	0.9236 AB	1.5786 A
After 2 years	0.8673 B	1.4457 AB

[1]Means with the same letter are not statistically different (Tukey's procedure)

[2]Expressed in feet, out of a 5 foot sampling line

[3]Subjected to 3-D seismic traffic or control

[4]Before, one year after, or two years after 3-D seismic activity

sampling period occured after a full growing season. Chabreck (1994) found evidence of regrowth of sprouts after marsh buggy tracks from 15 October - 9 January within the same year. The management burn after the first post-seismic sampling period could also have influenced recovery of the study area. Any vegetation that could have been compacted to prevent recovery was removed by the fire.

The significant BACI interaction in the *Cyperus oderatus* model indicates that the seismic activity did change the relative occurrence of that species (Table 5). We chose a posteriori to evaluate this species because it typically responds to disturbance, and appeared to perfectly delineate the survey lines in some areas the first year after the survey. The data reflects that it did increase on treatment plots 190 times over pre-seismic conditions, while the increase on control plots was only about 2-fold (Table 6). The effect was only short-term, as control and treatment values were not different the following year (Table 6).

The significant time-by-line interaction in every model indicates that the natural vegetative changes that occured did not change the same way for every line (Tables 1, 3, 5, and 7). Part of the reason we chose the 4 lines is because we expected natural variation even within the same management unit. To some extent, each line represents a different micro-habitat of subtle elevation, hydrologic, and edaphic differences within the Unit 6 intermediate marsh. It is therefore not surprising that a time-by-line interaction is significant for each model.

Table 5. Results of BACI model for effect of 3-D seismic activity on *Cyperus oderatus* in Unit 6 of Rockefeller State Wildlife Refuge, Cameron Parish, Louisiana.

Source	NDF	DDF	F	Prob > F
Treatment	1	520	5.31	0.0216
Time	2	520	51.90	0.0001
Treatment*Time (BACI)	2	520	4.69	0.0096
Line	3	104	3.63	0.0154
Treatment*Line	3	520	1.84	0.1394
Time*Line	6	520	3.61	0.0016
Treatment*Time*Line	6	520	1.99	0.0654

Table 6. Least-squares means[1] of *Cyperus oderatus* values[2] by treatment[3] and time[4] in Unit 6 of Rockefeller State Wildlife Refuge, Cameron Parish, Louisiana.

Time	Treatment	Control
Before	0.0099 AB	0.0103 A
After 1 year	1.8781 AB	0.0187 A
After 2 years	0.0112 B	0.0107 AB

[1]Means with the same letter are not statistically different (Tukey's procedure)

[2]Expressed in feet, out of a 5 foot sampling line

[3]Subjected to 3-D seismic traffic or control

[4]Before, one year after, or two years after 3-D seismic activity

The significant treatment effect in all models indicates that there was a difference, regardless of seismic activity, between treatment and control sites. Though we attempted to match vegetation types within each pair, we unintentionally and consistently chose control sites with more dense vegetation (Tables 2 and 4). The significant time effect in the TEV and *Cyperus oderata* models also indicates that these variables changed over time, regardless of seismic activity. These facts, though unrelated to the primary objective of this study, point out the need for analyses that control for both natural temporal changes in marsh vegetation and researchers' inadequate attempts to subjectively identify paired controls.

This seismic operation represents both a worst-case and best-case scenario relative to other coastal 3-D seismic surveys. Technological advances have far surpassed what was available at the time of this survey. Recent 3-D surveys have employed radio telemetry for transmitting seismic signals and monitoring receiving equipment, thus greatly decreasing the need for airboat maintenance of signal recording equipment (Ensminger et al. 1997). Other recent improvements include enhanced communications, increased use of helicopters, and slant geometry line patterns that minimize the need for damaging right-angle turns by marsh buggies (Ensminger et al. 1997).

Conversely, this seismic survey at Rockefeller SWR probably represents a best-case scenario in terms of monitoring and communications. Seismic activities were carefully planned and closely monitored by Seismic Section and Fur & Refuge Division staff of the Louisiana Department of Wildlife and Fisheries. Two pre-planning meetings, weekly operations meetings, and a post-cleanup meeting of all involved parties (refuge staff. Seismic

Table 7. Results of BACI model for effect of 3-D seismic activity on open water in Unit 6 of Rockefeller State Wildlife Refuge, Cameron Parish, Louisiana.

Source	NDF	DDF	F	Prob > F
Treatment	1	520	6.55	0.0120
Time	2	520	0.84	0.4314
Treatment*Time (BACI)	2	520	2.76	0.0647 [1]
Line	3	104	0.14	0.9367
Treatment*Line	3	520	0.57	0.6368
Time*Line	6	520	4.30	0.0003
Treatment*Time*Line	6	520	1.32	0.2489

[1]Logistic analysis resulted in non-significant Wald's statiscis (X2 = 3.3454, 2 df, P = 0.1877).

Table 8. Least-squares means[1] of open water values[2] by treatment[3] and time[4] in Unit 6 of Rockefeller State Wildlife Refuge, Cameron Parish, Louisiana.

Time	Treatment	Control
Before	0.0306 AB	0.0316 A
After 1 year	0.0592 AB	0.0260 A
After 2 years	0.0448 B	0.0289 AB

[1]Means with the same letter are not statistically different (Tukey's procedure)

[2]Expressed in feet, out of a 5 foot sampling line

[3]Subjected to 3-D seismic traffic or control

[4]Before, one year after, or two years after 3-D seismic activity

Section staff, Mobil operations managers, and field foremen) greatly enhanced communications and avoided or quickly resolved any problems. Chabreck (1994) attributed the success of the seismic survey on Sabine NWR to careful planning of access routes and use of the most environmentally sensitive equipment available.

Access to the project area via Superior Canal also contributed to the success of the project. Superior Canal provided an extensive and central access route to virtually the entire study area, thus minimizing the need for excessive buggy and airboat traffic on the vegetated wetlands.

We emphasize that these results, though encouraging relative to vegetation, do not represent other potential effects on the ecosystem. For example, Knott et al. (1997) found that despite good vegetative recovery at a coastal marsh study site in South Carolina, pipeline construction virtually eliminated several invertebrate species. We suggest further study on seismic impacts not evaluated by this study, such as impacts of disturbance on migratory and resident wildlife.

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Taxation of Private Marshland: The Use Value Method Applied to Louisiana's Coast

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ABSTRACT: Public benefits, many of them offsite, of privately owned and managed marshlands are well documented. One cost of ownership, property taxes, was high enough in the view of some private marshland owners to prompt a legal challenge of the basis of Louisiana marshland taxation. The stakes were high regarding impacts to stewardship of the 80 percent of Louisiana's 1.4 million ha of coastal marsh privately owned. In response to the landowners' pending legal proceedings, the 1995 Louisiana Legislature passed Act 230. The Louisiana Tax Commission (LTC) was directed to determine uniform marshland use values, for ad valorem taxation purposes. Use value's basis is net income from surface use, not market value. To develop a use value table in accordance with Act 230, a survey research study design was implemented. Average use value was obtained by dividing the average net income from surface uses by the appropriate capitalization rate. The use values obtained from the 9.41 percent capitalization rate were: \$9.67 ha^{-1} (freshwater), \$3.72 ha^{-1} (brackish) and \$1.59 ha^{-1} (saltwater). LTC procedure provides that average assessed value be obtained by multiplying the use value by 10 percent. The study's resulting assessed value per acre was far below levels in use by LTC. The research revealed a dubious future for the use value approach to taxing Louisiana marshland. Landowners appear headed toward tax reductions. County property tax revenues, which help support many vital local services, will decrease significantly as a result. This could affect the relationship between landowners and communities striving to maintain and restore public benefits of marshland.

Introduction

Seventy-five percent of marshland in Louisiana is privately owned (McBride 1992). The marshland is subject to property taxation. The Louisiana constitution directs assessors to utilize one method of determining the value on which to apply millage. This method is use value. Market value, potential value and other alternatives are excluded. Thus, the constitution restricts taxation to income from surface uses. Well known off-site functions and values of marshland as well as mineral revenue cannot be included in tax estimation.

The Louisiana Tax Commission establishes marshland tax guidelines for assessors. From 1974 to 1995 the marshland guidelines were set without the rigor afforded to use value calculations for farm land and timber. Two marshland owners filed separate lawsuits claiming the lack of rigor resulted in higher taxes. In these cases landowners documented that average per acre surface revenues were below the rates traditionally used for marshland by the Louisiana Tax Commission (LTC). In response the Louisiana Legislature in 1995 passed Act 230 directing LTC to prepare a use value tax table for freshwater, brackish and saltwater

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marshland. LTC contracted with the Louisiana State University Agricultural Center, the same organization doing the use value estimates for agriculture.

History of Use Value

Use value for property tax assessment of land originated from encroaching urban and suburban property uses in traditional agricultural and forestry areas. Assessment of farmland near urban areas, on the basis of its market value resulted in taxes above that which could be supported by agricultural uses. As a result, strong incentives were created to shift the land to residential or industrial uses. Not all land influenced by urban sprawl could be quickly transferred to these higher valued uses. Further, many landowners wanted to continue to use their land for agricultural production.

Use value was conceptualized, developed and implemented for agriculture and forestry property owners being impacted by assessments based on high market value and to allow owners to continue agricultural use of land. Use value is normally determined on the basis of the estimated net income per acre for the specific class of land. Uniformly applied use value tables are then developed for each land class.

Survey Design and Methodology

Mail questionnaires and personal interview techniques were used to collect the data necessary to calculate use value as delineated in Act 230. The same procedure used by LTC to obtain use value for farmland and timberland was used in this survey.

Louisiana's 1.4 million ha of coastal marshlands are usually separated into four wetland categories, freshwater, intermediate, brackish and saltwater. Act 230 mandates that only three marshland classifications be identified: freshwater, saltwater and brackish marsh. After consultation with the Marshland Use Value Advisory Committee (MUVAC), it was decided by consensus that intermediate marsh be placed within the brackish marsh category.

To assist study participants identify these three marsh types the following descriptors were included in the questionnaire: (1) Fresh marsh - marshes with no salinity, dominated by typical freshwater plants such as water hyacinth, alligator weed and bull tongue; (2) Brackish marsh - marshes influenced by a mixture of fresh and salt water, dominated by plants such as wiregrass, three-corner grass and to a limited degree, bullwhip; (3) Salt marsh - tidally influenced marsh subject to high salinity levels with dominant plants being Spartina or smoothcord grass, wiregrass, black rush and saltgrass. The seven selected surface uses were alligator egg sales, commercial alligator harvest, aquaculture, fur trapping, hunting enterprises, recreational fishing and cattle grazing.

Coreil (1995) found 44 percent of marshland owners with tracts of less than 200 ha derived no revenue from surface use. Another 27 percent received revenue less that \$25 ha^{-1}. The MUVAC comprised of assessors and agency personnel recommended the data collection process include marshland owners with holdings of 400 ha or more. Each parish's respective assessor's office provided the names and mailing addresses of landowners meeting the criterion.

Content validity of the survey was pre-tested through a focus group consisting of members of the MUVAC and Louisiana Cooperative Extension Service personnel with experience in survey methodology and design. Mail survey and personal interviews produced data on 0.5 million ha.

Respondents were asked to classify land holdings into the three marshland classifications outlined previously. Distribution of marshland types included 25 freshwater marsh respondents, 22 brackish marsh respondents and 18 saltwater marsh respondents. Twenty-four percent of the respondents owned more than one marshland type.

Saltwater marsh recorded the highest total acreage (approximately 253 thousand ha) followed by freshwater marsh (165 thousand ha). Brackish marsh recorded the smallest acreage at 106 thousand ha. Saltwater marsh accounted for nearly half of the reported marsh area in the survey (48%).

Survey responses accounted for 37% of the total marshland in Louisiana. Eliminating publicly

held lands results in total private holdings of approximately 1 million ha. The study, therefore included 50 percent of the privately owned marshland in Louisiana.

