

/ ASPECTS OF MARSH ACCRETION AND GEOCHEMISTRY:

BARATARIA BASIN, LOUISIANA /

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## ABSTRACT

Rapid land loss in coastal Louisiana has accentuated the need to understand processes of marsh accretion and degradation and their role in coastal nutrient cycles. This study examines the rates and processes of vertical accretion, material flux, and availability of nutrients and sulfides in selected freshwater, intermediate, brackish, and salt marshes in a Mississippi River delta flank depression, Barataria Basin.

Approximately similar short term vertical accretion rates were identified by  $^{137}\text{Cs}$  dating in the freshwater, intermediate, brackish, and salt marshes, highest rates occurring in natural levee deposits. Whereas local gradients were clearly defined by the supply of waterborne mineral sediments, regional patterns indicate more complex relationships independent of regional subsidence or mineral sediment supply. These results suggest that accretionary processes and the response to changes in mean sea level may vary between vegetatively defined marsh types.

The accumulation of sedimentary minerals, phosphorus, iron and manganese, was largely governed by the flux of mineral sediments, defined in turn by the hydrologic energy gradient. By contrast to iron, phosphorus and manganese were subject to significant post-depositional remobilization and did not accurately conform to the distribution of mineral sediments. Organic carbon and nitrogen constitute a roughly constant volumetric proportion of marsh soils so that accumulation of these elements was defined by the vertical accretion rate only.

The distribution of sulfides was consistent with a sulfate-limited model, with a secondary dependence on the abundance of sedimentary iron. Bulk density was important in defining variations in the availability of iron and manganese, but phosphate and ammonia showed inconsistent behavior. Factors influencing nutrient availability are potentially complex and include anthropogenic inputs. Vertical redox gradients were the primary factor governing extractable nutrient and sulfide profiles.

## INTRODUCTION

Marshes constitute a major facet of coastal environments on the North American continent. By virtue of their spatial extent alone, these systems bear significantly on man's efforts to successfully manage the coastal zone. Until recently this coastal environment has been poorly understood, and consequently has been the subject of severe and frequently indiscriminate exploitation. The sensitive nature of these systems (Craig et al., 1979) and their significant role in coastal nutrient cycles (Kjerfve and McKellar, 1980; Nixon, 1980) has become increasingly apparent from recent investigations. Thus, an understanding of the integral mechanisms controlling the stability of this environment is imperative for its successful management and continued existence.

The Gulf Coast of Louisiana is fringed by a vast expanse of wetland recognized for its exceptional natural productivity and economic potential. Fresh, brackish, and saline marshes and associated water bodies cover some 3.2 million hectares, equivalent to 41% of all coastal wetlands in the United States (Turner and Gosselink, 1975). Despite apparently rapid accretion in these marshes (DeLaune et al., 1978), land loss accelerated by increased land use pressures, has accentuated the need to comprehend the stabilizing relationships of this system.

Previous work concerning the accretionary stability and material flux in Louisiana marshes (DeLaune et al., 1978, 1979, 1981) has centered on the salt marsh. Various other studies (Brannon, 1973; Chabreck, 1972; Bruppacher et al., 1973; Ho et al., 1976; Rainey,

1979) have confronted aspects of nutrient availability and diagenesis in the various marsh environments. However, processes of marsh accretion and nutrient flux in brackish, intermediate, and freshwater marshes remain poorly understood.

This study was designed to achieve the following objectives in selected freshwater, intermediate, brackish, and salt marsh soils of Barataria Basin, Louisiana:

- (i) to establish the rate and nature of recent sedimentation using the  $^{137}\text{Cs}$  dating technique,
- (ii) to determine the total reservoir and flux of selected elements including organic carbon, nitrogen, phosphorus, iron and manganese,
- (iii) to determine the availability of selected plant nutrients including phosphates, ammonium, iron and manganese, and of sulfides to a depth of 45 cm in these soils.

## SETTING

### A. GEOLOGY

The Mississippi River deltaic plain is subject to rapid local and regional subsidence resulting from several concomitantly operative processes. Basement sinking (downwarping) caused by sediment load, consolidation of the sediments of the Gulf Coast geosyncline, local consolidation, and tectonic activity have been used in evaluating apparent movements of land and sea (Kolb and Van Lopik, 1966). Regional subsidence, estimated to be greater than 1 cm/year at the present shoreline (Swanson and Thurlow, 1973), decreases with distance inland.

For thousands of years, distributaries of the Mississippi River have switched back and forth across the deltaic plain distributing sediments in a series of overlapping and contemporaneous deltaic lobes (Frazier, 1967). Periods of seaward progradation and fluvial dominance alternate locally with periods of land loss and marine transgression. Relict distributary channels and natural levee ridges form an elevated arterial array bounding the lower interdistributary basins.

The Barataria Basin is one such interdistributary basin, bounded on the east by the Mississippi River and on the west by its most recently abandoned channel, Bayou Lafourche (Fig. 1). Ordinarily, annual overbank flooding from the river would provide a supply of fresh sediment, water, and nutrients to the system. In recent years the basin has been deprived of this river-borne sediment through natural stream diversion and flood control measures. Barataria Basin is thus a



Fig. 1. Map of Barataria Basin showing study areas and marsh environments. (After Bahr and Hebrard, 1970).

well-defined sedimentary and hydrologic unit in which accretionary processes rely largely on locally reworked sediments and *in situ* plant production.

Unlike southern coastal plain marshes, peat is a significant constituent of the recent sediments of the Mississippi River delta. Kolb and Van Lopik (1966) recognize three major sedimentary marsh types:

(i) Floating marsh or "flotant" a mat of roots and other plant remains admixed with a fine muck 10-35 cm thick, underlain by 1-5 m of organic ooze that grades with depth to clay. Consolidation of the ooze results in a black organic clay in which organic matter composes more than 50% of the deposit.

(ii) Brackish-intermediate marsh, vegetative mat and muck 10-20 cm thick, underlain by 0.3-3.1 m of coarse-to-medium textured fibrous peat, underlain in turn by a fairly firm blue-gray clay or silty clay containing lenses of dark clays rich in organic matter. Only 10-20% of these deposits were thought to consist of inorganic sediments.

(iii) Saline to brackish marsh, a mat of roots, stems, and leaves 10-50 cm thick, underlain by a rather firm blue-gray clay. Clays are increasingly firmer and freer of organic matter with depth. Silt to fine sand content may range as high as 30%, clays rich in organics average 50% and peat typically between 15% and 30% of the deposit.

The clay mineral suite of Barataria Basin is composed primarily of montmorillonite with small amounts of kaolinite and illite (Ho, 1971).

#### B. PHYSICAL FACTORS

Tidal inundation and wind-induced water movements are the major physical processes governing sedimentation and sedimentary chemistry in Barataria Basin salt marshes (Brannon, 1973; Baumann, 1980). The lower basin area is flushed daily by tides with a mean linear amplitude of approximately 50 cm. Hydraulic energy is greatest in the salt marsh environment and diminishes inland. Water levels in brackish, intermediate, and freshwater systems parallel the gross seasonal pattern of the salt marsh (Byrne et al., 1976) but short-term tidal oscillations affect only the saline and brackish environments. Marsh substrates are permanently saturated and reduced conditions prevail.

#### C. VEGETATIVE ZONATION

Four primary vegetative marsh units can be identified which generally correlate with substrate type and salinity (Fig. 1): salt marsh, brackish marsh, intermediate marsh, and freshwater marsh (Bahr and Hebrard, 1976). Freshwater marsh (salinity  $\leq 1^{\circ}/\text{o}$ ) covers 19% of the basin, and is characterized by Panicum hemitomon, Eleocharis sp. and Sagittaria falcata. Brackish and intermediate marshes ( $1-10^{\circ}/\text{o}$ ) together cover approximately 20% of the basin. The brackish marsh is vegetated largely by Spartina patens, Distichlis spicata and small amounts of S. alterniflora. The intermediate marsh is characterized by a significant increase in the dominance of S. patens and the absence of S. alterniflora. Salt marsh ( $10-20^{\circ}/\text{o}$ ) covers approximately 14% of

the basin and is widely interspersed with open water bodies. S.  
alterniflora comprises roughly 60% of the vegetation but up to 95%  
locally; Juncus romerianus and Distichlis spicata are also common.

## REVIEW OF THE LITERATURE

### A. MARSH ACCRETION

#### 1. Methodology

A wide variety of methods have been used in the determination of recent marsh accretion rates. The vast majority of investigations have employed visual particulate tracers (Bloom, 1967; Harrison and Bloom, 1974; Richard, 1978; Stearns and McCreary, 1957). The historic record (Flessa et al., 1977), stratigraphy and radiocarbon dating (Redfield, 1972), palynology (Mudie and Byrne, 1980), and  $^{210}\text{Pb}$  (Muzyka, 1978) have also been used with a variable degree of accuracy. DeLaune et al. (1978) were the first to apply fallout radiocesium ( $^{137}\text{Cs}$ ) to the dating of marsh sediments, following the successful application of this method in lacustrine (Pennington et al., 1973) and deep water marine environments (Schreiber, 1968). Most evidence to date supports the use of  $^{137}\text{Cs}$  in the dating of a variety of sedimentary systems (Aston and Stanners, 1979; Krishnaswamy et al., 1971; Robbins and Edgington, 1975), but its behavior under reduced and vegetated conditions is poorly understood. The ensuing paragraphs review briefly present knowledge of the behavior of  $^{137}\text{Cs}$  in sedimentary environments as related to dating techniques.

$^{137}\text{Cs}$  was introduced into the world environment in the early fifties. Nuclear fallout, recorded in the sedimentary profile, peaked in 1963 with a secondary peak in 1959 (Hardy, 1975). Application of  $^{137}\text{Cs}$  profiles in the determination of sediment accretion rates over

the past two decades is facilitated by the ubiquitous occurrence of this isotope in recent sedimentary environments and its facile (direct) analytical determination. The method is subject to the constraint that the sedimentary profile of  $^{137}\text{Cs}$  relates accurately to the yearly fallout pattern. While this has been confirmed in specific lacustrine environments (Pennington et al., 1973), the general applicability of the method is limited by site-specific perturbations which include (i) desorption coupled with diffusion, (ii) physical and biophysical disturbance, (iii) selective adsorption by specific sedimentary size fractions, and (iv) plant translocation.

$^{137}\text{Cs}$  is strongly adsorbed in the fine grained mineral and organic fraction of sediments, even in areas of differing sedimentary characteristics (Alberts and Muller, 1979; Davis, 1963; Gardner and Skulberg, 1964; Tamura, 1964). Because of the high distribution coefficient and the almost irreversible fixation of this element in illite interlattice sites (Laggerwerff et al., 1976; Lomenick and Tamura, 1965; Prout, 1958), alteration of the sedimentary  $^{137}\text{Cs}$  profile by desorption processes is unlikely. Evidence from marine (Aston and Stanners, 1979; Duursma and Bosch, 1970; Duursma and Gross, 1971) and lacustrine environments (Carrigan et al., 1967; Lomenick and Tamura, 1965; Pennington et al., 1973) suggests that diffusive mixing in the presence of adsorption is insignificant.

Evidence to date suggests that  $^{137}\text{Cs}$  is generally not subject to reductive remobilization. Cyclic release from seasonally reduced lacustrine sediments was recorded by Alberts et al. (1979) but neither stable Cs nor  $^{137}\text{Cs}$  showed significant release from the sediments

of other similarly reduced systems (Eyman and Kevern, 1975; Hakonson and Whicker, 1975). Under controlled Eh conditions, DeLaune and Reddy (personal communication) found that greater quantities of this isotope were desorbed from Mississippi River sediments incubated under oxidized as opposed to reduced conditions. Profiles from the reduced marsh sediments of Louisiana (DeLaune et al., 1978) further support a model of insignificant mobility under reduced conditions.

The impact of plant uptake and translocation on sedimentary <sup>137</sup>Cs profiles is, as yet, poorly understood. Garten and Paine (1977) recorded an inverse relationship between <sup>137</sup>Cs availability and soil moisture content, organic matter content, extractable potassium, and percentage of very fine sand and silt. However, D'Souza and Mistry (1980) found that flooding enhanced the uptake of this isotope by rice plants grown in pellustrent and oxisol soils. Total soil concentration of <sup>137</sup>Cs apparently does not influence uptake by two wetland species, Sagittaria latifolia and Polygonum punctatum (Sharitz et al., 1974).