Alligator hunting was the most prevalent revenue-earning surface use reported, accounting for one third of all survey responses (32%) followed by hunting enterprises (26%). When sales of alligator eggs are included, alligator related enterprises account for nearly one half (45%) of the revenue earning surface uses reported. In total, alligator and hunting enterprises account for nearly three-quarters (71%) of the revenues derived from marshland surface uses.

Alligator hunting is the primary surface use identified in freshwater marsh accounting for approximately one third, 31%, of the revenue earning surface uses. Other important freshwater marsh surface uses included hunting enterprises and alligator egg sales. In total these three surface uses accounted for 69% of the revenue sources in the freshwater marsh with alligator enterprises alone accounting for 48% of the total.

In brackish marsh, alligator enterprises were even more important in revenue generation. Alligator hunting and egg collecting accounted for 38% and 14%, respectively. Revenue from hunting enterprises comprised another 27% of surface revenue. In total these three uses accounted for 79% of the revenue sources for brackish marsh.

Saltwater marsh was the only category to not record alligator hunting as the primary revenue earning source. Hunting enterprises reported the highest surface use (30%) with alligator enterprises (eggs and hunting combined) accounting for 31% of the revenue.

Three conclusions can be drawn from the surface use data: (1) the importance of alligator enterprises among marshland owners, particularly for those owning freshwater and/or brackish marsh; (2) the importance of hunting enterprises in revenue generation; and (3) the apparent differences in use characteristics for saltwater marshes, i.e. alligator enterprises.

Range of Income

Freshwater marshland owners recorded the highest percent area with a positive income (41%) followed by brackish (21%) and saltwater (8%). The positive-income area for the brackish and saltwater marsh categories was roughly equivalent with 22 thousand ha of brackish marsh and 20 thousand ha of saltwater marsh reporting a positive return.

Gross revenue per ha are calculated by summing the marsh category revenues and dividing by the total marsh area. The average gross revenue per ha for freshwater marshland ($5.51) was eight times higher than for saltwater ($0.69).

The range of gross revenue per acre by marsh type is shown in Table 1. Gross revenue per ha ranges were calculated by dividing individual property income by the respondent's reported area for each specific marsh type.

Table 1. Range of gross revenue per hectare by marsh type.

	low	high	avg. income ha^{-1}
Freshwater Marsh	$0.32	$18.82	$5.51
Brackish Marsh	$0.12	$16.75	$4.17
Saltwater Marsh	$0.35	$6.42	$0.69

Net income per ha was calculated by summing the positive net income (profit) and dividing by the total marshland acreage. Freshwater marsh had the highest average net income at $0.91 ha^{-1}. Brackish marsh averaged $0.35 ha^{-1} with saltwater marsh reporting the lowest net income at $0.15 ha^{-1}. One of the primary reasons for the low per ha estimate is the fact that many marshland owners report either no income or losses from surface uses. In this study, 40% of the freshwater marsh respondents, 64% of the brackish respondents and 67% of the saltwater marsh respondents reported losses. Since the total marsh area is used in the calculation of the average net income per ha, the average net income per ha is considerably reduced by the large amount of area

reporting no net income. This is the procedure set by the use value procedures of LTC, not the researchers.

Calculation of Use Value

The average use value per ha was obtained by dividing the average net income by the appropriate capitalization rate. This rate used as a divisor is a key component to the use value. The lower the rate, the higher the use value. The rate is comprised of yields on U.S. government securities. Therefore, the calculation of the capitalization rate is not subject to manipulation. The rate prevailing at the time of the study as 9.41 percent.

The freshwater marsh net income level of \$0.91 ha^{-1} divided by the capitalization rate .094 (i.e. 9.41 percent) results in a use value of \$9.67 ha^{-1}. The brackish and saltwater marsh use values are \$3.72 and \$1.59 ha^{-1}, respectively. Millage for determining the tax due is applied to 10 percent of the LTC use value. The term applicable to these estimates is assessed value. The assessed value estimates are \$0.967 ha^{-1} fresh marsh, \$0.327 ha^{-1} brackish marsh and \$0.159 ha^{-1} salt marsh.

Conclusions

The research revealed a dubious future for the use value approach to taxing Louisiana's marshland. There is too much disparity between the LTC's undocumented assessed values and the study's findings of lower values for there to be rapid movement to lower assessed values. The LTC has since receiving the report acted to reduce the assessed values. This must be a slow process so that parish tax collections can be cushioned from the trend toward lower tax revenue from marshlands. For example, Louisiana voters in November 1998 passed by a wide margin a constitutional amendment permitting parishes to retain 50 percent more oil and gas severance taxes collected in their jurisdictions.

There are other alternatives not related to surface marshland use values that counties and the Louisiana legislature will have to evaluate. These include: 1) setting a minimum assessed value, 2) establishing financial budgets for key marshland uses, 3) increased federal and state oil and gas revenue sharing opportunities in coastal parishes and 4) reviewing the constitutional prohibition on a market value approach.

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COASTAL FLORA AND FAUNA

Effects of Estuarine Environmental Factor Gradients on the Tolerance and Physiology of the Southern Oyster Drill, *Stramonita* (= *Thais*) *haemastoma*; An Important Predator of the American Oyster, *Crassostrea virginica*

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ABSTRACT: The southern oyster drill, *Stramonita haemastoma*, is an important predator of the American oyster, *Crassostrea virginica*. *S. haemastoma* consists of two genetically-differentiated groups; *S. h. canaliculata* from the Gulf of Mexico and *S. h. floridana* from south Padre Island, Texas and the east coast of Florida. *S. h. canaliculata* from Caminada Pass, Louisiana exhibited maximum accessory reproductive tract size in April and July while the gonad-digestive gland complex reached maximum size in April, September and October. *S. h. canaliculata* was very tolerant of exposure to constant hypoxia (28 day LC_{50} of 5-19% oxygen saturation between 10 and 30‰ S at 10- 30°C). Depression of heat production to 9% of the normoxic rate occurred within hours. *S. h. canaliculata* tolerated diurnal variation in percent oxygen saturation from 100% to 25% over 28 days. The benthic stage of both subspecies was more euryhaline than indicated by conch line observations(> 15‰ S) with a 28 day LC_{50} of 3.5-7.1‰ S for *S. h. canaliculata* and 7.3‰ S for *S. h. floridana;* southern oyster drills were osmoconformers and excellent water volume regulators. Normal intracapsular development and hatching of larvae occurred at 20‰ S and above at 22 and 28°C. The average monthly water temperature ranged from 10 to 30°C. Temperature had a far greater effect on feeding rate and the energy budget than salinity. The percentage of snails attracted to oysters in a Y-maze increased from 15 to 25°C and from 10 to 25‰ S. Salinity affected crawling velocity at 20, 25, and 30°C. Snails ceased feeding at temperatures below 12.5°C and buried into the sediment where they were capable of acquiring energy from primary amines. The energy budget was depressed at 25 and 30°C compared with the energy budget at 20°C. *S. h. canaliculata* is extremely euryhaline, eurythermal and euryoxic.

Introduction

The Role of *Stramonita haemastoma* in Louisiana's Estuaries

The southern oyster drill is a significant predator of the American oyster, *Crassostrea virginica* (Gmelin) in the South Atlantic and Gulf States (Galtsoff 1964; Butler 1985). In Louisiana, *Stramonita (Thais) haemastoma* (Gray) is one of the oyster's worst enemies (Waldo 1957); the major refuge of the oyster is below 15‰S which is known as the conch line (St. Amant 1938, 1957). Seed oysters are obtained by dredging from low salinity state-managed seed grounds east of the Mississippi River and transported to private leases in higher salinity where they are allowed to fatten prior to marketing (Pollard 1973). Salt water intrusion has constricted the area of the seed oyster grounds. The distribution and density of *S. haemastoma* on the oyster seed grounds has been mapped and it is only found at salinities greater than 15‰S (Breithaupt and Dugas 1979).

The oyster drill appears to select prey in an optimal fashion; predator and prey size and density set important constraints on prey choice. *S. haemastoma* may severely prey on the bivalves *Ischadium recurvum* and *Rangia cuneata* at high

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salinities and limit those prey to low salinities in estuaries (Brown and Richardson 1987). Small oyster drills can only prey on small oysters (Garton 1986; Brown 1997). At wave exposed sites, 10 to 39% of the oyster drills are only capable of consuming oysters less than 50 g wet weight, the median size oyster in the intertidal reef, because of their smaller size (Brown 1997).

Distribution, Taxonomic Status, and Life History

The systematic status of Thaidid molluscs has been recently revised. The southern oyster drill has been placed in the genus *Stramonita* based on anatomical data (Kool 1987, 1989). Clench (1947) and others (Abbott 1974; Andrews 1971) considered *S. haemastoma canaliculata* (Gray 1839) and *S. haemastoma floridana* to be two subspecies of the type *S. h. haemastoma* Linne which occurs in the Mediterranean Sea and along the west African coast. Two genetically-differentiated groups of southern oyster drills were identified by allozyme variation at 18 electrophoretically-detected loci from nine collections in the southeastern United States and northern Gulf of Mexico at a genetic level that is characteristic of congeneric species in other taxa (Liu et al. 1991). *Stramonita haemastoma canaliculata* like snails were collected at six locations in the northern Gulf of Mexico; *S. h. floridana* like snails were collected at South Padre Island, Texas and Marineland and Ponce Inlet, Florida. Neither shell nor radular characters were useful in differentiating the two groups. The collection from Caminada Pass, Louisiana contained 96% *S. h. canaliculata* like, 2% *S. h. floridana* like, and 2% putative hybrid snails. The Ponce Inlet and Marineland, Florida collections also had representatives of both genetic groups and hybrid snails (Liu et al. 1991).

Stramonita haemastoma, like prosobranchs in general, has a life span that ranges from 1-20 years (Butler 1985). In the northern Gulf of Mexico, oyster drills attain sexual maturity at 8-12 months after metamorphosis from the veliger larval stage (Hoese et al. 1972). Growth rate appears to be a direct response to food consumption and ambient temperature (Butler 1985). Seasonal change in biochemical content is predominantly due to changes in the size of the female capsule-albumin gland complex and the gonad-digestive gland complex (Belisle and Stickle 1978). The capsule-albumin gland complex peaks in size in April and July, the testis-digestive gland in April and September, while the ovary-digestive gland reaches maximum size in April. Copulation has been observed from April until June. Egg capsule deposition occurs from six hours until sixty days after copulation (Roller and Stickle 1988). Egg capsule structure of *S. h. canaliculata* (Roller and Stickle 1988) and *S. h. floridana* (D'Asaro 1966) is similar. Each egg capsule of *S. h. canaliculata* contains 3246 ± 21 (SE) embryos embedded in a viscous albuminous fluid. Developmental patterns have been followed to hatching for *S. h. floridana* (D'Asaro 1966) and *S. h. canaliculata* (Roller and Stickle 1988). Teleplanic veliger larvae emerge from the egg capsule, spend more than 90 days in the plankton and can be dispersed over a large geographical range before settling and undergoing metamorphosis to snails (Scheltema 1978). The life cycle of *S. haemastoma* is depicted in Fig. 1.

Bioenergetic Considerations

Research on the bioenergetics of the southern oyster drill has yielded several important principles that are also operative along intensity gradients of environmental factors in other species of marine invertebrate carnivores. The primary component of scope for growth to vary as a function of environmental factor gradients is energy consumption (C) or absorption (Ab) (Stickle 1985). The ratio of energy consumption or absorption to energy expenditure due to respiration (R) and excretion (U) was found to be 5.1:1 for salinity and temperature gradients (Stickle 1985) and 3.1:1 for a hypoxia gradient (Das and Stickle,1993). Stickle (1985) reported similar C or Ab:(R + U) ratios for the gastropods *Thais lapillus* across salinity and temperature gradients (Stickle and Bayne 1987) and *Thais lima* exposed to the water soluble fraction (WSF) of Cook inlet crude oil (Stickle et al. 1984). Both *T. lapillus* and *T. lima* prey on bivalves by boring a hole in the shell and using an eversible proboscis with a radula to ingest prey tissue. In contrast, *Stramonita h. canaliculata* secretes a toxin from the hypobranchial gland which paralyzes the

Life cycle of *Stramonita haemastoma*

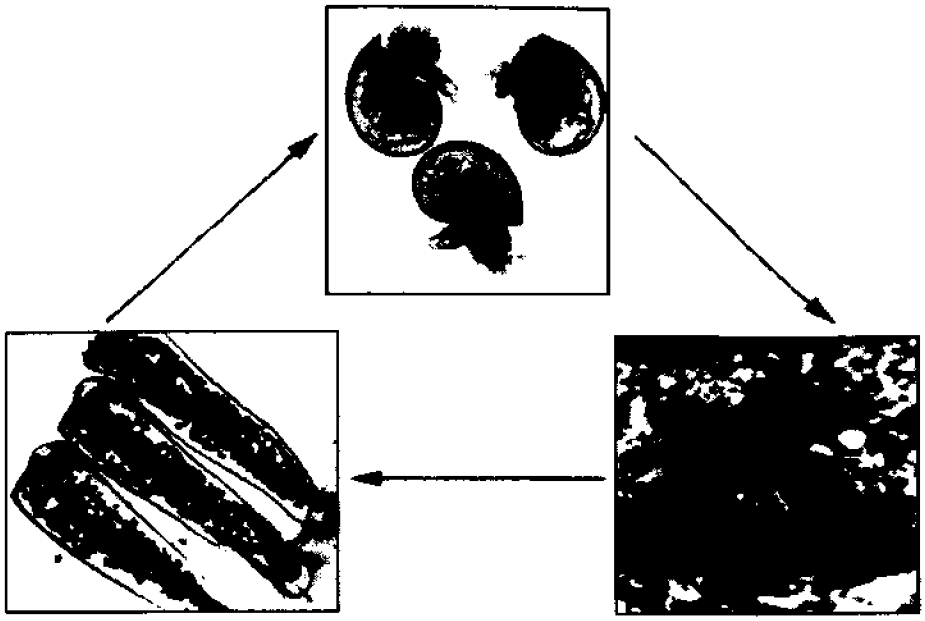

Fig. 1. Life history of *Stramonita haemastoma canaliculata*. Snails deposit egg capsules on hard substrata. Teleplanic veliger larvae emerge into the water column and can spend over 90 days in the plankton before settlement and metamorphosis to juvenile snails.

adductor muscle and allows individual or groups of oyster drills to ingest tissue through the gaping valves (Roller et al. 1995) with the proboscis-radular complex (Roller et al. 1984). Small oyster drills cannot feed efficiently on large oysters (Garton 1986), perhaps because they cannot produce a sufficient amount of toxin.

Like other marine invertebrate carnivores, a higher percentage of *S. h. canaliculata* cease feeding altogether, rather than exhibiting a reduced feeding rate at the extremes of the zone of capacity adaptation along environmental factor gradients (Stickle 1985). This phenomenon was also found to exist in the sea star *Leptasterias hexactis* and the gastropod *Thais lamellosa* along a salinity gradient and in *Thais lima* along a WSF of Cook Inlet crude oil gradient (Stickle 1985). Multiple regression equations were generated for feeding data in an experimental salinity-temperature matrix for both *Stramonita h. canaliculata* and for *Thais lapillus* (Stickle 1985). More than 55% of the southern oyster drills ceased feeding below 7‰ S and 13°C. Isolated *S. haemastoma* were used in the salinity-temperature matrix study reported in Stickle (1985). Likewise, more than 90% of *T. lapillus* ceased feeding during the three week experiment at salinities below 25‰ S and 8.3°C. Group feeding did not occur in groups of *T. lapillus*, probably because the snails fed through the bore hole.

The first study to document a positive relationship between scope for growth (an animal's energy budget) and the number of heterozygous loci (heterozygosity) in feeding marine invertebrates was conducted on *S. h. canaliculata* (Garton 1984). Six of 25 enzyme loci assayed were found to be polymorphic and a significant positive relationship existed between total heterozygosity and scope for growth of snails adapted to all three experimental salinities; 7.5, 20, and 35‰ S at 21°C. The increased scope for growth of heterozygotes was correlated with significantly greater feeding rates on juvenile *Crassostrea virginica*. The number of

heterozygous loci per snail explained 16% of the variance of scope for growth at 7.5‰ S, 14% at 20‰ S, and 15% at 35‰ S. In contrast, only two of 12 enzyme loci assayed in *Thais lamellosa*, which produces crawl away juveniles, were found to be polymorphic (Garton and Stickle 1985). Evidence of heterozygote superiority in scope for growth of *T. lamellosa* was present; however, there was insufficient genetic variability for correlation of scope for growth with heterozygosity.

The primary energy source, from a biochemical perspective, of a number of species of carnivorous marine invertebrates exposed to environmental factor gradients is protein (Stickle 1985). O:N ratios were determined in order to assess whether protein or carbohydrate/lipid was the primary substrate catabolized. O:N ratios in the 2-8 range indicate that protein is the substrate catabolized but O:N ratios as high as 200-300 indicate that carbohydrate and/or lipid is the primary catabolic substrate. Within the zone of capacity adaptation, O:N ratios ranged from 3.1-28.5 in the five species of marine invertebrate carnivores studied indicating that protein was the primary substrate catabolized (Stickle 1985).

Variation in Dissolved Oxygen, Salinity and Temperature

Variability of dissolved oxygen, salinity and water temperature has been well documented in coastal Louisiana. Seasonal development of hypoxic water masses offshore has been well characterized and is related to fresh water and nutrient input from the Mississippi River (Rabalais et al. 1986a,b). Average diurnal variation in dissolved oxygen at Port Fourchon and Caminada Pass, Louisiana over a 14-d recording period in June 1992 ranged from 25-175 Torr or 16-122% oxygen saturation (unpublished). Likewise, tidal variation in salinity has been documented by a salinometer probe placed 0.33 m off the bottom for 791 days at a location 14.5 km north of Barataria Pass (Hewatt 1951). Daily amplitude of salinity variation varied as follows: 0-5‰ S per day = 41%; 5-10‰ S per day = 35%; 10-15‰ S per day = 15%; 15-20‰ S per day = 7%; 20-25‰ S per day = 2%; and 25-30‰ S per day = 1% of the days. Average monthly change in salinity at Barataria Pass only varied from 17.5‰ S in May-June to 26.5‰ S in November (Barrett 1971). Average monthly water temperature at Barataria Pass varied between 12.8°C in December-January and 30.4°C in August (Barrett 1971). Monthly mean salinity and water temperature was obtained from only three to five discrete determinations per month.

The objective of this review is to summarize our knowledge of the effects of variation in dissolved oxygen, salinity and water temperature on the southern oyster drill *S. haemastoma*.

Effects of Dissolved Oxygen on *Stramonita haemastoma*

The southern oyster drill is tolerant of hypoxia with 28-day LC_{50} values ranging from 8-29 Torr O_2 (5- 19% oxygen saturation) (Kapper and Stickle 1987; Stickle et al. 1989). In contrast, the 28-day LC_{50} of juvenile *Callinectes similis* is 43 Torr O_2 (28% saturation) and that of *C. sapidus* is 106 Torr O_2 (68% saturation) (Das and Stickle 1993). Oxyregulatory ability of *S. h. canaliculata* varied inversely with temperature and did not exhibit acclimation after 28-day exposure to 53 Torr O_2 (34% saturation). Activity ratios of the four pyruvate oxidoreductase enzymes identified in *Stramonita h. canaliculata*, alanopine dehydrogenase, lactate dehydrogenase, strombine dehydrogenase, and octopine dehydrogenase were not induced in foot tissue after 28-day exposure to 15-100 Torr O_2. 100 Torr = 64% oxygen saturation (Kapper and Stickle 1987). Adenylate energy charge (AEC = the molar concentration of ATP ± 1/2 the molar concentration of ADP divided by the sum of the molar concentration of ATP + ADP + AMP) and arginine phosphate concentrations, the primary muscle high energy source of phosphate bond energy, were reduced at 10- 32% saturation compared to snails maintained at 64 - 100% saturation (Kapper and Stickle 1987). Metabolic heat production, a measure of total metabolic rate, declined linearly with severity of hypoxia as did energy consumed and absorbed from predation on the bivalve *Ischadium recurvum* (Das and Stickle 1993). In contrast, energy consumption and absorption rate of juvenile *Callinectes sapidus* did not decline at hypoxic levels

down to 47% oxygen saturation, but was significantly lower at 32% saturation (Das and Stickle 1993). Consequently, Scope for Growth of the southern oyster drill varied directly with oxygen partial pressure down to 16% oxygen saturation, while surviving juvenile blue crabs did not exhibit a significant decline in Scope for Growth at reduced oxygen tension. *S. h. canaliculata* is very tolerant of exposure to constant hypoxia (28 day LC_{50} of 5-19% oxygen saturation between 10 and 30‰ S at 10- 30°C). Depression of heat production to 9% of the normoxic rate occurs within hours. *S. h. canaliculata* tolerates diurnal variation in percent oxygen saturation from 100% to 25% over 28 days (Das and Stickle 1993). Ingestion of food requires more time in the southern oyster drill than in the blue crab.

Exposure of oyster drills to anoxia has shed additional light on adaptations to hypoxia. The anoxia LT_{50} varied from 10 to 28 days and was not affected by salinity or temperature in the 10-30°C and 10-30‰ S matrix design (Stickle et al. 1989). Metabolic heat flux decreased to 9% of the normoxic rate within hours of transfer to anoxia compared to a decrease to only 32% of the normoxic rate in Callinectes sapidus exposed to 25% oxygen saturation (Stickle et al. 1989). Adenylate energy charge declined from 0.79 to 0.58 after one day exposure to anoxia, and arginine phosphate concentration declined by 95% (Kapper and Stickle 1987). Adenylate energy charge cycled at the expense of arginine phosphate with cyclic exposure to anoxia from normoxia.

Effects of Salinity on *Stramonita h. canaliculata*

The effects of salinity on the tolerance and capacity adaptations of snails and the intracapsular development of *Stramonita h. canaliculata* has received considerable attention. The low salinity 28-day LC_{50} of *Stramonita h. canaliculata* collected at four sites was 3.5-7.1‰ S while that of one population of *Stramonita h. floridana* was 7.3‰ S (Fig. 2; Liu, 1990). The high salinity 28-day LC_{50} of the four populations of *S. h. canaliculata* was 44.4-51.5‰ S whereas that of one population of *S. h. floridana* was 54.8‰ S. Seasonal salinity ranges at the collection sites were from 2-40‰ S for *S. h. canaliculata* and was 37‰ S at Marineland, Florida (Liu 1990). *S. h. canaliculata* is an excellent euryhaline osmoconforming species which exhibits excellent volume regulation in direct transfer (Kapper et al. 1985) and fluctuating salinity

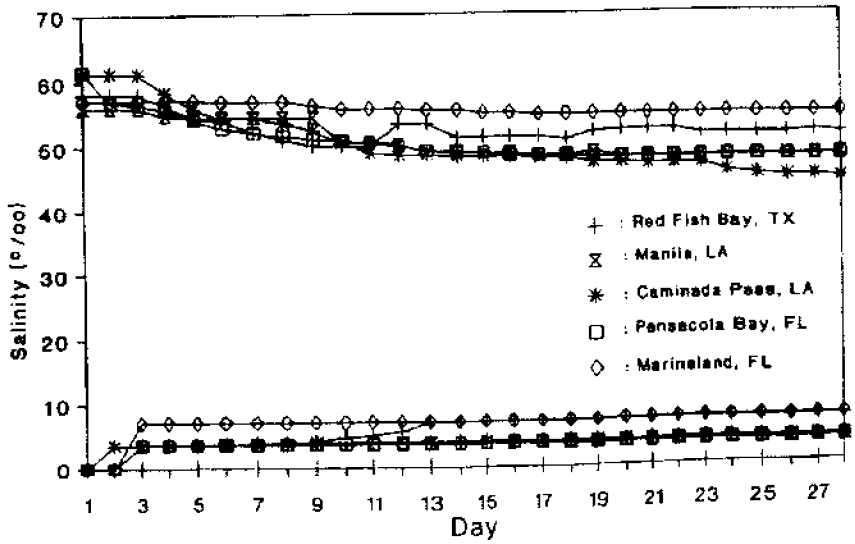

Fig. 2. The upper and lower 28-day LC_{50} of *Stramonita haemastoma canaliculata* and *S. h. floridana* as a function of salinity. *S. h. floridana* was collected at Marineland, Florida and *S. h. canaliculata* was collected at the remaining four sites.

experimental designs (Stickle and Howey 1975; Hildreth and Stickle 1980). Alanine and glycine are the principal intracellular osmotic effectors to be regulated during both low and high salinity transfers but aspartate, taurine, proline and glutamate each contributed more that 5% of the free amino acid pool. The extracellular fluid compartment came into rapid equilibrium with the ambient seawater in both direct transfer and fluctuating salinity experiments (Stickle and Howey 1975; Hildreth and Stickle 1980).