Available evidence thus supports the use of <sup>137</sup>Cs in sedimentary dating, even under reduced conditions.

## 2. Vertical Accretion Rate

Marsh systems require that the altitude of the sediment surface remain adjusted to long-term water level changes in order for plants to survive. Viable natural systems achieve a delicate balance between accretionary processes of mineral and organic sediment accumulation, and those factors governing relative sea level, including regional subsidence, local substrate compaction, and eustatic sea level

rise. That the buildup of salt marshes is related to the supply of mineral sediments is evidenced by numerous studies (including Chapman, 1960; DeLaune et al., 1978; Redfield, 1972). Frequency and duration of inundation are important in defining patterns and rates of mineral sedimentation, but high energy climatic events are frequently the primary agent of sedimentary influx (Baumann, 1980; Richard, 1978). McCaffrey (1977) suggested that salt marshes lacking an abundant silt supply need not build up in this way, and that vertical accretion determined by vegetative growth and organic detrital accumulation.

Salt marsh sedimentation rates and processes have been examined in a variety of coastal areas on the North American Atlantic, and Gulf Coasts; these results are summarized in Table 1. Chapman (1960) reviewed available data on the marshes of western Europe including Guilcher and Berthois (1957), Jakobsen et al. (1935, 1955), Oliver (1929), Ranwell (1963), Richards (1934), and Steers (1939): accretion in the mid and upper zones of European marshes is usually between 2 and 10 mm/y which approximates values for the eastern United States, but Oliver (1929) and Oliver et al. (1929) cite values as high as 200 mm/y in Dutch S. townsendii marshes. Accretionary processes in typically organic and nonsaline marsh environments are poorly documented at this time.

#### B. MATERIAL ACCUMULATION IN MARSHES

Concomitant with the vertical accretion of marshes is the accumulation of detrital minerals. The long term fate of these materials is of consequence to estuarine nutrient cycles and is governed

Table 1. Summary of available marsh accretion data from the North American Atlantic seaboard.

Location	Marsh Type	Accretion Rate (mm/y)	Mean Sea Level Rise (mm/y)		Source
			Mean Sea Level Rise	Net hood*	
MA	<u>S. alterniflora</u>	18.3 (1.5-51.8)	3.4	S	Hedfield (1972)
CT	<u>S. alterniflora</u>	8-10	2.5	P	Bloom (1967 in Richard, 1978)
CT	<u>S. patens</u>	2-5	2.5	P	Harrison and Bloom (1974)
NY	<u>S. alterniflora</u>	4.0	2.9	Pb-210	Murphy (1976)
NY	<u>S. alterniflora</u>	4.7-6.3	2.9	Pb-210	Arcenano and Woodwell (1975)
NY	<u>S. alterniflora</u>	2.0-6.2	2.9	P	Richard (1978)
NY	<u>S. alterniflora</u>	2.5	2.9	H	Fleeg et al. (1977)
DE	<u>S. alterniflora</u>	5.1-6.3	3.8	P	Stearns and McCrea (1957)
DE	<u>S. alterniflora</u>	5.0	3.8	Pb-210	Lord (1980)
<i>(aborts)</i>				Ca-137	Hopkinson (pers. comm.)
GA	<u>S. alterniflora</u>	3-5		Ca-137	Hopkinson (pers. comm.)
LA	<u>S. alterniflora</u> (Deltaic Plain) (i) streamside	13.5	9.2	Ca-137	Delaune et al. (1978)
	ii) inland	7.5	5.2	Ca-137	
LA	<u>S. alterniflora</u> (Chenier Plain)	6.7	9.2	Ca-137	Delaune (pers. comm.)
	(4.4-8.2)				
LA	<u>S. alterniflora</u> (Deltaic Plain) (i) streamside	15.2	9.2	P	Bauman (1980)
	ii) inland	9.1			

\* S = Stratigraphy, P = Visual Particulate, H = Historic Record

by physical, chemical, and biological remobilization processes within the sediments and by exchanges across the sediment-water interface.

Sedimentary profiles provide a convenient record by which these processes may be assessed. The role of marsh sediments in estuarine nutrient cycles is, as yet, poorly understood. Available information concerning the abundance, accumulation, and remobilization of selected elements (including organic carbon, nitrogen, and phosphorus) in marsh soils is summarized in Table 2.

#### 1. Organic Carbon

The organic carbon content of North American salt marsh sediments studied to date ranges from 4% (Nixon, 1980) to 34% (Brannon, 1973). In general, few sediments exceed 10% organic carbon, the notable exceptions being the marshes of the Mississippi River delta and north eastern marshes (Frey and Bason, 1978; Macdonald, 1977).

The distribution and accumulation of organic carbon in nonsaline (freshwater, intermediate, and brackish) marshes in Louisiana has not been accurately established. Brupbacher et al. (1973) found a relatively uniform dry weight concentration in all environments, whereas Kolb and Van Lopik (1966) suggest a general decline from freshwater marshes toward the Gulf Coast.

Detailed profiles of organic carbon in marsh soils are few and interpretation of existing data in the context of material cycles is confused by irregular variations in mineral sediment supply and uncertainties in belowground production estimates. Profiles from Louisiana (Brannon, 1973; DeLaune et al., 1981) and Georgia (Valiella

Table 2. Summary of available data on organic carbon, total nitrogen and phosphorus in marsh soils in the United States.

Location	Sample Interval (cm)	Organic Carbon (%)	Total Nitrogen (%)	Phosphorus* (ug/g)	Source
<u>Salt Marsh</u>					
RI (Providence River)	0-5	5.2	.49	31-53 (E)	Nixon (1980)
RI (Narragansett Bay)	0-5	4.1	.24	10-42 (E)	Nixon (1980)
RI (Block Island Sound)	0-5	4.4	.36	17-28 (L)	Nixon (1980)
NC	0-15	1.6		7-71 (E)	Broome et al. (1975) in Nixon (1980)
DE	0-60	4-24			Lord (1980)
LA (Coastal Plain)	0-30	0.8-36.0	0.05-2.14	4-190 (L, Dry)	Chabrek (1972)
LA (Barataria Basin)	0-90	9-34	0.4-2.0	20-35 (E)	Brannon (1973)
LA (Barataria Basin)	0-45	5.4-25.4	1.5-1.3	381-712 (T)	Delaune et al. (1981)
LA (Barataria Basin)	0-20		.75		Ho et al. (1976)
GA		1.5-5.0	.10-.15		Hathen et al. (1977)
GA				293-431 (M)	Maye (1972) in Delaune et al. (1976)
GA	0-80	6.3-14.7			Barnes et al. (1973) in Frey and Sason (1978)
<u>Salt Marsh</u>					
LA (Barataria Basin)	0-30	8.4-31.4	.6-2.47	3-63 (E, Dry)	Chabrek et al. (1972)
LA (Barataria Basin)	0-20	10.1			Ho et al. (1976)
<u>Intermediate Marsh</u>					
LA (Barataria Basin)	0-30	29.8-41.7	1.9-3.1	3-4 (L, Dry)	Chabrek et al. (1972)
<u>Freshwater Marsh</u>					
LA (Barataria Basin)	0-30	30.1	1.6?	10 (E, Dry)	Chabrek et al. (1972)
LA (Barataria Basin)	0-20		1.44		Ho et al. (1976)
WI (Theresa Marsh)		40.4-43.4	1.75-1.88	50-203 (E)	Klopatek (1975)
WI (Shakey Marsh)	Surface			250-400 (T)	Lee et al. (1975)
FL (Everglades)	0-15	47.2-50.5	1.16-3.29	5440 (T)	Volk et al. (1975)

\*E = Extractable, T = Total, L = Leachable

et al., 1979) generally reflect the depositional rather than the diagenetic history of accumulation. By contrast, profiles from highly mineral salt marsh substrates in Delaware (Lord, 1980) show strong indications of subsurface remobilization, presumably as a result of microbial decomposition: organic carbon decreased predictably with depth from 24% to less than 5% within 45 cm of the sediment surface.

DeLaune et al. (1981) calculated organic carbon accumulation rates of  $393 \text{ g/m}^2/\text{y}$  and  $273 \text{ g/m}^2/\text{y}$  respectively in streamside and inland areas of a Louisiana salt marsh, equivalent to 77% and 47% of the estimated aboveground production. The sedimentary accumulation of organic matter and the relative contribution of belowground and aboveground components is an important but largely unknown facet of marsh nutrient dynamics.

## 2. Total Nitrogen

Evidence from several studies (Brannon, 1973; DeLaune et al., 1981; Haines et al., 1977) suggests that nitrogen is largely bound in the organic fraction of salt marsh sediments. The concentration and accumulation of nitrogen is thus constrained by the occurrence of organic matter as described in the preceding paragraphs. The total nitrogen content of salt marsh sediments ranges from less than 0.1% (Chabreck, 1973) to more than 1.4% (Ho et al., 1976), both values recorded in Louisiana. Most marsh sediments appear to contain less than 0.5% nitrogen by weight (Table 2).

DeLaune et al. (1979) found nitrogen accumulation rates of  $13.4 - 21.0 \text{ g/m}^2/\text{y}$  in Louisiana salt marshes, in good agreement with

Nixon's (1980) estimate ( $5-20 \text{ g/m}^2/\text{y}$ ) based on "typical" figures from a variety of salt marshes. A progressive increase in the dry weight nitrogen content of marsh sediments has been recorded from salt through less saline marsh environments in Louisiana (Ho et al., 1976; Bruppacher et al., 1973) but its accumulation in nonsaline marshes has not been established.

### 3. Phosphorus

Nixon (1980) has reviewed available information concerning the fate of sedimentary phosphorus in salt marshes. Virtually all data are for "available" forms, which do not include insoluble mineral, or organically bound fractions. DeLaune et al. (1979) report the only detailed profile of total sedimentary phosphorus in this environment: concentrations decline sharply from a surface value of  $650 \mu\text{g/g}$  to less than  $500 \mu\text{g/g}$  within 10 cm of the surface. Based on the same data, DeLaune and Patrick (1980) estimated a phosphorus accumulation rate of  $1.7 \text{ g/m}^2/\text{y}$ , equivalent to more than 60% of that deposited on the marsh surface; the remaining  $0.6 \text{ g/m}^2/\text{y}$  thus represents the portion remobilized and lost to estuarine waters. Based on the same data, Nixon (1980) computed a theoretical phosphorus export range of  $0.40-1.80 \text{ g/m}^2/\text{y}$  for eastern salt marsh environments, which he attributed to both physicochemical and biological remobilization processes. There is evidence to suggest that both plant uptake (Reimold, 1972) and diffusion across the sediment-water interface (Gardner, 1975; Nixon et al. in Nixon 1980) contribute to this flux but the precise contribution of each component is unknown.

#### 4. Iron and Manganese

Investigation of the behavior of metals in shallow water marine and freshwater sediments has been directed primarily at the major contaminants such as lead and cadmium. As nonpoint pollution sources, iron and manganese have received little attention in recent literature. Available information includes the studies of McCaffrey (1977) in a Connecticut S. patens marsh, and DeLaune et al. (1981) in the S. alterniflora marshes of Louisiana. These studies are consistent in illustrating the role of water-borne mineral sediments in supplying iron and manganese to the marsh surface, while emphasizing their different post-depositional behavior.

As in the case of phosphorus, manganese is subject to significant post-depositional remobilization and loss, primarily as a result of diffusion across the sedimentary interface (Nixon, 1980; Lord, 1980), resulting in pronounced subsurface depletion. McCaffrey (1977) recorded concentrations ranging from 1200  $\mu\text{g/g}$  at the surface to  $\leq 500 \mu\text{g}$  at depth in the sedimentary profile from which influx and remobilization rates of  $0.22 \text{ g/m}^2/\text{y}$  and  $0.1 \text{ g/m}^2/\text{y}$ , respectively, were calculated. DeLaune et al. (1981) found concentrations of 85-120  $\mu\text{g/g}$  to a depth of 45 cm in the soil profile, yielding influx rates of  $0.37 \text{ g/m}^2/\text{y}$  and  $0.13 \text{ g/m}^2/\text{y}$  in streamside and inland areas, respectively. Available salt marsh data indicate that in contrast to manganese and phosphorus, iron is not subject to significant post-depositional migration; profiles thus reflect the history of accumulation of mineral sediments. DeLaune et al. (1981) found average dry weight iron

concentrations of 17735  $\mu\text{g/g}$  to a depth of 45 cm in streamside sediments and 13089  $\mu\text{g/g}$  in inland areas, with respective accumulation rates of  $58.8 \text{ g/m}^2/\text{y}$  and  $19.9 \text{ g/m}^2/\text{y}$ . Additional information concerning the behavior of iron and manganese in salt marsh sediments is available from Barnes et al. (1973, in Frey and Bason, 1975), and from Lord (1980).

Volk et al. (1975) recorded concentrations of iron and manganese in freshwater marsh histosols in the Florida everglades of 5500-7520  $\mu\text{g/g}$  and 97-218  $\mu\text{g/g}$  respectively but no information is available about the dynamics of these metals in less saline marsh environments.

#### C. PHYSICOCHEMISTRY, SULFIDES, AND AVAILABLE NUTRIENTS

The availability of plant nutrients and sulfides in flooded soils is largely governed by the abundance of parent minerals, and by microbially mediated oxidation-reduction processes. Mechanisms governing these chemical and microbial transformations in flooded soil systems are relatively well understood. The ensuing section reviews briefly important aspects of the redox chemistry of flooded soils, as applied to sulfides and selected plant nutrients (including phosphates, ammonium, iron, and manganese) with appropriate examples from marsh environments where possible.

##### 1. Physicochemistry

Eh and pH are the two basic physicochemical parameters by which the redox condition of flooded soils is assessed. The principles and interpretation of these parameters are described by several texts

including Garrels and Christ (1965) and will not be discussed further.

The pH of submerged soils generally converge on 7 from acid or basic conditions as Eh declines (Ponnamperuma, 1972). Available data for the pH of marsh soils (Table 3) are almost entirely from the saline environment. Eh estimates are less readily available: Brannon (1973) recorded values typically between +100 mV and -200 mV in Louisiana salt marshes. The physicochemistry of less saline (brackish, intermediate, and freshwater) marshes has not been adequately studied.

## 2. Sulfides

Sulfides, a major product of anaerobic metabolism in reduced soils and sediments, occur in a wide variety of forms which include hydrogen sulfide ( $H_2S/HS^-$ ), metal sulfides, organosulfides and polysulfides. Whereas sulfate reduction and sulfide accumulation in marine systems is sometimes limited by metabolizable organic matter (Gooch, 1968; Reimold and Daiber, 1970; Brummer et al., 1971), sulfate is a major factor limiting sulfide production in some freshwater systems (Yamane and Sato, 1961). Connell and Patrick (1969) showed that sulfide accumulation begins at an Eh of approximately -150 mV, but is limited to the pH range 6.5-8.5. Precipitation of metal sulfides at the expense of soluble forms is favored in the presence of high metal concentrations, neutral to basic pH and low Eh conditions (Rickard, 1969).

Ho et al. (1971) found total (soluble + acid volatile) sulfides averaging 249  $\mu g/g$  to a depth of 200 cm in Louisiana salt

Table 3. Summary of available marsh pH data.

State	Marsh Type	pH	Source
CT	Salt	6.0 - 6.6	Friedman and Gavish (1970)
FL	Salt	7.1	Coultas (1970)
LA	Salt	7.0 - 7.8	Ho (1971)
LA	Salt	5.0 - 7.0	Brannon (1973)
DE	Salt	4.0 - 7.2	Lord (1980)
GA	Salt	5.6 - 7.3	Barnes et al. (1973)
WI	Freshwater	6.4 - 6.5	Klopatek (1975)

marshes with lower concentrations in brackish (100  $\mu\text{g/g}$ ) and freshwater marshes (23  $\mu\text{g/g}$ ). Sulfide maxima (1436  $\mu\text{g/g}$ ) appeared consistently in the upper 40 cm of the profile. Also in Louisiana, Brannon (1973) recorded total sulfide concentrations of 30-500  $\mu\text{g/g}$  with highest values in the streamside area. Combined sulfides comprised the bulk of the total sulfide budget; low levels of iron in these soils were thought to limit the accumulation of mineral sulfides. Other studies of sulfide accumulation in marsh systems (Howarth and Teal, 1977; Lord, 1980; Nedwell and Abram, 1978) have been concerned primarily with the salt marsh environment.

### 3. Iron and Manganese

The plant availability of trace metallic nutrients is strongly influenced by the soil redox condition. Iron and manganese are directly involved in reactions affecting their valence state and thus speciation in flooded soils.

One of the most important consequences of oxygen depletion in submerged soils is the reductive solvation of insoluble ferric compounds with the release into solution of ferrous ions. The oxides and oxyhydroxides of ferric iron are the minerals most affected by these reducing reactions (Gotoh and Patrick, 1974): available iron increases at the expense of reducible forms as both pH and redox potential decrease. Ponnamperuma (1972) lists the major chemical consequences of iron reduction as (i) increase in the water-soluble concentration of iron, (ii) increased availability of phosphorus and silica, and (iii) displacement of cations from the exchange complex.

In addition to oxidation reactions, precipitation of iron as insoluble ferrous sulfides ( $\text{FeS}$ ,  $\text{Fe}_2\text{S}_4$ ,  $\text{FeS}_2$ ) may significantly lower its availability, especially in iron deficient soils (Mitsui, 1954).

The behavior of manganese and iron under various Eh and pH conditions is very similar (Turner and Patrick, 1968), except that the redox kinetics of manganese are much faster than iron. Manganese is thus subject to more rapid and complete mobilization under reduced conditions. In addition manganese sulfides are more soluble than those of iron. pH is the primary factor controlling manganese availability below pH 5.

Iron and manganese availability in salt marsh soils has been addressed by Brannon (1973) and Lord (1980). Brannon found extractable iron and manganese concentrations of 50-500  $\mu\text{g/g}$  and 2-19  $\mu\text{g/g}$  respectively in Louisiana salt marshes. Mineral sediments were considered important in governing the local distribution of these metals. Lord measured "carbonate-extractable" concentrations of 100-2520  $\mu\text{g Fe/g}$  and 2-90  $\mu\text{g Mn/g}$  in a Delaware salt marsh. Both metals were subject to pronounced profile variation, higher concentrations invariably occurring in the near-surface layers. Considerable seasonality was reported in both Louisiana and Delaware studies. Further work is needed to identify patterns of availability in less saline marshes in Louisiana.

#### 4. Phosphates

Phosphate occurs in the soil in both mineral and organic combinations. Although organophosphates may constitute a substantial

reservoir in some soils (Hesse, 1962), slow mineralization precludes this component from participation in short term changes in phosphate availability in flooded soils. Of the major inorganic forms, reductant soluble phosphate (especially ferric phosphate), and those precipitated with iron or aluminum oxides and hydroxides are considered important factors in controlling the availability of this element in soil redox systems (Patrick and Khalid, 1974). A substantial increase in phosphate availability on flooding is a well established phenomenon (Shapiro, 1958). The release mechanisms were summarized by Patrick and Mahapatra (1968) as (i) the reduction of insoluble ferric phosphate to more soluble ferrous phosphate, (ii) the reduction of the hydrated ferric oxide coating on clay and silt particles resulting in the release of occluded phosphates, (iii) phosphate displacement from ferric and aluminum phosphates by organic anions, (iv) hydrolysis of ferric and aluminum phosphates, and (v) anion exchange between clay and organic anions. In addition to Eh, factors such as pH, clay mineralogy, and abundance of calcium, aluminum, iron, magnesium, and organic matter have been identified as important in controlling phosphate availability in flooded soils (Gambrell and Patrick, 1978; Ponnamperuma, 1972; Shapiro, 1958).

Figures describing the availability of various forms of phosphate in marsh sediments are summarized in Table 1. The availability of phosphates in Louisiana salt marshes was described by Brannon (1973), who found a direct correlation with the abundance of mineral sediments. Dry sediment extractions conducted by Chabreck (1972) in these marshes are of limited value in establishing the relative availability of this

nutrient under natural waterlogged conditions. The relative abundance of available phosphates in freshwater, intermediate, brackish, and salt marshes in Louisiana are thus largely unknown.

##### 5. Ammonia

Nitrogen occurs predominantly in the organic form in salt marsh soils (Haines et al., 1978; Brannon, 1973) with ammonia the most abundant mineral phase. Microbial reduction of sediments following submergence favors processes that remove available nitrogen, such that it is an important factor limiting primary productivity in flooded soils, including salt marshes (Broome et al., 1975; Sullivan and Daiber, 1974; Patrick and DeLaune, 1976). A complex array of factors control the availability of ammonia in flooded soils, including inorganic and microbially mediated processes of mineralization, fixation, denitrification, and atmospheric exchange. Ammonia may exist in solution or in exchangeable and fixed forms (Bremner, 1965; Allison et al., 1953), largely in association with the organic fraction rather than clay (Rosenfield, 1979). The reader is referred to reviews by Gambrell and Patrick (1978) and Ponnampерuma (1972) for detailed reviews of the nitrogen cycle in flooded soil systems.

Ammonia availability in marsh soils remains sparsely documented at this time. DeLaune et al. (1976) and Casselman (1979) report concentrations ranging from 1.1  $\mu\text{g/g}$  to 200  $\mu\text{g/g}$  in Louisiana salt marshes, higher concentrations occurring at depth in all cases. Inland soils contained more ammonium than streamside deposits, presumably in response to the hydrologic gradient. Haines et al. (1977) report a

value of 120-450 mg NH<sub>4</sub>-N/m<sup>2</sup> from Georgia, but unusual units preclude any comparison with other data. The relative abundance of ammonium in the various freshwater, intermediate, brackish, and saline marshes of Louisiana has not been established.

## MATERIALS AND METHODS

### A. SOILS

The soil materials used in this investigation were taken from the marshes of Barataria Basin, Louisiana. Representative sites in each of the salt, brackish, intermediate, and freshwater marshes (Fig. 1) were selected on the basis of vegetation type and soil morphology. With few exceptions, substrates to a depth of 60 cm could be classified as clayey peat, peat, or muck (Bruppacher et al., 1973). An unusual site in the salt marsh comprised mainly silty clay and silt. Soil morphology and material composition varied considerably with depth.

### B. FIELD METHODS

#### 1. Soil Samples

Marsh soils were sampled in two phases. More than 30 cores for the determination of sedimentation rate were taken from various streamside and inland locations over a 12-month period beginning April 1979, which, selected cores were also used for analysis of total content of organic carbon, nitrogen, phosphorus, iron and manganese. Duplicate cores for "fresh" chemical analysis (physicochemistry, sulfides and available nutrients) were taken at bimonthly intervals from October 1979-August 1980. These sites were located in apparently healthy inland marshes at a distance of approximately 20 m from the adjacent water body (bayou or embayment).

Thin walled aluminum core liners, 15 cm in diameter, with sharpened cutting edges were used to effect compaction-free sampling of these low-density marsh soils. The core liner was carefully twisted into the substrate to a depth of approximately 0.5 m. Surrounding soil material was removed with a shovel, and the core liner was capped before raising it to the marsh surface. The core was then sealed with a second P.V.C. cap and then secured using nylon straps. Cores were frozen on return to the laboratory and analyzed within 21 days of collection.

## 2. Vegetative Productivity

In order to establish a rough estimate of aboveground production at each marsh site, peak vegetative standing crop was sampled on one occasion during October of 1979. Live and dead standing material, cut 2 cm from the substrate surface, was collected from four randomly selected  $1/2 \text{ m}^2$  quadrats at each inland chemical study site.

Clip plot samples included the following species:

- i) salt marsh: Spartina alterniflora
- ii) brackish marsh: Spartina patens + 10% Spartina alterniflora
- iii) intermediate marsh: Spartina patens
- iv) freshwater marsh: Panicum hemitomon

The samples, stored in burlap sacks, were dried at  $60^\circ\text{C}$  until constant weight, then weighed to the nearest gram.

### 3. Eh

In situ Eh measurements were made at the inland marsh sites to complement wet chemical analyses. Measurements were made at 6 cm intervals to a depth of 43 cm in the soil profile, using an especially designed "multi-switch" platinum electrode probe, in series with a Beckman Zeromatic SS33 field multimeter and calomel half cell. Duplicate readings were made at each interval.

## C. LABORATORY METHODS

### 1. Sedimentation Rate

Frozen sediment cores were extruded from core liners and accurately cut into 2 cm horizontal sections using a band saw, and allowing for the thickness of the saw blade. Wrapped in aluminum foil, these sections were dried to constant weight at 105°C and then weighed to the nearest gram. Soils were then finely ground and stored in plastic "Ziplock" bags.