The rate of oxygen consumption of 30‰ S acclimated southern oyster drills was significantly higher than that of snails acclimated to10‰ S (Findley et al. 1978), but there was no significant difference in metabolic heat flux of acclimated oyster drills at 30 and 10‰ S indicating an increased rate of anaerobic heat production at 10‰ S under both normoxic and anoxic conditions (Liu et al. 1990). After transfer of snails from 10 to 30‰ S under normoxia, heat flux was depressed initially to 38% of the pre-transfer rate, but the rate recovered after 14 h to a rate that was 56% higher than the normoxic rate at 10‰ S. After transfer of snails from 30 to 10‰ S under normoxic conditions the standardized heat flux rate decreased to 28% of the pre-transfer rate, followed by a 20-h period of recovery to the control rate. With the exception of a 20 - 10 - 20 ‰ S semi-diurnal pattern of fluctuating salinity, the rate of oxygen consumption of oyster drills declined as salinity fluctuated in either direction from the acclimation salinity and increased as ambient salinity returned to the acclimation salinity (Findley et al. 1978). This pattern of oxygen consumption occurred during semi-diurnal and diurnal patterns of fluctuating salinity.

Neither the rate of predation nor the Scope for Growth of *S. h. canaliculata* preying on small *Crassostrea virginica* varied significantly with salinity between 10 and 35‰ S at 10, 20, 25 or 30°C but there was a direct relationship between Scope for Growth or feeding rate and salinity at 15°C. Feeding rate and Scope for Growth of the southern oyster drill was significantly lower at 7.5‰ S at all temperatures than occurred between 20 and 35‰ S (Stickle 1985). Benthic *S. h. canaliculata* are extremely euryhaline with respect to exposure to constant salinity gradients at 10‰ S and higher and at temperatures > 15°C; in addition the feeding rate is not significantly different in oyster drills exposed to diurnal patterns of 30 –10 -30‰ S or 10 - 30 -10‰ S fluctuating salinity for 21 days than in snails feeding at 10 or 30‰ S (Garton and Stickle 1980).

Tolerance and developmental rates of the intracapsular stages of the southern oyster drill are sensitive to environmental salinity (Roller and Stickle 1989). Snails deposited egg capsules when acclimated to 12.5, 15, 20, 25, 30, and 35‰ S at 22 and 28°C. Mortality at 12.5‰ S and the presence of abnormal veligers at 12.5 and 15 ‰ S limit larval tolerance to above 15‰ S. No data exist on the effects of salinity on the LC_{50} and developmental rates of the teleplanic larvae.

Effects of Temperature on *Stramonita h. canaliculata*

Seawater temperature strongly affects the resistance and capacity adaptations of southern oyster drills. The 28-day LC_{50} of summer acclimatized *Stramonita h. canaliculata* from Port Fourchon, Louisiana is 35.7°C (unpublished) and represents their summer resistance adaptation.

Energy expenditure rate of *Stramonita h. canaliculata* due to oxygen consumption and nitrogen excretion (principally NH_4^+) was significantly higher at 25°C than at 10, 15, 20, or 30°C (Stickle 1985). Temperature gradients accounted for more variance (ANOVA) in total maintenance costs (47.1%) than salinity gradients (11.7%; Stickle 1985).

Southern oyster drills preying on oysters as individuals enter cold torpor below 12.5°C at 20‰ S (Garton and Stickle 1980), cease feeding on juvenile oysters (Garton and Stickle 1980) or exhibit minimal feeding on juvenile oysters (Stickle 1985) and enter the sediment over the winter months (unpublished). Snails can actively uptake free amino acids from pore water under anoxia at relatively high primary amine concentrations (216 –364 μM) during the winter months.

Oyster drills exhibit evidence of heat stress when they are maintained and individually fed juvenile oysters (4-5 cm length) at 25 and 30°C throughout the salinity range (except at 20‰ S and 30°C) (Stickle 1985). A much higher percentage of variation in Scope for Growth was accounted for by variation in seawater temperature (35.4%) than in salinity (12.7%) in the experimental matrix of 10- 30°C and 7.5- 35‰ S (Stickle 1985).

Attraction of the southern oyster drill to oysters in a Y-maze is sensitive to temperature and salinity (Fig. 3). The percentage of oyster drills attracted to oysters increased directly with temperature from 15 to 25°C and was lower at 15 and 30°C. Attraction to prey was positively correlated with salinity up to 25‰ S. Crawling velocity was minimal at 15°C and was affected by ambient salinity at 20, 25, and 30°C. Mean crawling velocities ranged from 0.07 cm min^{-1} at 15°C and 10‰ S to 18.0 cm min^{-1} at 25°C and 20‰ S.

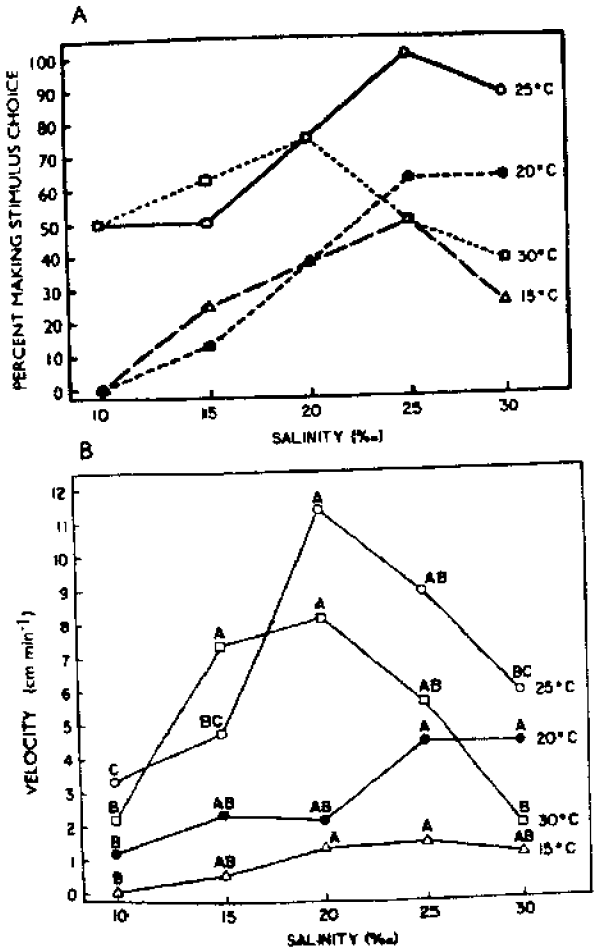

Fig. 3. Chemoattraction of *Stramonita haemastoma canaliculata* to 3-4 cm *Crassostrea virginica* as a function of temperature and salinity in a Y-maze. A) The percent of oyster drills attracted to oysters; B) oyster drill crawling velocity. Crawling velocities sharing a common letter at different salinities at a temperature are not significantly different from each other (Waller-Duncan Test).

Group feeding on large oysters is maximum at 30°C but the feeding rate of individual snails on large oysters as a function of water temperature is unknown. There is no difference in the feeding rate of individuals and groups of oyster drills on large oysters at 20°C and 20‰ S (Garton and Stickle 1980). Both prey size and the number of snails preying on large oysters influence the temperature at which the highest feeding rate occurs. Group foraging occurs frequently in the estuary but snails feeding in groups on small and large oyster prey exhibited reduced per capita feeding rates (Brown and Alexander 1994). However, the fraction of oyster flesh consumed was 21% higher in snails feeding in groups resulting in similar per capita oyster tissue consumption and snail growth rates for solitary and group predators.

Egg capsule deposition occurs during the late spring (April-May) and ceases as water temperature reaches 30°C (unpublished). Capsules were deposited only at 20 and 28°C in southern oyster drills that were acclimated to 20, 22, 28, and 30°C and salinities from 10-40‰ S (Roller and Stickle 1989). Optimal intracapsular survival and development to hatching occurred at 20 and 25‰ S at 22 and 28°C. Survival to hatching was higher and development faster at 28°C than at 22°C for all salinities (Q_{10} values for hatching = 1.60-2.64). Weight specific oxygen consumption and ammonium excretion rates and the O:N ratios were higher in post-hatching veligers at 28°C than at 22°C for all salinities tested (Roller and Stickle, 1989). The benthic stage of *Stramonita h. canaliculata* is very euryoxic with respect to constant and diurnal exposure to hypoxic water throughout its seasonal temperature range. Snails of both subspecies are also euryhaline (LC_{50} range = 3.5-54.8‰ S), but intracapsular developmental stages only develop normally above 15‰ S at 22 and 28°C. Seasonal variation in water temperature is the primary

environmental factor that controls the feeding and energy budget of the southern oyster drill, *Stramonita haemastoma canaliculata* which preys on the American oyster, *Crassostrea virginica*.

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Agricultural Wetlands and the Conservation of Colonial Wading Birds in Louisiana

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ABSTRACT: Increasing conversion of natural wetlands to agricultural lands has affected many wildlife species that depend on wetland habitats. But the effects of such land conversion have not been entirely negative. The rapidly growing crayfish aquaculture industry in southern Louisiana has provided valuable foraging habitat for many species of birds, including ducks, shorebirds, and colonial wading birds (herons, egrets, and ibises). Colonial wading bird populations in Louisiana have increased dramatically during the past twenty years, and these changes are significantly correlated with annual changes in the total acreage devoted to crayfish aquaculture. The increasing use of crayfish farms by wading birds poses a challenging conservation problem, and may eventually result in significant economic impact.

Introduction

Wetlands throughout the southern states have been drained and converted to agricultural and other uses at a rapid rate (Tiner 1984). The conversion of natural habitats to agricultural or industrial use often have negative effects on wildlife (Vitousek et al. 1997). The creation of artificial wetlands through the conversion of natural habitat to irrigated fields and aquaculture ponds, on the other hand, has had a beneficial impact on populations of wading birds that forage in these new habitats (Fleury and Sherry 1995). The rapid growth of crayfish aquaculture has provided wading birds with over 50,000 hectares of high quality supplemental foraging habitat (Fleury 1996). Crayfish farms have been a primary factor in the exponential increase of populations of several species of colonial wading birds in Louisiana (Fleury and Sherry 1995).

These expanding populations have created new problems for the conservation and management of colonial wading birds. Wading birds respond opportunistically to large-scale changes in habitat quality (Frederick et al. 1996, Fleury 1996, Fasola and Ruiz 1996). White Ibis, for example, may relocate geographically in response to environmental stress (Frederick et al. 1996). Such regional population shifts may put predatory birds in direct economic conflict with aquaculture farms. This paper reviews the response of colonial wading birds to the growth of the crayfish industry, and considers the conservation and management problems these expanding populations may pose.