$^{137}\text{Cs}$   $\gamma$ -activity in soil sections was determined using an Ortec model 8011-1620-s coaxial, lithium-drifted germanium detector in association with an Ortec model 6240 B multichannel analyzer system. Samples were counted for a variable period determined on the basis of  $^{137}\text{Cs}$  activity, but not exceeding  $2 \times 10^4$  seconds. Calibration was achieved using standard additions of  $^{137}\text{Cs}$  to 100 gram soil samples.

### 2. Bulk Density

Bulk density was calculated from the oven-dry weight of core sections of known (uniform) volume ( $336 \text{ cm}^3$ ). These figures were

used to characterize soil types and to convert dry weight data to a volume based unit.

### 3. Organic Carbon, and (Total) Nitrogen, Phosphorus, Iron and Manganese

Analyses of organic carbon, total nitrogen, phosphorus, iron and manganese were performed on selected cores after  $^{137}\text{Cs}$  analysis was complete. Composite samples were selected at 4 cm depth intervals (0-2, 4-6, etc.) to a depth of 38 cm in the soil profiles.

#### (a) Organic carbon.

The carbon content of accurately weighed 200-300 mg samples was determined by dry combustion. Organic matter content was calculated from organic carbon using a multiplier of 1.724 (Wilson and Staker, 1932).

#### (b) Total (Kjeldahl) nitrogen.

Total nitrogen was determined by macro-kjeldahl digestion of accurately weighed 2-3 g samples (Bremner, 1965). The distillate was collected in 25 ml of 0.1 N  $\text{H}_2\text{SO}_4$ , and  $\text{NH}_4^+$  determined by back titration against 0.1 N NaOH using a mixed indicator of bromocresol green and methyl red.

#### (c) Phosphorus, iron and manganese.

Accurately weighed (0.5 g) soil samples were completely digested by the nitric-perchloric acid method (Standard Methods, 1975, pp 169-170). The digestate was diluted to 100 ml with deionized water prior to further analysis.

Iron and manganese concentrations were determined using a Fisher model 90-750 (ICAP) emission spectrometer.

Orthophosphate in the digestate was determined colorimetrically by the standard molybdate blue method (EPA-625-/6-74-003, 1974) using a Bausch Lomb model 560 UV-visible spectrophotometer. A 2 ml subsample of the original dilution produced a response within the linear range of the method.

#### 4. Physicochemistry, Sulfides, and Extractable Nutrients

Immediately prior to analysis, frozen cores were extruded and sectioned with a band saw. Composite sub-samples were taken at 6 cm intervals (0-6, 7-12, etc.) to a depth of 48 cm. Each sample was analyzed for pH, specific conductance, sulfides, and (extractable) phosphates, ammonia, iron and manganese. Pore water chlorinity measurements were made on selected cores only.

##### (a) pH, chlorinity, and specific conductance.

Pore waters were extracted from 50 ml soil samples by centrifugation under air-free conditions. Pore waters were analyzed as follows:

(i) pH: Beckman Zeromatic SS33 pH meter coupled with Corning model 97 pH electrode.

(ii) Chlorinity: Buchler model 42500 digital (potentiometric) Chloridometer.

(iii) Specific conductance: Lab-Line 11000, Lectro MHO Meter standardized against 0.1 N KCl.

## (b) Sulfides.

Sulfides were evolved and collected using a modified method of Connell (1966). Sulfides were analyzed as combined, and as "free" water-soluble forms. Combined sulfides are defined to include acid-volatile species, primarily metal sulfides but also bisulfide ion ( $\text{HS}^-$ ) present in the soil. Acid insoluble sulfides cannot be detected by the method but copper sulfide is the only common form. Free sulfide was that fraction displaced from the soil slurry by purging with argon.

Approximately 100 g of frozen soil was accurately weighed into an argon-purged erlenmeyer flask containing 100 ml of water. Free sulfides were evolved and collected in (0.1 N) zinc acetate solution by purging with argon for one hour with continuous stirring. Combined sulfides were evolved into fresh acetate solution by addition of 40 ml of concentrated sulfuric acid followed by argon purging/stirring for a further hour. Care was taken to exclude air from the system at all times.

Sulfides trapped as zinc sulfide were determined iodimetrically by reaction with excess acidified (.025 N) iodine, followed by back titration against (.025 N) sodium thiosulfate using starch indicator.

## (c) Extractable nutrients.

Soil extractions were performed using a modified method of Turner and Patrick (1968). One hundred ml of a 1N-sodium acetate solution adjusted to pH 4.5 was used in a 1:5 soil to extracting solution (weight) ratio. The extraction mixture was shaken for one

hour in polyethylene centrifuge bottles, subsequently centrifuged, and filtered through 0.45  $\mu\text{m}$  pore size filtration discs. Extracts were adjusted to pH 1.5 with concentrated hydrochloric acid and stored at 4°C until analyzed.

(i) Phosphate.

Phosphates were determined colorimetrically by the standard molybdate-blue method (EPA-625-/6-74-003, 1974) using a Bausch and Lomb model 560 UV-Visible spectrophotometer. A 10 ml aliquot of the extract produced a response within the linear range of the method.

(ii) Ammonia.

Ammonium nitrogen in the extract was isolated by steam distillation (Bremner, 1965) after adjusting the pH to 9.5. The ammonia collected was determined colorimetrically by Nesslerisation (Vogel, 1960, p 643).

(iii) Iron and manganese.

Iron and manganese were determined directly with the use of a Perkin-Elmer model 360, atomic absorption spectrophotometer.

D. DATA ANALYSIS

1.  $^{137}\text{Cs}$  Profiles

Marsh accretion rates were determined directly from  $^{137}\text{Cs}$  activity profiles: the depth of burial (mm) of the 1963 fallout peak was divided by the time interval, 16 years, from deposition (1963) to sample collection (1979) to yield the vertical marsh accretion

rate (mm/year). Pre-1963 deposits were examined for horizons corresponding to 1959 and 1954 fallout markers.

$^{137}\text{Cs}$  activities, calibrated by standard additions, were averaged over all depths counted to obtain the mean value for each soil type. Mean peak activities were calculated by averaging the maximum values corresponding to 1963.

## 2. Physicochemistry, Sulfides and Total and Extractable Nutrients

Physicochemical and chemical soil properties were averaged by depth for each marsh type, these results presented as profiles. Overall means for freshwater, intermediate, brackish and salt marshes were calculated by averaging over all depths (0-48 cm), and the variance examined using standard statistical methods (analysis of variance and Duncan's Multiple Range test). Material flux calculations are discussed in the text as appropriate.

## 3. Use of Units

### (a) $^{137}\text{Cs}$ .

$^{137}\text{Cs}$  activity profiles were expressed in volume base units, pCi/2 cm core section. Profiles were not altered by expression on a dry weight basis. Weight base data were used for comparison of activities in the various soil types examined.

### (b) Nutrients.

Depending on the context in which they are used, nutrient concentrations in marsh soils are best expressed either on a soil volume basis or on a dry weight basis (Brannon, 1973). The plant

availability of a nutrient is best expressed in units of volume ( $\mu\text{g}/\text{cc}$  wet soil) whereas dry weight concentrations are generally of greater value in the interpretation of general soil characteristics and dynamics. Unless otherwise stated, the following principles constitute the unitary basis for the ensuing discussion: (i) total nutrient concentrations are expressed on a dry weight basis in discussion of distribution amongst marsh types and in soil profiles; and (ii) extractable nutrients and sulfides are expressed primarily in units of volume for the comparison of marsh types, but dry weight units are used for the discussion of soil profiles.

## RESULTS AND DISCUSSION

### A. MARSH ACCRETION

#### 1. Rates and Processes

##### (a) General.

$^{137}\text{Cs}$  profiles from the marsh soils of Barataria Basin (Table A1, Fig. A1) indicate that rapid vertical accretion is an ongoing process in freshwater, intermediate, brackish and salt marsh soils studied. Accretion rates (Table 4) were highly variable, ranging from a maximum of 1.69 cm/y in the streamside area of the brackish marsh, to as little as 0.31 cm/y in the freshwater marsh inland areas.

Mean accretion values of approximately 1.5 cm/y in streamside and 0.65 - 0.75 cm/y in inland areas are consistent with those previously reported from the salt marshes of Barataria Basin (DeLaune et al., 1978; Baumann, 1980). Together these results confirm the relatively rapid aggradation of the marshes of the Mississippi River delta as compared to those on the Atlantic and Pacific coasts of the United States (Table 1).

##### (b) Local gradients.

Accretion was invariably more rapid in streamside or natural levee areas than in adjacent inland areas (Table 4). Fig. 2 clearly shows that the thickness of post-1963 deposits is greatest (15-30 cm) in natural levee deposits, thinning rapidly inland to attain a relatively uniform value (c 10 cm) within 40 m of the marsh creek or lakeshore. This phenomenon is a natural consequence of what

Table 4. Marsh accretion rates (mm/y) based on the 1963 Cs-137 fallout peak. (S = streamside, I = inland).

Marsh Environment	Site	N*	Accretion Rate	
			Mean	Range
Freshwater	S	2	10.6	0
	I	6	6.5	3.1-6.9
Intermediate	S	3	13.5	13.0-14.0
	I	6	6.4	3.8-10.6
Brackish	S	3	14.0	10.6-16.9
	I	7	5.9	3.8-8.1
Salt	S <sup>†</sup>	2	13.5	0
	I	6	7.5	5.6-9.4

\* Number of cores represented.

† From Delaune et al. 1979.

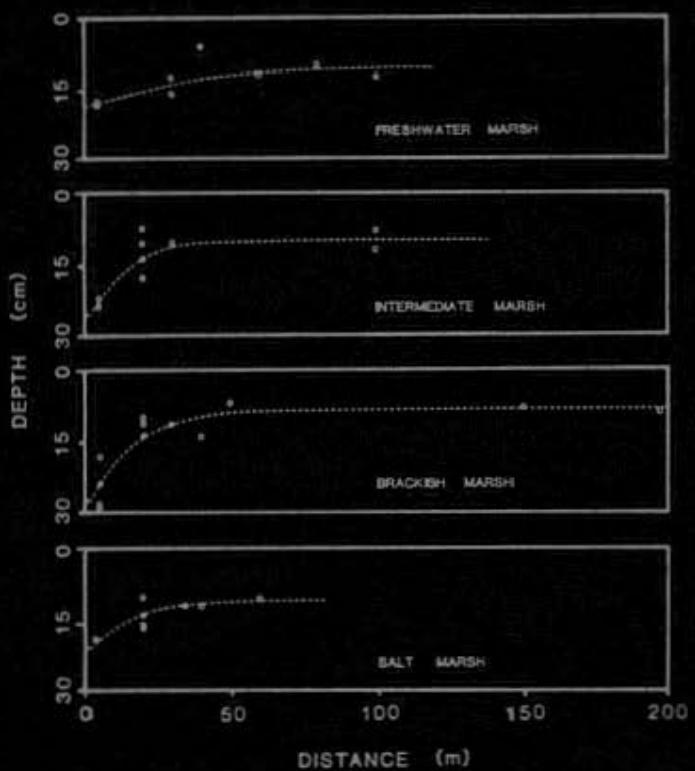


Fig. 2. Variation of  $^{137}\text{Cs}$  (1963) horizon depth with distance from adjacent water body (lake, bayou, or embayment).

is termed "lateral deposition" in response to hydraulic and associated sedimentation gradients extending from the stream banks. Although this is broadly accepted as the basis for minor topographic variations in Louisiana marshes, the relative importance of allochthonous mineral sediments and autochthonous organic sediments in maintaining marsh elevation is poorly understood. It is important to realize that the more rapid growth of natural levee deposits represents not only greater deposition of mineral sediments, but also of organic matter: bulk organic matter density is approximately constant in these soils, so that organic matter accumulation and vertical accretion vary concomitantly. It also follows that the total dry matter content and accumulation consist of a varying inorganic component superimposed on a uniform organic component. Based on similar observations, McCaffrey (1977) suggested that organic matter accumulation may sustain and control vertical accretion in marshes deficient in mineral sediments. This theory introduces the added dimension of possible physiological controls in defining local and regional variations in accretion rates.