Waterbirds and Aquaculture

For as long as people have farmed fish, birds have bothered fish farmers. Both humans and

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colonial waterbirds are at the top of the aquatic food chain, and even share similar tastes in prey species. This historic conflict has become more important in the past twenty years as aquaculture has developed into a major global industry (Huner 1994, Price and Nickum 1995). Many species of birds prey on aquaculture crops. The USDA's Southern Regional Aquaculture Center has identified several "problem species" on aquaculture farms in southern states (Stickley 1990): Double-crested Cormorant (*Phalacrocorax auritus*), Anhinga (*Anhinga anhinga*), American White Pelican (*Pelecanus erythrorhynchos*), Hooded Merganser (*Lophodytes cucullatus*), Great Blue Heron (*Ardea herodias*), Great Egret (*Casmerodius albus*), Snowy Egret (*Egretta thula*), Tricolored Heron (*Egretta tricolor*), Green-backed Heron (*Butorides striatus*), Black-crowned Night Heron (*Nycticorax nycticorax*), Yellow-crowned Night-Heron (*Nycticorax violaceus*), White Ibis (*Eudocimus albus*), Belted Kingfisher (*Ceryle alcyon*), Osprey (*Pandion haliaetus*), and various species of gulls and terns (Laridae). Wading birds are problem species at both crayfish and catfish farms.

Wading Birds and Crayfish Farms

Colonial wading birds regularly forage in Louisiana's crayfish farm ponds. These new foraging habitats have significantly contributed to the rapid increase of wading bird populations in southern Louisiana (Huner 1994, 1995, Fleury 1994, Fleury and Sherry 1995, Fleury 1996). Crayfish are an important food item for several species of wading birds. Winter populations of these species correlate significantly with the annual harvest of swamp crayfish, and these same species populations correlate more significantly with annual changes in land area devoted to crayfish aquaculture (Fleury and Sherry 1995). These correlations are based on annual Christmas Bird Count data (CBC), which estimate winter bird populations, but Breeding Bird Survey (BBS) data, gathered in Summer, have corroborated these trends. Methodological problems with such data are beyond the scope of this paper, but Fleury and Sherry (1995) review some of the major concerns. Fleury (1996) suggests that CBC data are especially appropriate for colonial species that regularly and predictably aggregate. Furthermore, BBS data suggested that these population changes are part of a broader pattern of changes in populations of Gulf Coast wading birds in response to human-induced changes in the natural landscape (Fleury 1996, Frederick et al. 1996).

Louisiana has an abundance of healthy wetland habitats (Williams and Chabrek 1986), and the wetland sites surveyed by Fleury (1996) had a diverse prey base and high levels of wading bird foraging success. Crayfish ponds, however, provide a foraging environment that is equivalent to or better than natural foraging habitats (Fleury 1996). Because individual farmers tend to manage their crops in the same way from year to year, farm pond hydrology is also more predictable in space and time.

Farm pond management follows the natural annual hydrological pattern in swamps, except that farm ponds are out of phase with the natural cycle. The wild crayfish harvest depends on the drainage of spring rains into southern wetland habitats. High spring water levels of the right temperature bring crayfish out of their burrows. Crayfish farms, however, are reflooded in Fall, which brings farm crayfish out of their burrows several months before the wild harvest begins. Here we describe a ricefield-crayfish pond management plan that is typical of about 60% of crayfish farms (Huner 1990, 1994, 1995).

Wading birds are first attracted to ponds in the Fall, when reflooding drives numerous terrestrial arthropods onto emergent vegetation. Flocks disperse when this resource has been exploited, and reassemble in 4-8 weeks when young crayfish reach about 5 cm in length, and the biomass of insects, tadpoles and other aquatic prey increases (Huner pers. obs.). Harvesting begins as early as November, and continues into May. Annual pond production averages 600-700 lbs/acre, with good ponds producing up to 1,500 lbs/acre (Huner unpub. data). Rice-crayfish ponds are usually drained in April and May, and permanent crayfish ponds are drained in May and June. Ponds may then be seeded for rice or other crops. By the end of the crayfish season, most unharvested mature crayfish have already burrowed to await the Fall reflooding.

Drawdown (i.e., draining ponds) marks the end of the crayfish harvest, and the period of major influx of wading birds into farm ponds (Martin 1979, Fleury 1994). Pond drawdowns strand high densities of prey in shallow pools. Drawdowns usually fall within the annual breeding season of wading birds, and supply a rich diet for both adult and juvenile birds (Fleury and Sherry 1995, Fleury 1996). The coincidence of maximum prey availability during drawdown and wading bird nesting cycles may be a primary factor linking the expansion of crayfish farm ponds with wading bird reproductive success (Fleury 1996).

Control of Waterbird Predation

Crayfish farmers have increasingly voiced concern about the rapidly growing flocks of predatory birds in their farm ponds (Fleury 1994). The measurement of damage to aquaculture farms from avian predation was recently highlighted as an important goal of wading bird research (Price and Nickum 1995). This is especially true in Louisiana, where the commercial crayfish crop is currently estimated at $35 million per year. The potential problem may have become more serious in the last several years, as winter populations of the most important predatory species on crayfish farms (Fleury 1996, Huner 1994), have continued to increase (Fig. 1).

Non-lethal control methods such as audio and visual frightening techniques, overstocking, distraction ponds, relocating birds, and covering ponds with nets have met with limited success (Draulans 1987, Littauer 1990a, 1990b, Boyd 1991, Cezilly 1992). Furthermore, these non-lethal techniques are expensive, with annual costs of more

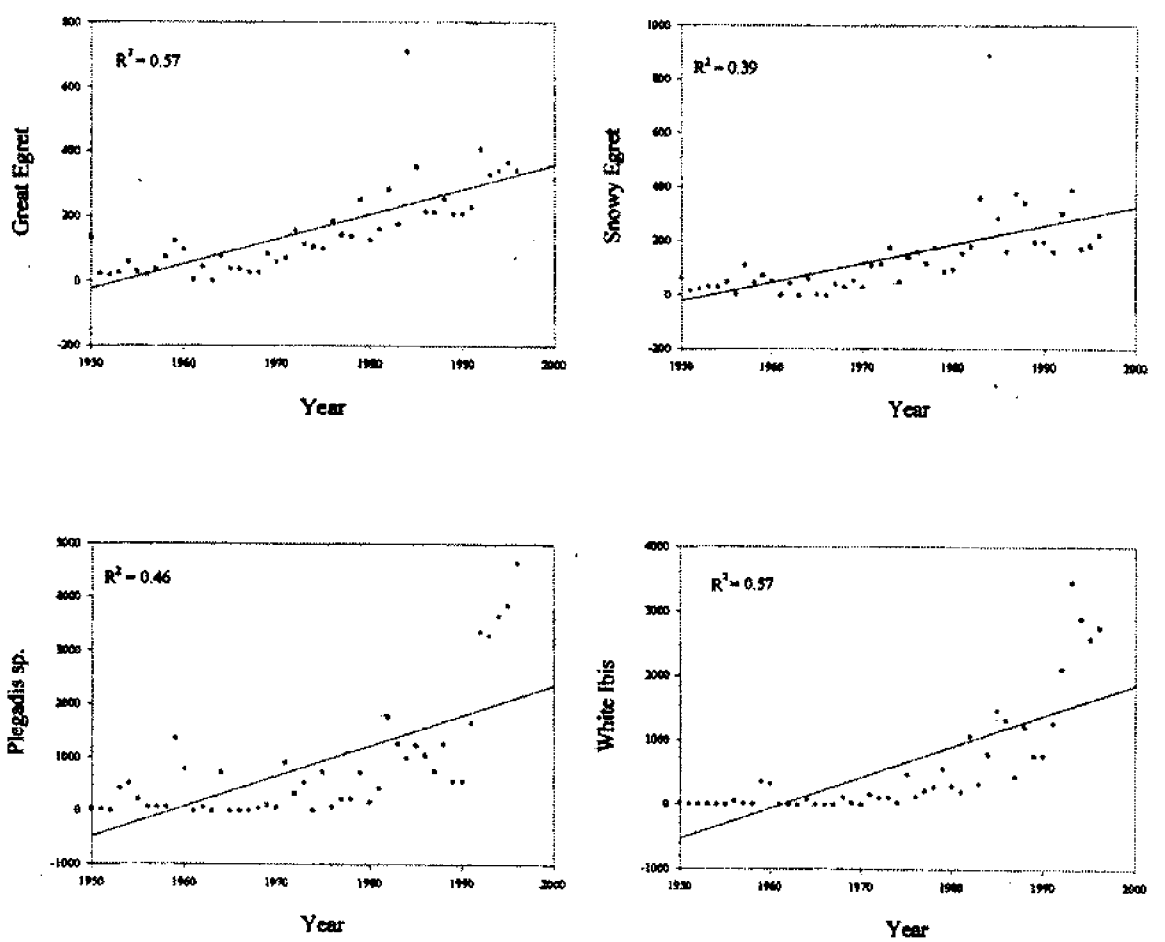

Fig. 1. Linear regression models of winter wading bird populations, 1950-1997, for selected species recorded in Louisiana Christmas Bird Counts, number observed per count. (Note: *Plegadis sp.* includes two species of dark ibis, the Glossy Ibis [*Plegadis falcinellus*] and the White-faced Ibis [*Plegadis chihi*]).

than $13,000 per five-month season (Littauer 1990b, Draulans 1987). Audio frightening methods include pyrotechnics such as screamer shells and "bird bangers" fired from a .22 caliber blank pistol, waterproof firecrackers, butane cannons, electronic noise generators, playback of recorded distress calls, and the firing of live ammunition, the use of which is obviously limited by the farmer's immediate environment. Visual frightening techniques include the use of scarecrows, regular vehicle patrols, hawk silhouette kites, helium balloons, and radio-controlled miniature aircraft (Littauer 1990b). Such frightening techniques are only effective for a short time before birds acclimate to the disturbance.

Netting ponds may be effective, although many birds quickly learn to go through or under the large mesh nets. But covering farm ponds with large-mesh nets requires a large initial investment and high maintenance costs. The only scientific study of the effectiveness of pond exclosure failed to exclude birds from the test ponds. Martin's (1979) attempts to exclude birds from test ponds were not successful, and further research is needed to establish the effectiveness of such techniques.

The pressure by some farmers for lethal control continues to mount with increasing bird populations, highlighting the need for further research on the ecological and economic impacts of these birds (Huner 1993). Although depredation permits for wading birds have been issued to Louisiana's catfish farmers, no USFWS permits for wading birds have been issued for crayfish farmers. U.S. Fish and Wildlife Service permits are only issued with the concurrence of state and local wildlife officials, and only after non-lethal control techniques have proven ineffective.

Lethal controls are already used against wading birds and cormorants on catfish farms, where the annual damage from water bird predation may exceed $2 million (Stickley et al. 1992, Glahn and Stickley 1995). Between 1965 and 1991, the Mississippi Delta catfish industry grew to over 50,000 ha of farm ponds. Along with this rapid increase in catfish farm acreage came a rapid increase in the numbers of cormorants, which flocked into catfish ponds by the tens of thousands (Glahn and Stickley 1995). The simultaneous increase of catfish pond acreage and cormorant populations suggests a cause-effect relationship that parallels the growth of Louisiana's wading bird populations (Weseloh et al. 1995, Jackson and Jackson 1995). Great Egrets and Great Blue Herons also forage along the edges of catfish ponds, and are generally viewed as serious agricultural pests. Several hundred wading birds have been killed each year on catfish farms, under U.S. Fish and Wildlife Service permits (Stickley 1990, Price and Nickum 1995, Littauer 1990a). Permits were issued to catfish farmers in Louisiana in 1990/91, for example, for 1,023 kills (however only 751 kills were reported; USFWS 1990/91 unpubl. data).

Economic Impact on Crayfish Farms

The impact of wading bird predation falls most heavily during the drawdown period (Martin 1979, Fleury 1996). None of the crayfish or fish available to wading birds during drawdown, however, have any significant commercial value. Most of the crayfish that carry the broodstock for next year's crop may have already burrowed before drawdown begins (Huner 1994). Bird predation could hurt farmers during drawdown, however, if most of the remaining crayfish at drawdown were females that had not yet bred and burrowed, but might still do so. If there were significantly more males than females remaining at drawdown, the economic impact of bird predation during drawdown would be negligible.

To test the hypothesis that most crayfish remaining at drawdown were male, we sexed crayfish sampled at wild and farm sites with a half-square meter drop trap, and compared observed frequencies with those expected from a 1:1 sex ratio. Farm pond samples taken before drawdown had significantly more females than males ($P<0.003$, $n=38$ traps). Crayfish in trap samples from farm ponds at drawdown ($n=65$ traps) and wild sites ($n=78$ traps), however, showed a balanced sex ratio, although wild samples contained a greater relative frequency of female crayfish. There was a significant shift in farm pond sex ratios between flooded and drawdown stages, with 71% female crayfish prior to drawdown, and 54% females at

drawdown. The missing females probably burrowed during the two to three weeks that typically separated pre-drawdown and drawdown samples. These crayfish are inaccessible to foraging birds. Further studies are needed on crayfish sex ratios in farm ponds, the timing of female burrowing, and the overlap of the crayfish life cycle with the annual cycle of the wading birds that use farm-ponds. These results suggest that depredation efforts should be focused on the period just prior to harvest to ameliorate any economic impact.

The effects of wading birds on the crayfish population during flooded stages, however, has not been as thoroughly examined. Martin (1979) found that flocks fed primarily on fish and other prey during flooded stages, and switched to eating primarily crayfish during drawdown. Only 9% of crayfish consumed by wading birds were of marketable size (7.5-12.5 cm long). Overall, less than 2% of the total commercial harvest was taken by wading birds (Martin and Hamilton 1985). Although some wading birds take large numbers of crayfish, others take more fish than crayfish, with negligible economic impact (Fleury 1996).

Anecdotal evidence, however, suggests the possibility of significant damage on some farms. Pond foraging by wading birds on Indigo Island showed a small peak following the Fall reflooding for each of two years censused (Martin 1979). Computer models developed at LSU suggest that predation following Fall reflooding could potentially damage the commercial crop (R. Romaire unpubl. data). Intense ibis predation at this time could produce losses of several hundred kilograms per hectare. Crayfish in the size range preferred for harvest normally mature in December-February, when prices are at their highest, which could increase the potential economic damage from post-flooding predation (R. Romaire unpubl data). Further research is needed to document the effects of wading bird predation, especially during the first several months of the crop cycle.

Wading birds may even improve the quality of the crayfish crop during flooded stages by thinning out overstocked ponds. High densities of crayfish can result in stunted crayfish that fail to reach marketable size (Avault et al. 1974, Huner and Barr 1991). An LSU farm pond with this problem yielded crayfish 2 cm or less at drawdown (Fleury 1996). Night-Herons and Little Egrets feeding on Israeli fish farms improved commercial stocks by feeding on predatory species, culling weak or injured fish, and reducing dense populations of fish fry (Ashkenazi and Yom-Tov 1996). The problem with relying on bird predation as a strategy for thinning crops, however, is like regulating a coal furnace. Once it gains momentum, it is difficult to control and nearly impossible to stop.

Wading Bird Predation and Crayfish Farm Management

The annual pattern of wading bird foraging, coupled with the crayfish life cycle, suggests that there are two periods when bird control efforts should be focused: when crayfish start to reach harvestable size after the Fall reflooding, and again in the weeks just before drawdown. If farmers concentrated their effort during these two periods of foraging activity, they could achieve the same results as seasonal control programs, but at greatly reduced cost.

Efforts to discourage foraging should also be species-specific. The White Ibis is the most important avian predator of pond-raised crayfish. Continued rapid increases in populations of this crayfish specialist might reach economically significant levels in the next decade (Fig.1; Fleury and Sherry 1995). Keeping pond water levels higher than usual until drawdown should discourage these short-legged foragers to some extent, but they have been seen foraging successfully in relatively deep water (T. Mikuska pers. comm., Fleury pers. obs.). Farms in the coastal prairie region of Louisiana are shallower (20-30 cm) than ponds in the alluvial valley region (30-45 cm), where all of the wading bird studies have been conducted. Coastal farms may thus be more susceptible to high levels of bird predation than alluvial ponds.

Drawdown is the most critical period for wading birds, because it coincides with their breeding season (Fleury and Sherry 1995). The minimal economic value of crayfish remaining in

the ponds at this time may not justify the economic effort of excluding birds from ponds. Artificial supplementation of breeding season diets, however, might eventually increase bird populations to the point where the economic effects become substantial during harvest season (Draulans 1987, Fleury 1996). One strategy that might keep bird populations from expanding further is to synchronize drawdowns, to minimize the number of high-density patches of prey that the birds can visit during the breeding season.

Well-managed crayfish farms not only benefit colonial wading birds, but also provide foraging habitat for a wide variety of other animals (Huner 1995). At USL's farm, for example, 176 species of birds have been seen, together with 26 species of mammals, 22 species of reptiles, 12 species of amphibians, and 26 species of fish (Huner, pers. comm.). Farmers could capitalize on their potential value by leasing ponds to duck hunters, promoting ecotourism during drawdown, and applying for state and federal support for farm ponds as important wildlife habitat. Due to the recent controversy over imported Chinese crayfish, Congress is now familiar with the nature and importance of the industry, and might be more inclined to support it in the form of farm subsidies or tax breaks for habitat management. Some crayfish farmers are already managing their impoundments for waterfowl during the off season, gaining up to $2,500 a year in additional income by leasing ponds to duck hunters (Nassar et al. 1991). When ponds are first flooded, in late September and early October, there is little available habitat for ducks and other game species, and farm ponds offer an excellent seasonal foraging habitat.

Successful control of wading bird predation on crayfish farms could significantly reduce wading bird populations in Louisiana. Several species of wading birds in Europe, for example, have come to rely on the irrigated rice fields that have largely replaced their natural foraging habitats (Fasola 1986, Fasola and Ruiz 1996, Fasola et al. 1996). A gradual shift to dry-land rice cultivation, in areas where wading birds have grown dependent on rice-field foraging, currently threatens several European species (Fasola and Ruiz 1996). Dry-land cultivation may reduce heron populations in Italy, Spain, and Greece by as much as 50%. Exclusion of wading birds from crayfish ponds could have similar effects on U.S. populations.

Human alterations of natural landscapes often have unpredictable consequences. The rise of the crayfish industry in Louisiana has inadvertently created prime foraging habitats for colonial wading birds, and the birds have responded opportunistically. Their rapid population growth has created a unique conservation problem for the industry that has helped create them. The future of the crayfish industry and the future of Louisiana's colonial wading birds are inextricably linked. Let us hope that both do well.

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Effects of Grazing on the Recovery of Oligohaline Marshes Impacted by Hurricane Andrew

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ABSTRACT: Hurricane Andrew made landfall in coastal Louisiana on August 24, 1992. The storm was especially damaging to oligohaline marshes, and created areas of scour. In these scour areas, unconsolidated or weakly rooted marsh was washed away by the storm. With the vegetation and root mat removed, bare substrate several centimeters lower than the original marsh elevation remained and became shallow ponds. We hypothesized that the lack of recovery in scours is caused by the combined stresses of grazing and increased flooding on seedling establishment. As a management option, we evaluated if plants could be established in the scour through plantings and the effect of grazing on the success of plantings. Four species were chosen to represent a large range of flooding and salinity tolerance: *Distichlis spicata*, *Paspalum vaginatum*, *Spartina alterniflora*, and *Spartina patens*. Two scours identified during an aerial survey of hurricane impacts were used in this study: Locust Bayou and Creole Bayou, both located in Terrebonne parish, LA. Natural colonization of ungrazed plots (installed in February, 1995) started in the spring of 1996. At Creole Bayou, the most abundant colonizers of ungrazed treatments were *Amaranthus australis*, *Scirpus validus*, *Sagittaria lancifolia*, *Scirpus americanus* and *Echinochloa walteri*. In contrast, *Scirpus americanus* and *Ammania coccinea* colonized the grazed scour at Creole Bayou. At Locust Bayou, only three species colonized the ungrazed treatments: *Scirpus americanus*, *Spartina patens*, and *Aster tenuifolius*. In 1997, five years after the passage of Hurricane Andrew, no natural colonization of the grazed scour at Locust Bayou had occurred. We attribute the difference in colonization between the two sites to the higher flooding frequency and duration at Locust Bayou. After one growing season, survival of transplants in the ungrazed treatments was significantly higher than in the grazed treatments for all species. No transplants survived in the grazed areas after the winter and no resprouting occurred during the next growing season. From the increase in cover of the transplants after two growing seasons, it appears that *Paspalum vaginatum* performs the best in both locations. *Spartina alterniflora* was the best performer at Locust Bayou, but the exclosures protecting these plants failed to keep out muskrat during the next winter. This species also performed well at Creole Bayou. *Distichlis spicata* is an unsuitable species for restoration of frequently flooded scours such as the scour at Locust Bayou. *Spartina patens* showed the lowest overall increase in cover. Revegetation of scour areas is important, because the shallow lakes formed in the scour at high water levels can lead to further erosion and land-loss in the area surrounding the scour. Through the exclusion of grazers, we have shown that parts of the scour with sufficient elevation will be colonized by several species that are common in oligohaline marshes. The revegetation of scours can be increased by plantings as long as they are protected from grazers.

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INTRODUCTION

Hurricane Andrew made landfall in coastal Louisiana on August 24, 1992. The hurricane had sustained winds of 54 m sec^{-1} and created a storm surge of 1–2 m. The storm was especially damaging to oligohaline marshes (Jackson et al. 1995). Guntenspergen et al. (1995) described six types of disturbance that occurred during the storm: (1) compression of the marsh surface, (2) deposition of thick sediment, (3) deposition of wrack, (4) deposition of thin sediment, (5) scour, and (6) salt burn. In this paper we describe the recovery of areas of scour by Hurricane Andrew.

In these scour areas, unconsolidated or weakly rooted marsh was washed away by the storm. With the vegetation and root mat removed, bare substrate several centimeters lower than the original marsh elevation remained and became shallow ponds. Surveys shortly after the hurricane showed that there was little or no recovery in the scour areas (Guntenspergen et al. 1995). Colonization of these shallow ponds was very slow, probably because the seed bank was removed with the vegetation and root mat. Guntenspergen et al. (1995) predicted little or no recovery in the scour areas, because elevations are too low for emergent plant establishment.

There are several herbivores in the coastal marshes of Louisiana, including waterfowl, deer, rabbits, muskrats and nutria. Grazing has been shown to reverse or halt succession (Bakker, 1985; Giroux and Bedard 1987; Shaffer et al. 1992) and could therefore be an additional factor in the lack of recovery in the scour areas. Previous studies have been shown that grazing can have a greater effect in the presence of an additional stress such as fire or salinity (Smith and Kadlec 1985, Taylor et al. 1994, Grace and Ford 1996). We hypothesized that the lack of recovery in scours is caused by the combined stresses of grazing and increased flooding on seedling establishment.

As a management option, we evaluated if plants could be established in the scour through plantings and the effect of grazing on the success of plantings.

METHODS

Natural Colonization

We selected two scours on the central Louisiana coast identified during an aerial survey of hurricane impacts (Sasser et al. 1994). One site was located on a canal east of Locust Bayou and the other south of Creole Bayou (Figure 1). Both scours were surrounded by oligohaline wiregrass marsh (Visser et al. 1996), a vegetation type dominated by *Spartina patens* (Ait.) Muhl. At each of these sites 3 replicate exclosures (2 x 2 m) were randomly established at the edge of the scour area. Each exclosure (ungrazed treatment) was paired with an area adjacent to the exclosure which was unprotected from grazing (grazed treatment). Species composition and percent cover (Braun-Blanquet scale; see Kent and Coker 1992) were recorded for all plots on February 2, 1995 at Creole Bayou, and on February 22, 1995 at Locust Bayou and after one growing season (October 9, 1995). However, no emergent plants had colonized by this time. Species composition and cover in this experiment were also recorded 6 times during 1996 and 1997 at the same times that the transplant experiment was visited (1 July 1996, 8 August 1996, 1 October 1996, 27 March 1997, 20 June 1997, and 15 September 1997).