According to Kolb and Van Lopik (1958), local sediment loading results in considerable local subsidence beneath minor landforms such as natural levees, so that accretion is manifest in a "downward" rather than "upward" growth. This concept is instrumental in understanding how streamside and inland marshes maintain the same approximate elevation despite the accretion differential as measured by <sup>137</sup>Cs dating. Assuming constant vertical accretion rates of 1.4 cm/y and 0.7 cm/y in streamside and inland areas respectively, a local topographic gradient of some 35 cm would be generated over a 50-year period.

Since such a gradient is nowhere evident in this system, it is reasonable to assume that subsidence accounts for a large part of the "apparent" accretion of the streamside marshes. This phenomenon may also explain why measured accretion rates sometimes exceed estimates of relative sea level rise (Table 1; Frey and Basan, 1978).

The characteristic variability of vertical accretion is evidenced by Fig. 2. For example, variation at the 20 m site in the intermediate marsh approximates the total lateral gradient in that system. This phenomenon severely limits the value of the typically low intensity studies reported in the literature to date; generalizations based on such data are clearly of little value unless the spatial coordinates in relation to sediment source are accurately specified, and in situ variability indicated.

(c) Regional gradients.

Accretion rates measured by <sup>137</sup>Cs dating in freshwater, intermediate, brackish and salt marsh environments were not significantly different (Table 3). That is to say, marshes throughout Barataria Basin appear to grow vertically at approximately the same rate in spite of seaward increasing gradients in hydraulic energy, mineral sediment supply (Table 4) and regional subsidence (Kolb and Van Lopik, 1978).

Although subsidence may be a major factor affecting marsh accretion and stability, the complexity of this phenomenon in Louisiana deltaic marshes negates estimates of precise rates that would apply regionally (Adams et al., 1976). In the context of marsh

accretion over the past two decades or so, slow geosynclinal subsidence, estimated at c 2 cm/century at the present Gulf shoreline (Adams et al., 1976), is likely of little consequence. Local subsidence patterns related to the age and distribution of recent underlying deltaic facies are probably of greater short term significance. Assuming marsh accretion rates reflect changes in relative water level, then the results presented here imply that, at least in the 16 years since 1963, subsidence has occurred almost uniformly in freshwater, intermediate, brackish, and salt marsh environments studied in Barataria Basin. Comparison of limited tide gauge data from Bayou Chevruiil in the freshwater marsh to the north and Bayou Rigaud in the salt marsh to the south (Baumann, 1980) support this hypothesis.

In view of the pronounced local variation in vertical accretion rate identified in response to local hydrologic and sedimentary gradients (b, above), the absence of similar regional correlations might appear somewhat paradoxical at first. However, this pattern can be better understood by considering that the study transect traverses four different vegetative marsh types in which the response to mineral sediment deficiency may vary considerably. In the less saline intermediate and freshwater environments, surface elevation is apparently sustained to a large, but variable extent by accumulation of organic detritus and the structural and buoyant properties of that organic matter. Organics comprise in excess of 80% of the dry weight of some of these soils and as such constitute the major structural component. "Flotant" marshes are a common feature of less saline environments. McCaffrey (1978) presents strong evidence to suggest that plant

production may sustain accretion in salt marshes lacking an abundant silt source. "The remarkable linearity of the organic matter accumulation curve shows that organic growth has uniformly occupied new living space created by the rise in relative sea level which suggests that peat formation rather than particle deposition is controlling accretion."

In view of the flotant phenomenon (Kolb and Van Lopik, 1958) in which the marsh surface apparently floats on an aqueous slurry, the relationship between rates of subsidence or mean sea level rise and the vertical accretion of marshes, is questionable. Under these circumstances, the apparent accretion rate measured by <sup>137</sup>Cs and analogous dating techniques, must be largely controlled by diagenetic consolidation processes (organic matter decomposition and compaction) within the flotant root mat itself, independent of changes in mean sea level. And since the distribution of flotant is mainly limited to the less saline (intermediate and freshwater) marshes, cross-environmental comparisons such as described in this study are surely of limited significance.

## 2. Methodology and Limitations

<sup>137</sup>Cs has been successfully applied to recent geochronological investigations in deep-water marine (Schreiber, 1968), lacustrine (Pennington et al., 1973) and salt marsh environments (DeLaune et al., 1978). The present investigation further illustrates the utility of the method in highly reduced, rapidly accreting organic substrates. The 1963 peak in <sup>137</sup>Cs profiles (Fig. A1) was, with few exceptions,

well defined; neither bioturbation nor chemical remobilization appear to have significantly altered the position of this most recent peak. However, poorly defined profiles at greater depth did not justify identification of earlier dated horizons (1954, 1959). Progressive burial and incorporation in the organic slurry at depth in many of these soils may lead to disruption of the <sup>137</sup>Cs profile through gas-related mixing, and density fractionation of sedimentary particulates.

Because land subsidence as a result of sedimentary consolidation is a time varying process in Louisiana (Adams et al., 1976), marsh accretion/ subsidence rates based on a 1963 dated horizon may not be extrapolated to earlier time periods. Furthermore it should be understood that these values represent an integrated 16 year rate; accretion in the short term may proceed more rapidly, or more slowly than these figures suggest.

The major disadvantage of the <sup>137</sup>Cs dating technique in respect of the present study was the sample count time (c 5.5 hours per core section). This is considered an important logistic limitation of an otherwise broadly applicable technique. Development of electronic hardware (automated sample changers) will significantly extend the utility of the method. As a result of such limitations, insufficient samples were analyzed to adequately characterize the extremely variable accretion rates of these marsh substrates. Mean accretion rates (Table 4) do not pretend to represent all marshes in Barataria Basin or in Louisiana, but instead are offered as representative of selected "typical" situations. The accuracy of these estimates is further

limited by the error inherent in the section thickness (2 cm) especially where accretion is slow: maximum error introduced in this way is estimated at 20%.

It has been suggested that changes upon freezing might be a source of error in this determination, but despite considerable aqueous expansion, the vertical disposition of fibrous peaty substrates was not altered significantly.

Finally, it is the authors opinion that post-depositional chemical remobilization does not significantly alter the vertical distribution of  $^{137}\text{Cs}$  in the short term, in this marsh system. In the absence of vertical (downward) leaching, simple diffusion would tend to equally distribute  $^{137}\text{Cs}$  over the entire profile: the relative position of peak activity is not altered by this process alone. Except possibly in the streamside areas, leaching is considered unimportant in these uniformly saturated soils.

### 3. Activity of $^{137}\text{Cs}$ in the Marsh Environment

Greatest  $^{137}\text{Cs}$  activity was measured in the low density freshwater marsh soils decreasing seaward toward the salt marsh (Table A2). Assuming uniform fallout over the study area, this pattern is possibly an artifact related to (i) the greater attenuation of radiation by high density mineral soils of higher molecular weight, and (ii) the geometric dissipation of radiation by the larger volume of high density more mineral soils when dry; the bulk density of marsh soils increases progressively seaward along the environmental transect

from freshwater marshes to the Gulf Coast (Table 4). The remobilization effects of tidal flushing and ionic strength are unknown.

B. DISTRIBUTION AND FLUX OF ORGANIC CARBON TOTAL (KJELDAHL) NITROGEN, TOTAL PHOSPHORUS, TOTAL IRON AND TOTAL MANGANESE

The sedimentary record constitutes a simple tool for the investigation of nutrient and contaminant dynamics in flooded soil systems, which has been used to some advantage in recent salt marsh studies (McCaffrey, 1977; DeLaune et al., 1981; Nixon, 1980). The ensuing section describes the spatial (environmental) and soil profile distribution of mineral sediments, organic carbon, nitrogen, phosphorus, iron, and manganese in the various freshwater, intermediate, brackish, and salt marshes of Barataria Basin. Data recorded in appendix B are summarized in Table 5, and depicted graphically in Figs. 3 through 13. This discussion is directed mainly at the inland soils since they are spatially more extensive, and therefore more representative of this marsh system.

1. Bulk Density and Organic C

(a) Distribution.

Mean bulk density integrated over a depth of 45 cm was greatest in the salt marsh with progressively lower values inland from the coast (Fig. 3). Values ranged from more than  $0.3 \text{ g/cm}^3$  in some salt marshes to as little as  $.05 \text{ g/cm}^3$  in freshwater and intermediate environments. Analysis of variance (Table B2) showed considerable

Table 5. Total concentrations and accumulation rates of organic carbon, nitrogen, phosphorus, iron and manganese.

Marsh Type	Vertical Accretion Rate Site <sup>a,b</sup> (cm/y)	Organic Carbon			Nitrogen			Phosphorus			Iron			Manganese		
		C <sub>D</sub> (%)	C <sub>B</sub> (g/m <sup>2</sup> /y)	C <sub>N</sub> (%)	C <sub>D</sub> (g/m <sup>2</sup> /y)	C <sub>B</sub> (%)	C <sub>N</sub> (g/m <sup>2</sup> /y)	C <sub>D</sub> (g/m <sup>2</sup> /y)	C <sub>B</sub> (%)	C <sub>N</sub> (g/m <sup>2</sup> /y)	C <sub>D</sub> (μM/g)	C <sub>B</sub> (%)	C <sub>N</sub> (g/m <sup>2</sup> /y)	C <sub>D</sub> (μM/g)	C <sub>B</sub> (%)	C <sub>N</sub> (g/m <sup>2</sup> /y)
Freshwater	S 1.06 I .63±.18	.11±.03 .09±.01	23.1±6.2 29.6±3.1	250 16±40	1.5±.3 1.8±.2	16 9±3	15.7±1.5 16.7±1.0	9.7±1.71 9.4±.82	1.0 .5±.1	1472±3912 995±2007	19 7±2	144±52 114±12	.15 -.07±.02			
Intermediate	S 1.13±.09 I .64±.16	.18±.06 .08±.01	18.6±3.4 29.4±2.4	415±28 156±38	1.23±.18 2.00±.16	10±3 11±3	15.5±2.3 15.0±1.0	6.68±.48 7.63±.24	1.5±.1 .4±.1	1869±2672 10079±1590	46±3 6±1	69±12 60±7	.17±.01 -.03±.01			
Brackish	S 1.40±.36 I .59±.12	.27±.02 .14±.01	12.5±1.6 23.7±1.9	493±121 183±37	.66±.04 1.31±.10	25±7 10±2	18.8±2.0 18.4±1.3	6.24±.35 6.64±.18	2.4±.6 .5±.1	20831±1251 11830±1152	79±30 10±2	100±12 77±7	.38±.09 -.06±.01			
Salt	S* 1.35 I .75±.14	.25 .29±.06	11.2 11.4±1.6	393 200±37	.62 .69±.10	21 11±2	18.1 18.9±1.9	4.89 6.68±112	1.1±.2 1.1±.2	16837±655 16837±655	29±26 29±26	60 111	.37 -.37			

<sup>a</sup> From DeLaune et al. (1979).

<sup>b</sup> S = Streamside, I = Inland.

overlap in the bulk density of most inland areas but with significantly higher values in the salt marsh system.

Organic carbon (dry weight %) varied significantly along the transect from saline to freshwater marsh environments, with progressively higher concentrations in the less saline environments to the north of the basin (Fig. 4). Values ranged from a maximum of almost 40% carbon (or 86% organic matter) in some intermediate marsh cores to as little as 5% in the salt marsh system. Analysis of variance (Table B2) showed similarly high concentrations in freshwater and intermediate soils, with lower values in the brackish substrate and again in the salt marsh.

Mineral sediments thus constitute a progressively greater fraction of soil solids south along the transect. This seaward gradient is a direct consequence of the hydrologic regime in Barataria Basin: whereas high energy marine processes provide an abundance of reworked mineral sediments to the salt marshes, this energy is progressively attenuated and dissipated inland from the coast, with a consequent decrease in the suspended load of floodwaters. Influx of fluvial sediments occurs locally in the upper reaches of the basin as a result of precipitation runoff from adjacent uplands. Thus organic plant remains constitute an increasing fraction of soil solids as the marine influence diminishes inland from the coast, and as such are of greatest structural significance in low density freshwater and intermediate environments.

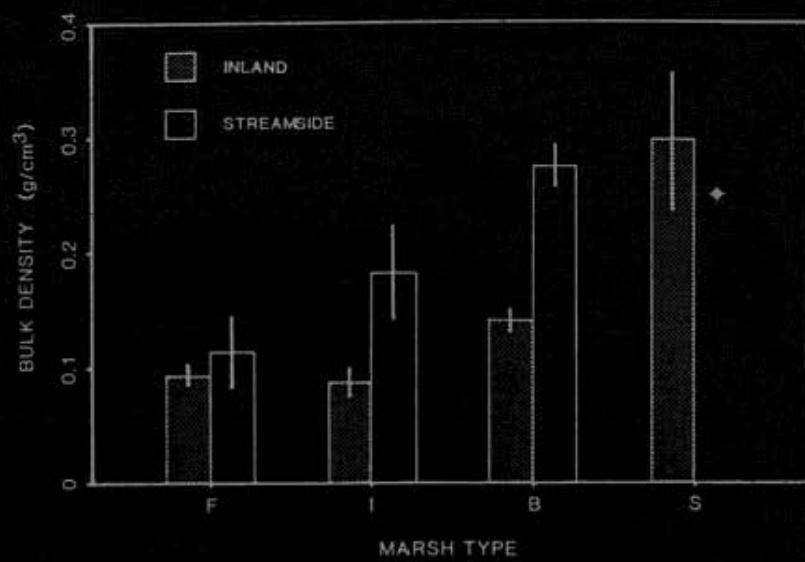


Fig. 3. Variation of bulk density with marsh type. (♦ from Delaune et al., 1981). Error bars represent 95% confidence limits.

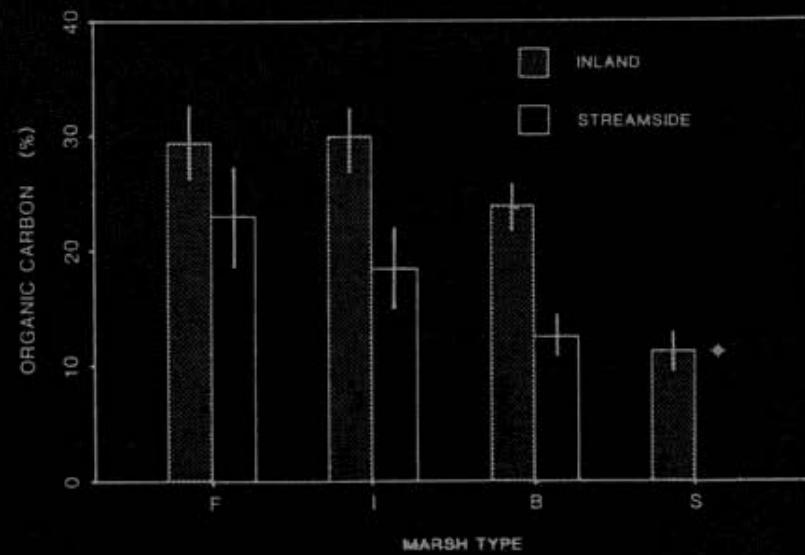


Fig. 4. Variation of organic carbon with marsh type. (♦ from Delaune et al., 1981). Error bars represent 95% confidence limits.

Bulk density ( $D$ ) was related to the organic carbon content ( $C_D$ ) in a non-linear, asymptotic fashion (Fig. 5), described by the relation:

$$D = .091 + 1.027 \times e^{-1.178(C_D)}$$

The asymptote at  $C = 100\%$  yields a value of  $\approx 0.09 \text{ g/cm}^3$  for the bulk density of pure organic matter, and a mineral sediment bulk density of  $1.118 \text{ g/cm}^3$  is predicted by the Y intercept.

Although empirically useful, the expression of bulk density as a function of the dry weight organic carbon content is misleading. The bulk organic carbon content does not change significantly (ranging between  $0.020 \text{ g/cm}^3$  and  $.024 \text{ g/cm}^3$  over all environments). It is therefore the abundance of mineral sediments that define bulk density differences in this system, and not organic matter.

(b) Flux.

Material accumulation is a major feature of rapidly accreting marsh systems such as are found in Barataria Basin. The rate at which this occurs may be estimated from the vertical accretion rate ( $R$ ) and the bulk concentration of the material ( $C_v$ ) using the expression

$$A = C_v \cdot R = \overline{C^D} \cdot D \cdot R$$

where  $\overline{C^D}$  is the steady state dry weight concentration and  $D$  the bulk density.

Mean values (Table 5) indicate a general seaward increase in organic matter accumulation, but with considerable overlap within

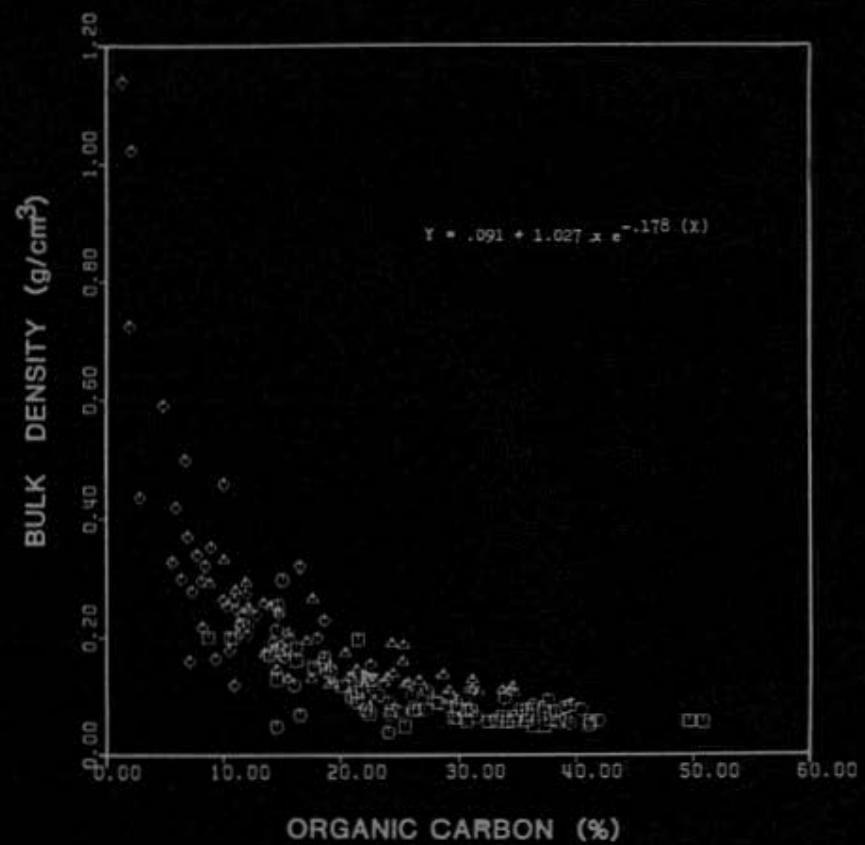


Fig. 5. Variation of bulk density with organic carbon. (□ freshwater marsh, ○ intermediate marsh, △ brackish marsh, ◊ salt marsh).

the confidence limits of the calculation. Comparable flux rates in all environments are predicted on the basis of the approximately uniform accretion rate and C density, referred to in preceding sections. Values computed for streamside areas were two to three times higher than adjacent inland areas, consistent with the vertical accretion gradient.

Clip plot peak (October) standing crop (PSC) estimates from the present study, combined with the annual production (AP) and peak standing crop estimates of Hopkinson et al. (1978) from Louisiana, suggest that organic matter accumulation (A) is approximately equivalent to 3% and 14% of the aboveground production in brackish and salt marshes respectively.<sup>1</sup> These values are substantially lower than those suggested by DeLaune et al. (1981) for a Barataria Basin salt marsh. However the uncertainty in these calculations is large: annual production estimates for Louisiana salt marshes vary considerably (Hopkinson et al., 1978; White et al., 1978) and little is known about the relative contribution of belowground and aboveground components to

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<sup>1</sup> Calculations based on values summarized below, assuming % organic matter = 1.72 x % organic carbon (Wilson and Staker, 1932). No data are available for freshwater and intermediate environments.

Marsh Type	PSC* (Hopkinson et al., 1978)	A* (From present study) (g/m <sup>2</sup> )	A* (g/m <sup>2</sup> /y)	(AP/PSC)PSC* $\times 10^2$
Brackish	2.5	2508	182.9	3
Salt	1.7	1648	199.5	14

that impounded by accretionary processes. Evidence to date (Payonk, 1975; Stroud, 1976; McCaffrey, 1977) suggests that roots constitute the primary organic building block of tidally inundated marsh deposits, but this relationship is likely to vary considerably with hydrology and local soil conditions, affecting both root production and detrital export.

The accumulation of mineral sediments in inland areas increased from  $412 \text{ g/m}^2/\text{y}$  in the freshwater marsh to as much as  $1927 \text{ g/m}^2/\text{y}$  in the salt marsh, with consistently higher values in streamside areas. This pattern provides the basis for the accumulation of nutrients with primarily mineral associations, which are discussed further in subsequent sections.

## 2. Total (Kjeldahl) Nitrogen and Carbon:Nitrogen Ratio

### (a) Distribution.

Total nitrogen (dry weight %) showed a significant decrease seaward along the transect, ranging from more than 2% in freshwater and intermediate marshes to as little as 0.6% in high-density salt marsh substrates (Fig. 6a, Table B2). These figures are comparable with literature values from a variety of wetland systems (Table 2). Total nitrogen profiles were erratic and unpredictable.

The distribution of total nitrogen was highly correlated with organic carbon ( $R = 0.87^{**}$ ) and can be described by the linear relation (Fig. 6b):

$$N = 0.140 + 0.055 C$$

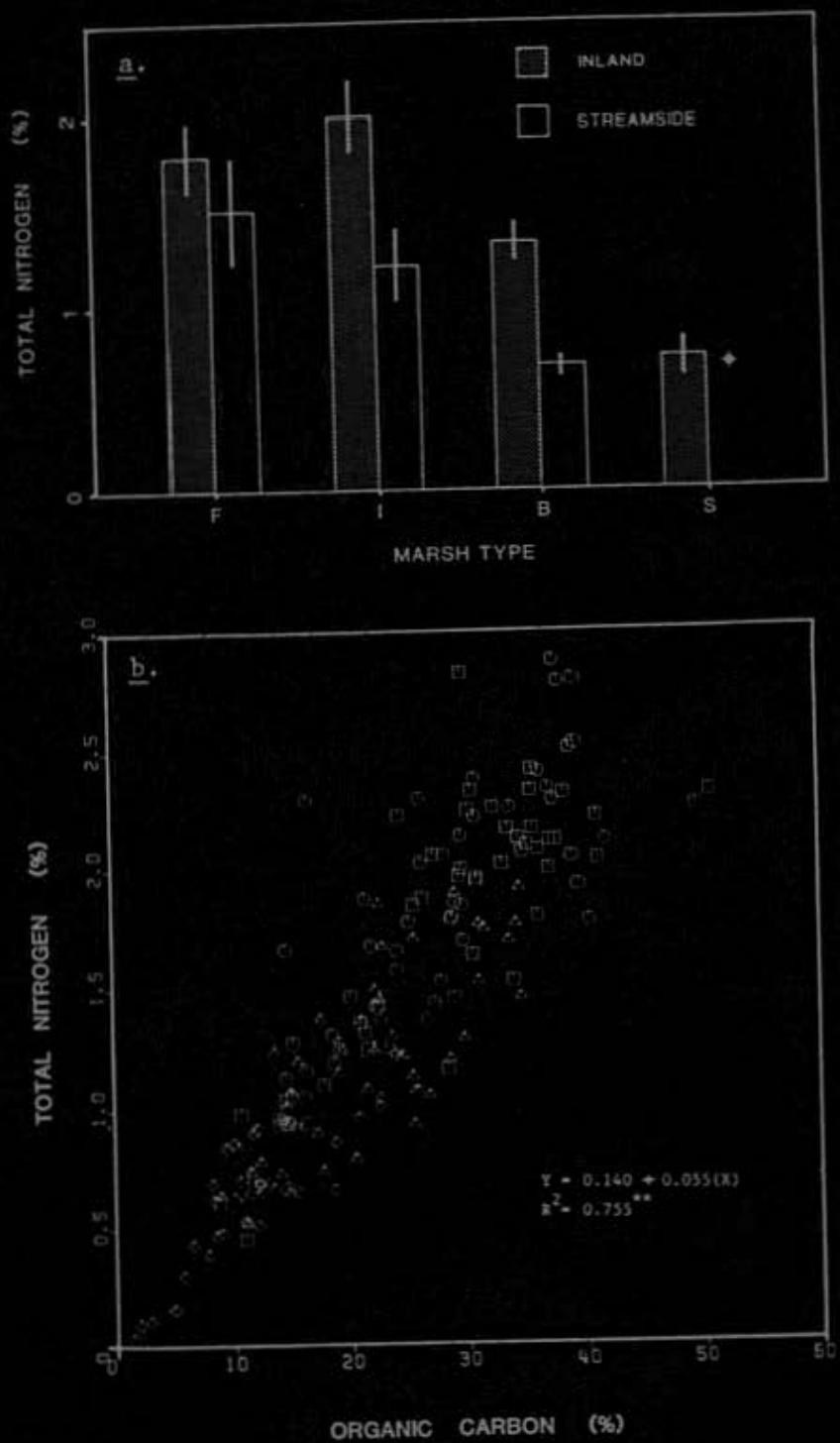


Fig. 6. Total nitrogen. a) Variation with marsh type; (♦ from Delaune et al., 1981). b) Variation with organic carbon; (■ freshwater marsh, □ intermediate marsh, ▲ brackish marsh, ○ salt marsh). Error bars represent 95% confidence limits.