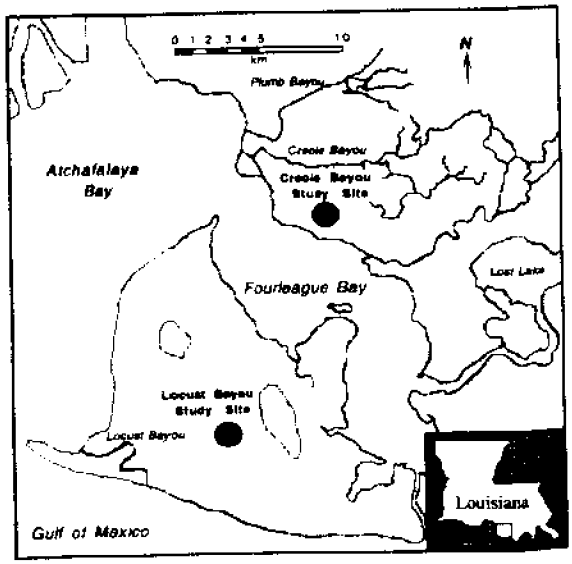

Figure 1. Location of the study sites in southwestern Terrebonne Parish, LA.

Plantings

In 1996, we implemented an additional experiment which consisted of 2 levels of grazing (grazed, ungrazed) and five plantings (*Distichlis spicata* (L.) Greene, *Paspalum vaginatum* Sw., *Spartina alterniflora* Loisel., *Spartina patens*, and not planted) with two replicates at the same two scour sites. Species were chosen to represent a large range of flooding and salinity tolerance. In June of 1996 we constructed 10 additional exclosures (2 x 2 m) in each area. These exclosures and adjacent control areas were planted with nursery grown plants on July 1 and 2, 1996. Ten sods (diameter 10 cm) of the randomly assigned species were planted in each replicate. Survival of the transplanted sods was determined on August 8 and October 1, 1996. In addition plant species which naturally colonized in either the grazed or ungrazed treatments were identified. Vegetation cover of transplants and colonizers was estimated to the nearest five percent on the same dates as the survival estimates. During the second year of the study, individual transplanted sods could not be separated and cover only was determined on March 27, June 20, and September 15, 1997.

Environmental Measurements

To account for differences in flooding stress and grazing pressure at the two scour sites we measured these two variables at both sites.

Flooding stress was estimated by determining the percentage of time each planting was flooded. Therefore, a water-level gauge was operated inside each scour from July 19, 1996 to September 15, 1997 and the elevation of the scour relative to the gauge was measured on October 1, 1996.

Grazing pressure was determined by estimating herbivore density and grazing frequency in the proximity of the experimental treatments. Four transects of 1 km surrounding the study area were flown by helicopter on May 13, 1997 (50 meter above the marsh at an approximate speed of 50 $km \cdot h^{-1}$). All herbivores within 25 meters of the transect were counted on both sides of the helicopter. To avoid counting the same animals twice, the transects were not connected with each other. To account for possible difference in visibility of animals among sites, we also performed ground surveys. Ground surveys consisted of four randomly selected sites surrounding each scour and consisted of six plots (1 x 1 m) per site for a total of 24 plots. In these plots, scat was counted and presence/absence of grazing either above ground (clipped stems) or below ground (digs) was recorded.

Statistical Methods

Paired t-tests were used to test for differences in survival of transplants due to grazing. Plots were paired within site such that the grazed and ungrazed treatment in each pair had similar elevations (flooding frequencies). Only the end-of-growing season survival percentages (October 1, 1996) were used in the analysis of survival. Due to the small sample size (2 replicates per treatment per site) normality of the survival data was not determined. However, the t-test is robust to departures from normality (Gilbert 1989).

Analysis of variance was used to determine the significance of differences in transplant cover. The design was a two-way factorial (treatment grazing with 2 levels and treatment planted species with 4 levels) blocked by site (2 levels) and split on date (5 levels). The distribution of cover was not significantly different from normality (Shapiro-Wilk statistic = 0.73, $P<W=0.0001$). Separate analyses of variance were performed on the ungrazed treatments only to determine differences among sites and transplanted species in end-of-season cover in 1996 and 1997.

RESULTS

Grazing and Flooding Differences Among Study Sites

Locust Bayou has a much higher number of grazers present than Creole Bayou (Table 1). This is mostly due to the high number of muskrat observed in Locust Bayou. Deer and rabbits are relatively rare, and were only observed near Creole Bayou. The relative density of grazers is also reflected in the grazing pressure at both sites (Table 2). Locust Bayou has a higher grazing pressure both above and below ground than Creole Bayou.

Table 1. Average herbivore and scat densities at Creole Bayou and Locust Bayou study sites.

Species	Density of animals[1] (number • km^{-2})		Density of scat (number • m^{-2})	
	Creole Bayou	Locust Bayou	Creole Bayou	Locust Bayou
Nutria	3.8 ± 4.9	10.6 ± 7.7	0	0
Common Muskrat	0	38.8 ± 23.5	0	1.6 ± 2.5
White-tailed Deer	0.6 ± 1.7	0	0	0
Swamp Rabbit	1.9 ± 3.5	0	0.0 ± 0.2	0

[1]Muskrat densities are in number of houses. Densities are provided ± one standard error.

Table 2. Estimate of grazing pressure at the two sites.

Grazed biomass	Grazing Pressure[2]	
	Creole Bayou	Locust Bayou
Above ground	12.5	58.3
Below ground	0.4	62.5

[2] Percentage of 1 m^2 plots at which evidence of grazing was present. Sample size was 24.

Both sites show similar ranges of water level fluctuations of approximately 1 m (Figures 2 and 3). However, the tidal amplitude at Locust Bayou is approximately 20 cm, while at Creole Bayou it is only 10 cm. The larger water-level fluctuations at Locust Bayou are due to wind driven setup and discharge from the Atchafalaya River. Due to the relative elevation of the scour surface, the substrate at Locust Bayou is almost never drained (2% of the time) and has an average water level of 15 cm above the scour. In contrast the Creole Bayou scour is drained 15% of the time and has an average water level of 11 cm above the scour surface.

Natural Colonization of Scours

Natural colonization of ungrazed plots (installed in February, 1995) started at Creole Bayou in the spring of 1996, and at Locust Bayou in the summer of 1996 (Figure 4). However, the exclosure with the lowest elevation was never colonized at both sites. Colonization by *Scirpus americanus* Pers. (previously *S. olneyi* Gray) and *Ammania coccinea* Rottb. was observed in the grazed scour at Creole Bayou in 1996 (Table 3). At Creole Bayou, the most abundant colonizers of ungrazed treatments were *Amaranthus australis* (Gray) Sauer, *Scirpus validus* Vahl., *Sagittaria lancifolia* L., *Scirpus americanus* and *Echinochloa walteri* (Pursh) Heller (Figure 4). A total of 11 different species were observed in the ungrazed treatments at Creole Bayou. However, *Spartina patens*, the dominant in the surrounding marsh, was absent. At Locust Bayou, only three species colonized the ungrazed treatments: *Scirpus americanus*, *Spartina patens*, and *Aster tenuifolius* L. (Figure 4).

Survival and Cover of Transplants

After one growing season, survival in the ungrazed treatments was significantly higher than in the grazed treatments for all species (Figure 5). At the end of the first growing season, a few transplants of each species survived in the grazed treatments of Creole Bayou, while only *Spartina patens* survived in these treatments at Locust Bayou. However, no transplants survived in the grazed areas after the winter and no resprouting occurred during the next growing season.

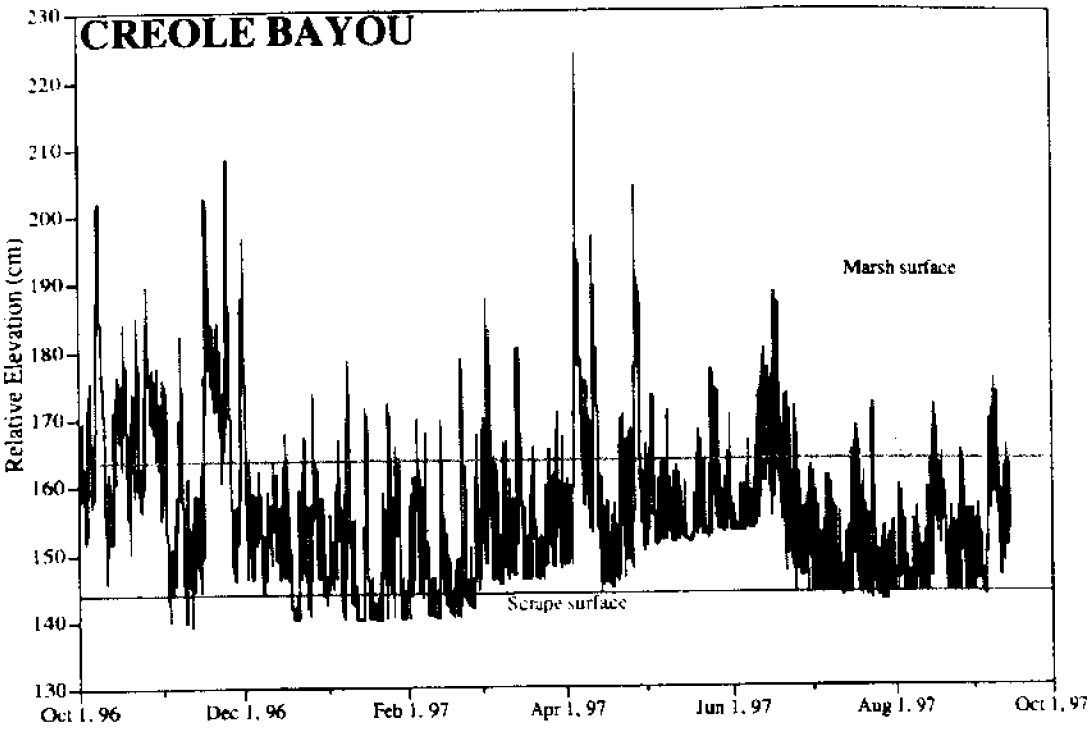

Figure 2. Hourly water level at Creole Bayou.

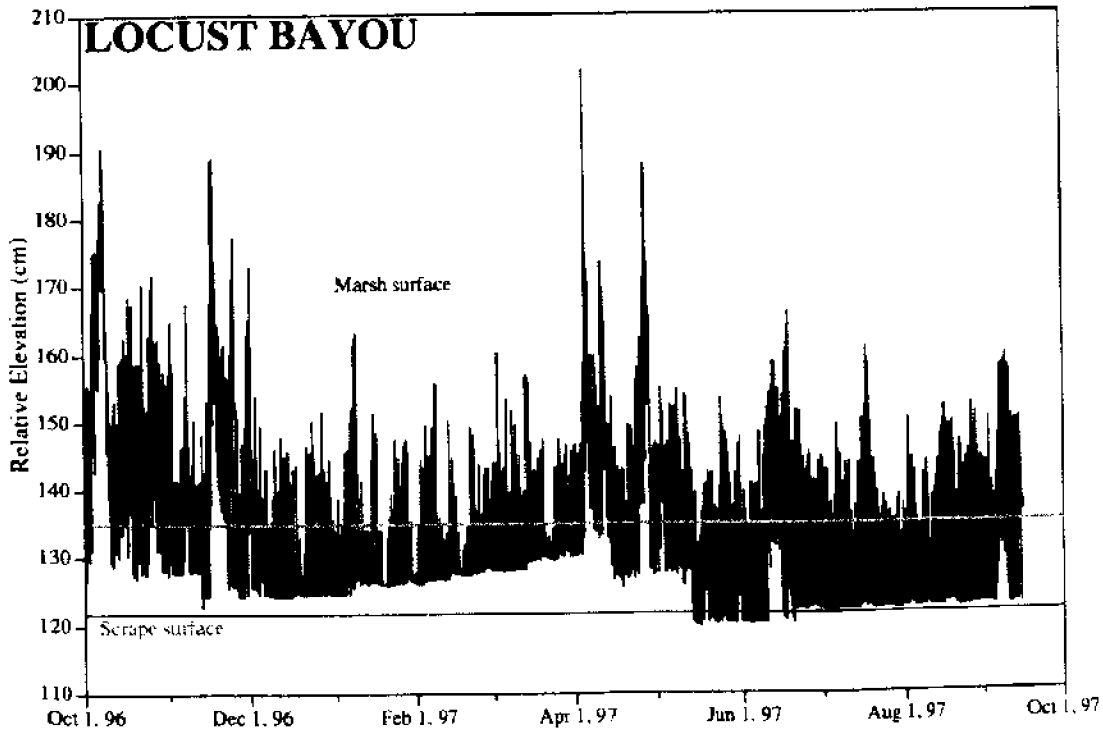

Figure 3. Hourly water level at Locust Bayou.