This result confirms the primarily organic association of nitrogen in the soils of Barataria Basin. Slow anaerobic mineralization of organic matter favors nitrogen immobilization by incorporation into plant tissue, thus contributing to an explanation of the observed deficiency in available nitrogen in salt marsh soils (DeLaune et al., 1976).

The slope of the above equation yields a C:N ratio of 18.2 integrated over all environments. Ratios computed separately for each environment (Table 5) showed an increase from freshwater and intermediate soils (c 15.5) to brackish and salt marshes (c 18.5), but statistical methods suggest that this was not a real trend. Broadly based studies by Chabrek (1972) indicate similar values (18-20) for all marsh environments in the Louisiana deltaic plain. However, that such a trend is real, is not unreasonable in view of the ecotypic gradient involved, encompassing differences in plant species and soil biochemistry (especially sulfate metabolism).

(b) Flux.

As with organic carbon, nitrogen accumulation proceeds at approximately the same rate in all marsh environments studied (Table 5), with deposition in natural levees exceeding that in inland areas by a factor of about 2. Nitrogen accumulation rates ranging from  $9 \text{ g/m}^2/\text{y}$  to  $30 \text{ g/m}^2/\text{y}$ , (freshwater and intermediate marshes, respectively) were comparable to those reported by DeLaune et al. (1981) for the salt marshes of Louisiana. This pattern is consistent with the primarily organic association of nitrogen, and the relatively uniform distribution of organic matter in soils throughout the basin.

### 3. Total Phosphorus

#### (a) Distribution.

Expressed on a dry weight basis, total phosphorus varied predictably along the marsh transect, concentrations declining from the freshwater marsh seaward through intermediate, brackish, and salt marsh environments (Fig. 7a). Statistical methods indicate that this trend is real, but that considerable overlap exists between adjacent intermediate, brackish and saline systems. Approximately 20% of the observed variance was predictable on the basis of soil depth, (estimated from sums of squares in analysis of variance model including independent variables 'marsh type' and 'depth'). Soil profiles showed significant enrichment in surface layers (Fig. 8a).

The distribution of total phosphorus (P) was positively correlated with organic carbon ( $C_D$ ) and can be described by a linear function (Fig. 7b):

$$P = 526 + 9.5(C_D) \quad R = 0.36^{**}$$

Together with a significant negative correlation with total iron ( $R = -.39^{**}$ ), these results suggest that phosphorus may be largely incorporated in organic matter in these soils. It is important to realize however, that this element is subject to post-depositional remobilization resulting in an apparent accumulation in surface soil layers which are largely organic. Thus it is possible that the weak correlations observed are coincidental, and do not reflect immobilization of phosphorus by organic incorporation.

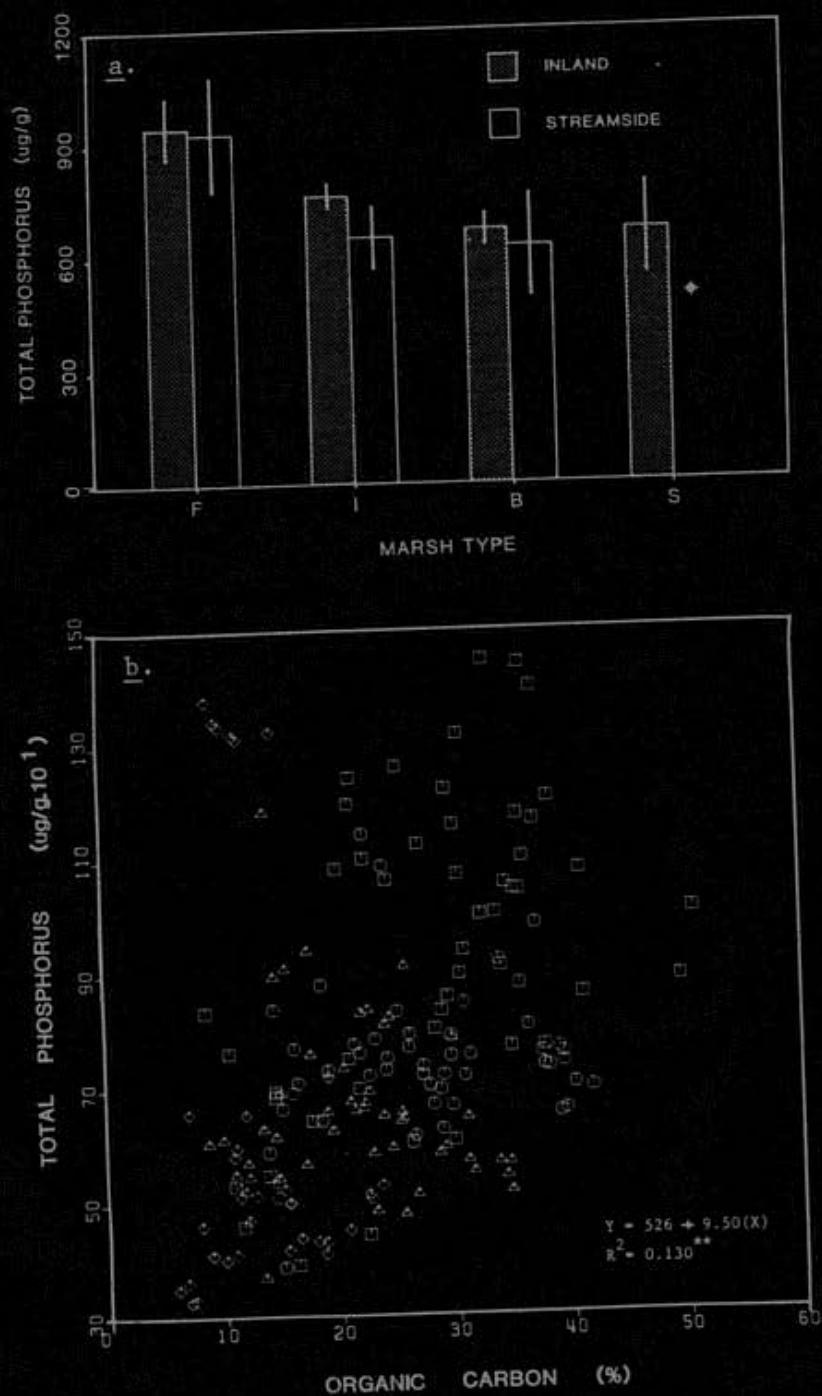


Fig. 7. Total phosphorus. a) Variation with marsh type; ( $\blacklozenge$  from Delaune et al., 1981). b) Variation with organic carbon; ( $\blacksquare$  freshwater marsh,  $\circ$  intermediate marsh,  $\blacktriangle$  brackish marsh,  $\diamond$  salt marsh). Error bars represent 95% confidence limits.

On the basis of the above correlation, it is reasonable to attribute the seaward decline in total dry weight phosphorus content to the parallel distribution of organic matter. But there is evidence from other estuarine systems (Fitzgerald, 1978) to suggest enhanced remobilization of this element in tidally inundated marsh soils relative to subtidal and supratidal (fluvial) deposits. The situation is further complicated in Barataria Basin by local variations in sediment source, and contamination by runoff from agricultural uplands adjacent to the freshwater marsh (Kemp, 1975). Freshwater marshes appear to be relatively enriched in both phosphorus and manganese.

Higher near-surface concentrations of phosphorus, apparent in all marsh environments sampled (Fig. 8; Table B1), is generally accepted as evidence for the remobilization at depth of mineral phosphates from flooded soil-plant-water systems. Recent publications, including Reimold (1972) and DeLaune and Patrick (1980), have been inclined to emphasize the role of Spartina in affecting phosphorus, depletion in subsurface sediments in salt marshes. Similar concentrations in surface layers and incoming sediments (DeLaune et al., 1981) support a model of mainly biological remobilization. However, such phosphorus profiles have been identified in sediments free of macrophytes: Carrigan and Flett (in press, 1981) showed that nearsurface accumulations of phosphorus in lake sediments probably result from rapid reductive remobilization at depth, coupled with upward diffusion, and oxidative precipitation with iron or manganese at or near the sediment-water interface. This mechanism has been invoked by Lord (1980) to

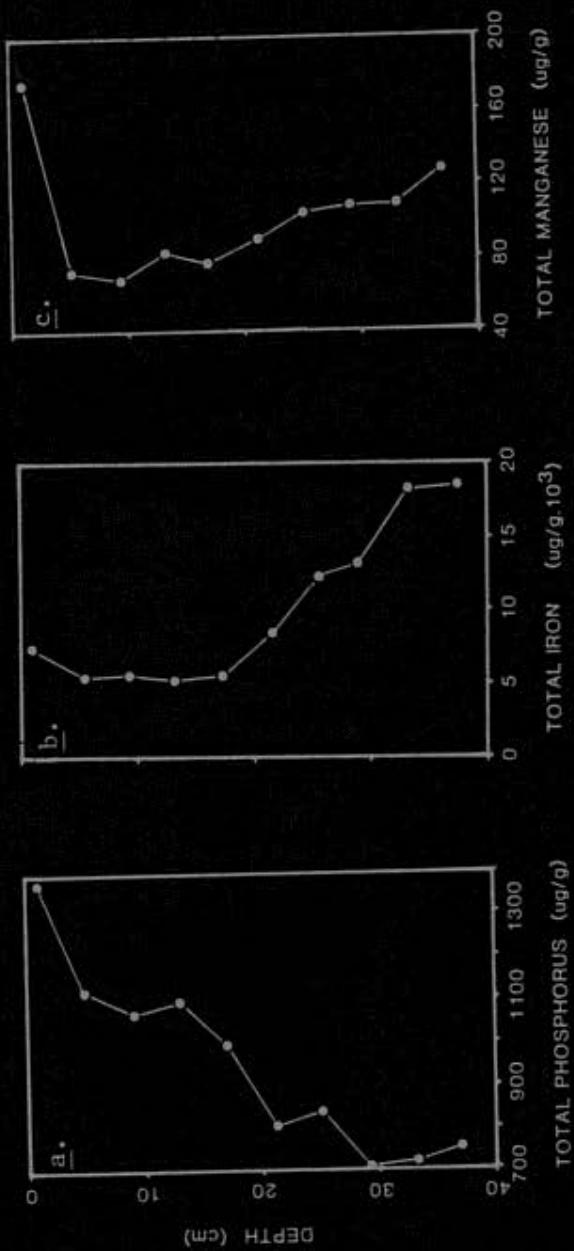


Fig. 8. Total nutrient profiles. (a) Phosphorus (b) Iron (c) Manganese.

account for the distribution in salt marsh soil profiles. If this is true, an enrichment of surface layers relative to incoming sediments may be anticipated; the limited data available for salt marshes (DeLaune et al., 1979) are somewhat inconclusive in this respect. Thus, while there is little doubt that uptake by macrophytes is a significant pathway by which phosphorus export is achieved, the relative role of physical and biotic processes remains to be decided.

(b) Flux.

Phosphorus accumulation as a result of marsh accretion proceeds most rapidly in the salt marsh with progressively slower rates in less saline environments in Barataria Basin (Table 5). The relative accumulation rates are given by the ratio 2.7: 1.3: 1.0: 1.2 for saline: brackish: intermediate: freshwater marsh inland areas respectively. Given the conventionally mineral association of phosphorus, and the relatively uniform vertical accretion rate, the positive flux is logically defined by the flux of mineral sediments to the marsh surface (section 1b above).