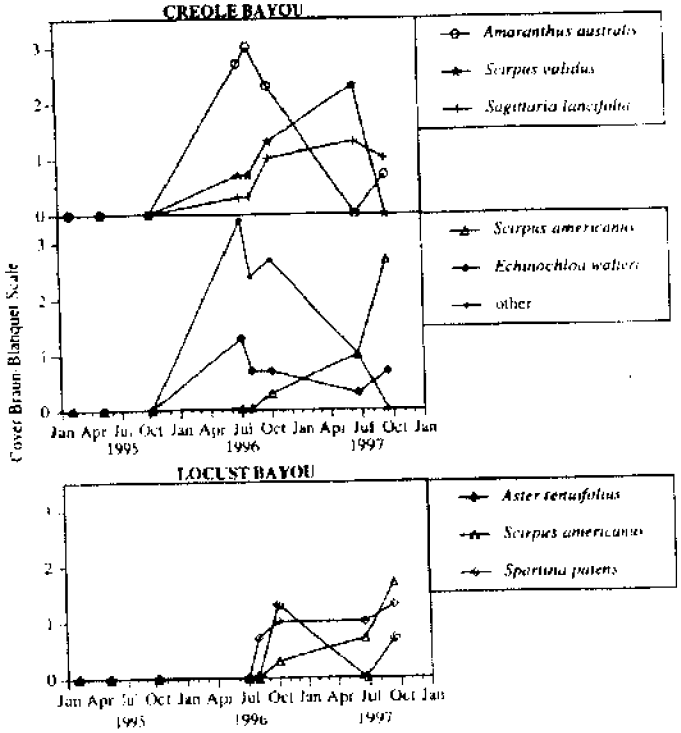

Figure 4. Natural colonization of ungrazed scours. Data from exclosures established in February 1995 within each scour.

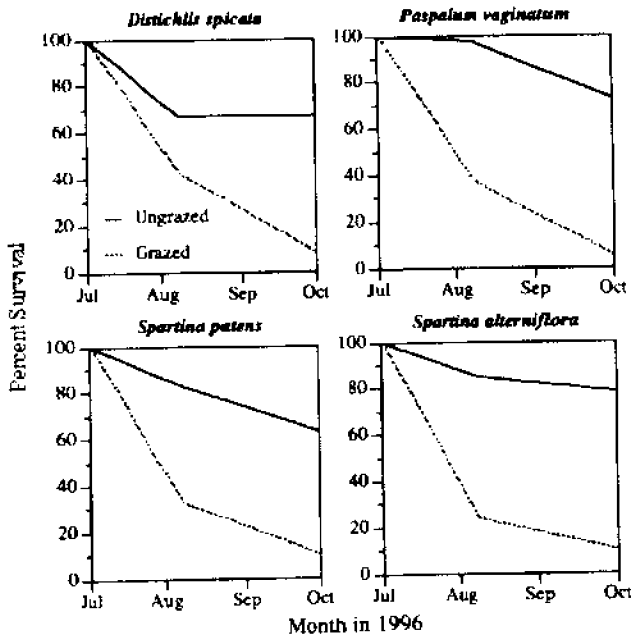

Figure 5. Survival of transplants in grazed and ungrazed treatments during the first year (1996). Data presented is the average survival for both sites.

Table 3. Natural colonizers of the plots planted with nursery stock at Creole Bayou. Numbers represent the number of plots that were colonized out of a total of ten 4 m² plots.

	1996		1997	
	Grazed Plots	Ungrazed Plots	Grazed Plots	Ungrazed Plots
Scirpus americanus	1	5	1	7
Sagittaria lancifolia	0	3	2	7
Ammania coccinea	2	1	0	0
Amaranthus australis	0	3	0	3
Paspalum vaginatum	0	1	0	1
Unknown dicot	0	0	1	1
Polygonum punctatum	0	0	0	3
Vigna luteola	0	0	0	2
Spartina patens	0	0	0	1

One month after planting, there was a significant difference between grazed and ungrazed treatments in plant cover for all species at all sites. Most species showed an increase in cover in the ungrazed treatments during the study, which slowed during the winter months (Figures 6 and 7). The exception was *Distichlis spicata* at Locust Bayou, which remained constant in cover since the planting. At Locust Bayou, both exclosures planted with *Spartina alterniflora* failed to protect the transplants from grazing during the winter, due to muskrat tunneling under the fence. Therefore, no data were obtained for this species during the second year of the study.

At Creole Bayou, nine other species colonized the ungrazed-planted treatments. Of these *Scirpus americanus* and *Sagittaria lancifolia* colonized the most plots (Table 3). Only one of the ungrazed-planted treatments was invaded by *Scirpus americanus* at Locust Bayou.

DISCUSSION

Our results show that grazing is a significant factor in the reduced recovery of areas scoured by Hurricane Andrew. Therefore, the lack of colonization of scour areas is not only due to decreased elevation, as postulated by Guntenspergen et al. (1995). Although the seedbank was removed during the hurricane in 1992, regeneration of the seedbank has occurred from adjacent marsh seed sources. Through the exclusion of grazers, we have shown that the parts of the scour with sufficient elevation will be colonized by several species (Figure 4 and Table 3) that are common in oligohaline wiregrass marshes (Visser et al. 1996). The most frequently flooded scour at Locust Bayou had fewer colonizing species (three) than the less frequently flooded Creole Bayou scour (11 species). Long-term monitoring of these exclosures will be necessary to determine if species composition will change over time to resemble the surrounding *Spartina patens*-dominated marsh. *Spartina patens* was only found in one exclosure at Locust Bayou and is a species with low seed production. This species also performed poorly when transplanted into the scour, probably because of the high frequency and duration of flooding in the scours. At both sites, *Scirpus americanus* was an abundant natural colonizer. This species also increased in cover when grazers were excluded from oligohaline marshes dominated by *Spartina patens* (Taylor et al. 1994).

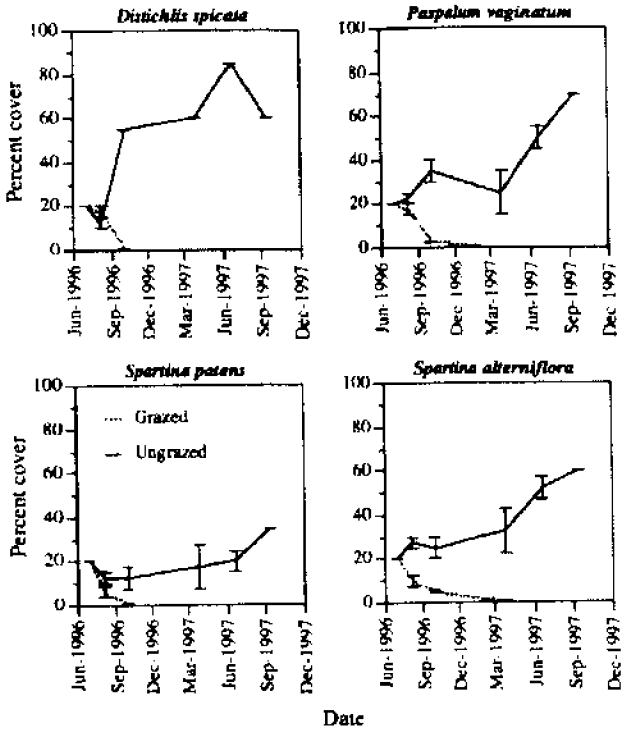

Figure 6. Percentage cover of transplants in grazed and ungrazed treatments at Creole Bayou.

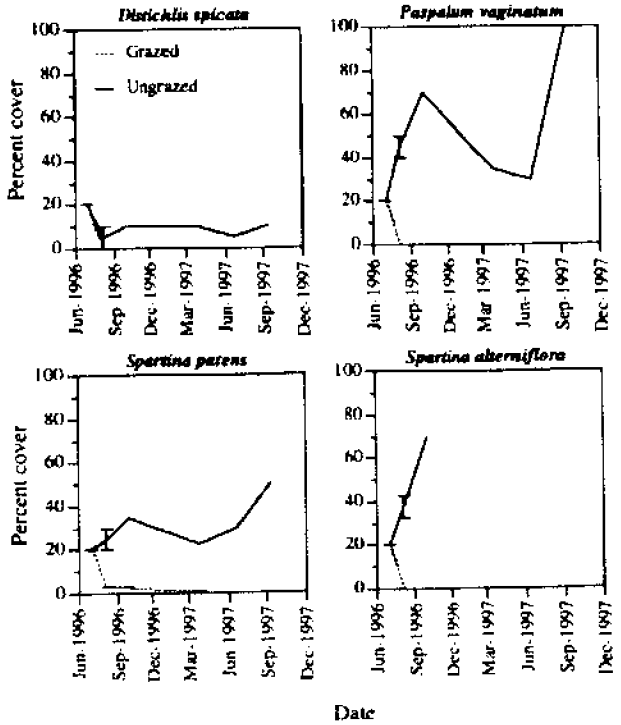

Figure 7. Percentage cover of transplants in grazed and ungrazed treatments at Locust Bayou. Exclosures planted with *Spartina alterniflora* failed to exclude grazers during the winter and therefore could not be considered ungrazed treatments in 1997.

Some colonization of the scour area at Creole Bayou occurred in the presence of grazing. This might be due to the lower grazing pressure as well as the lower frequency of flooding in this area. In contrast, the more heavily grazed and more frequently flooded Locust Bayou scour showed no colonization in grazed areas. The area surrounding the Locust Bayou scour experienced a rapid increase in muskrat densities during the study (Noel Kinler, Louisiana Department of Wildlife and Fisheries, personal communication). These rapid local increases in population density of muskrat are well documented (O'Neil 1949, Lowery 1974, Condrey et al. 1995). Average muskrat densities in oligohaline marshes are 21 houses•km^{-2} (Palmisano 1972). Therefore, our estimate of 40 houses•km^{-2} (Table 2) is relatively high.

From the increase in cover of the transplants after two growing seasons, it appears that *Paspalum vaginatum* performs the best in both locations. *Spartina alterniflora* was the best performer at Locust Bayou, but the exclosures protecting these plants failed to keep out muskrat during the next winter. This species also performed well at Creole Bayou. *Distichlis spicata* is an unsuitable species for restoration of frequently flooded scours such as the scour at Locust Bayou. *Spartina patens* showed the lowest overall increase in cover.

Revegetation of scour areas is important, because the shallow lakes formed in the scour at high water levels can lead to further erosion and land-loss in the area surrounding the scour (McGinnis 1997). Through the exclusion of grazers, we have shown that parts of the scour with sufficient elevation will be colonized by several species that are common in oligohaline marshes. The revegetation of scours can be increased by plantings as long as they are protected from grazers. When protected from grazers, *Paspalum vaginatum* increased most in cover in both oligohaline scours. We, therefore, recommend the use of *Paspalum vaginatum* for future restoration efforts.

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