Significantly higher concentrations in the uppermost soil layer support the concept of rapid post-depositional remobilization and migration of phosphorus in reduced marsh soils. Assuming a steady state residual concentration ( $C_h^D$ ) at depth in the sediment, the fraction (F) lost by this process may be estimated if the concentration in incoming sediments ( $C_s^D$ ) is known:

$$F = 1 - \frac{C_h^D}{C_s^D}$$

For the salt marsh (from Table B1):

$$\begin{aligned} F &= 1 - 650/830 \\ &= 0.22 = 1/5. \end{aligned}$$

where  $C_S^D$  and  $C_h^D$  are estimated from table B2, and  $C_S^D$  assumed equal to the surface layer concentration. Thus approximately 1/5 of the phosphorus brought into salt marsh systems by sedimentation is remobilized and presumably re-enters the floodwaters in a more available form. Equivalent figures calculated for brackish, (.36 - .49) intermediate (.31 - .32) and freshwater (.33 - .55) marshes, though admittedly crude, indicate some uniformity in the extent to which depletion proceeds. This compares with Nixon's (1980) estimate of .30 - 0.60 for the salt marsh environment.

Given the vertical accretion rate of the marsh, the rate of phosphorus remobilization ( $L$ ) may be estimated using the expression:

$$L = C_S^D \cdot D \cdot F \cdot R.$$

where  $C_S^D$ , D, F and R are defined as above. Using the figures in Tables 5 and B2, loss rates of .24 and .26, .16 and .68, .24 and 2.32, and  $\underline{c}$  .39 g/m<sup>2</sup>/y were calculated for inland and streamside areas of the freshwater, intermediate, brackish, and salt marshes respectively. These results suggest a more rapid remobilization in streamside deposits, especially in more saline tidally inundated soils; values compare favorably with previous estimates of phosphorus flux from salt marshes (DeLaune and Patrick, 1980; Nixon, 1980), and thus support the concept that remobilization processes in flooded marsh soils may constitute an

important mechanism for the regeneration of available phosphorus in estuarine and nearshore waters.

#### 4. Total Iron and Manganese

##### (a) Distribution.

Total iron and manganese were generally most abundant in high density salt marsh sediments, declining inland through the brackish and intermediate systems (Figs. 9a, 10a). While iron was lowest in the freshwater marsh, manganese was enriched in this soil. Natural levee deposits invariably yielded more metals than adjacent inland areas. Analysis of variance (Table B2) showed significant variation on the environmental transect, but with considerable overlap between adjacent marshes.

The soil profile distributions of iron and manganese were characteristically different. Iron was highly variable and largely independent of the depth of burial; slight upper layer accumulations apparent in cores from the freshwater and intermediate marshes only. By contrast, abruptly higher manganese concentrations in surface layers were consistently identified in all marshes with predictably lower concentrations at depth (Table B1, Fig. 8). Approximately 15% of the variance observed in manganese profiles could be predicted by depth.

The occurrence of iron (Fe), was largely explained by the distribution of mineral sediments as measured by bulk density (D) (Fig. 9b):

$$\text{Fe} = 7168 + 41430 \cdot D \quad R = .45^{**}$$

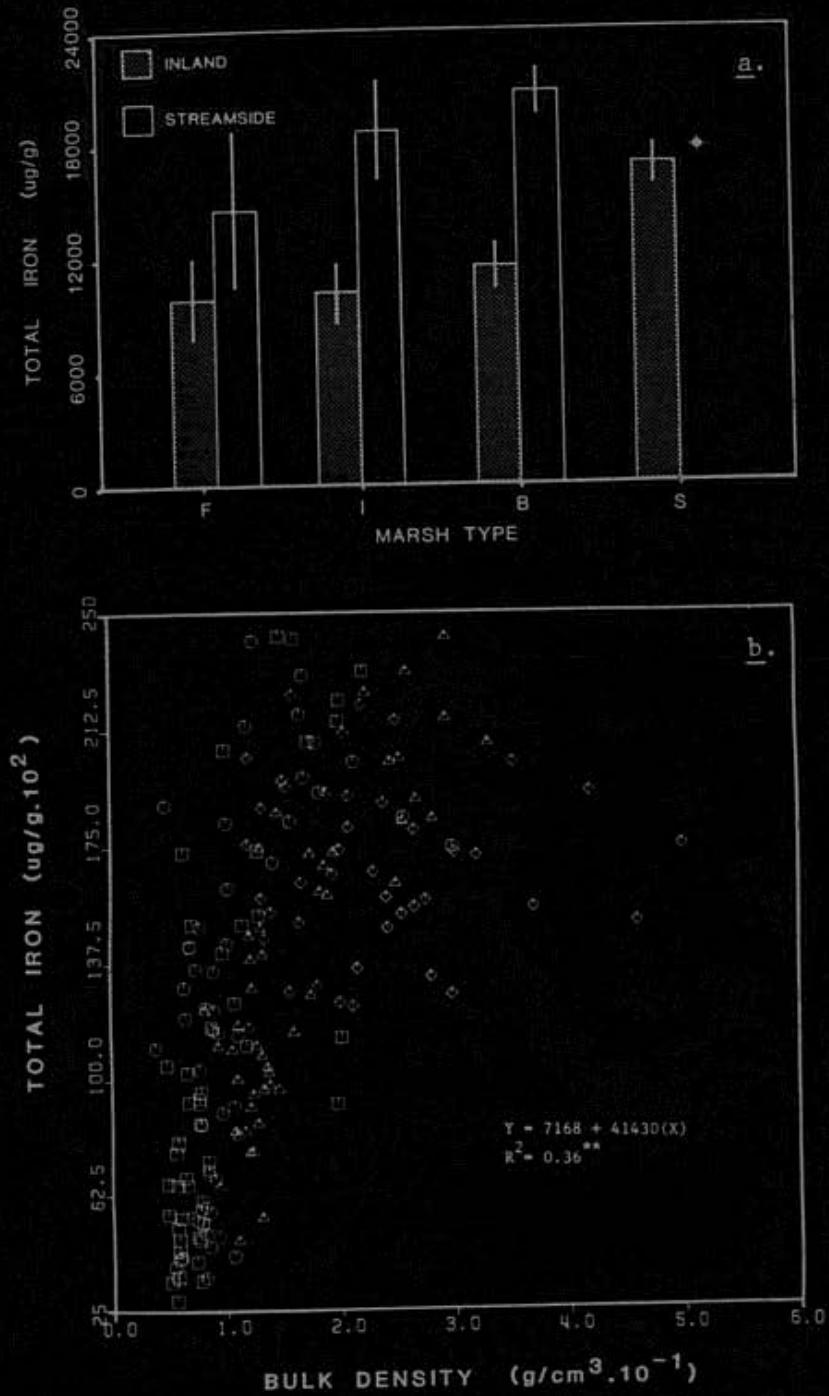


Fig. 9 . Total iron.(a) Variation with marsh type; ( $\blacklozenge$  from Delaune et al. 1981).(b) Variation with bulk density; ( $\blacksquare$  freshwater marsh,  $\circ$  intermediate marsh,  $\blacktriangle$  brackish marsh,  $\diamond$  salt marsh). Error bars represent 95% confidence limits.

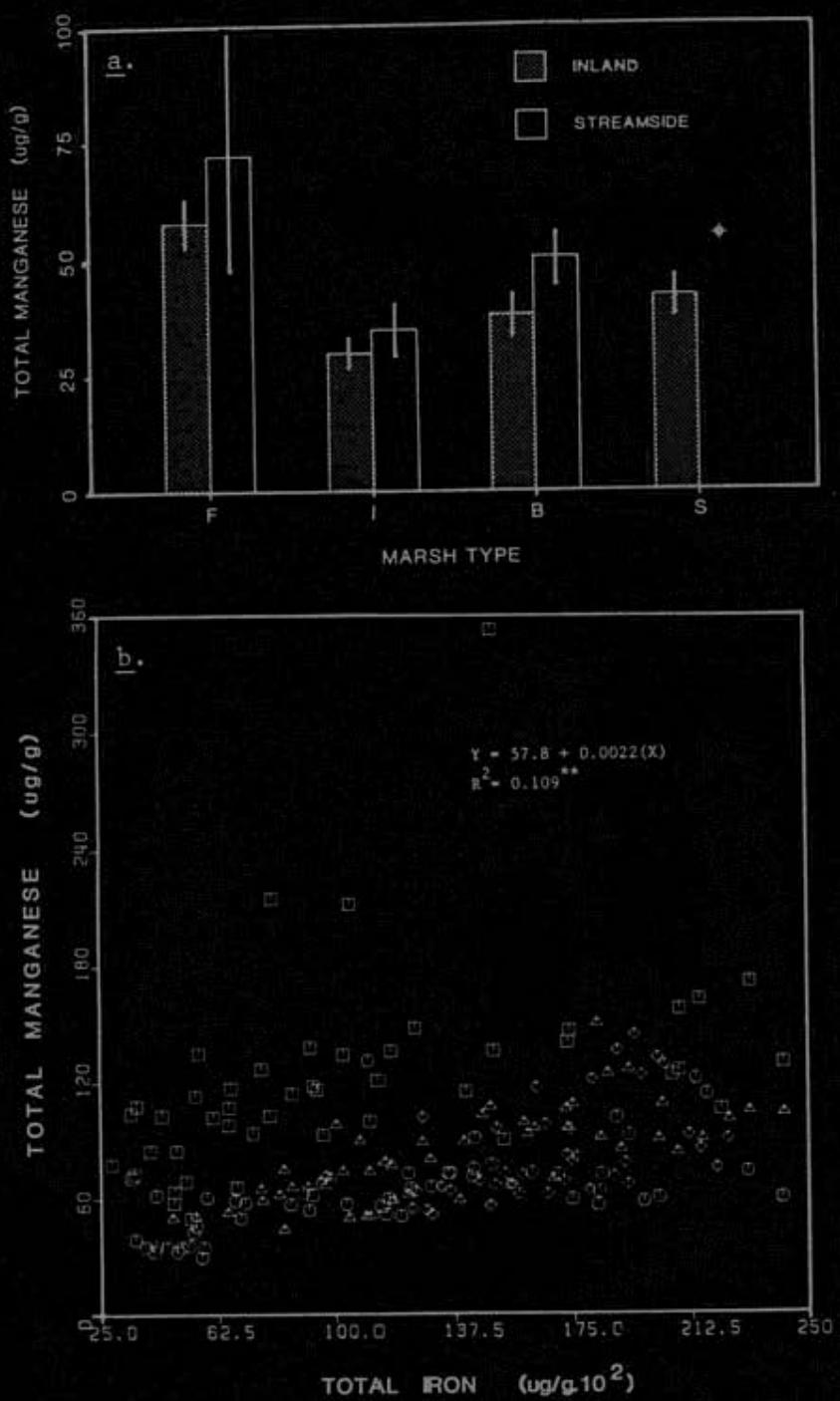


Fig. 10. Total manganese. (a) Variation with marsh type; (♦ from Delaune et al., 1981). (b) Variation with total iron; (□ freshwater marsh, ○ intermediate marsh, ▲ brackish marsh, ◆ salt marsh). Error bars represent 95% confidence limits.

Manganese was not significantly correlated with bulk density, also showing a relatively uniform distribution with respect to iron (Fig. 10b).

$$\text{Mn} = 57.8 + .0022 \text{ Fe} \quad R = 0.33^{**}$$

Contrasting behavior of manganese and iron under reduced, sulfidic conditions is expected on the basis of general chemical properties. Results presented here conform to this expectation. The gross environmental distribution of both metals is governed primarily by their mutual mineral association and the seaward increase thus reflects the parallel gradient in mineral sediment supply. Correlation between iron and bulk density would be expected if inorganic material was the source of iron delivered to the marsh surface, and if that element was not subsequently remobilized. The observed surface accumulation of manganese is consistent with a model of remobilization of solid phases at depth coupled with upward diffusion and oxidative precipitation at, or loss from, the surface. Reduction of residual metal oxides results in the release of soluble Mn<sup>II</sup> and Fe<sup>II</sup>. But whereas iron is effectively immobilized under reduced sulfidic conditions, by precipitation as highly insoluble FeS and ultimately FeS<sub>2</sub>, manganese remains predominantly in solution. Hence, elements such as iron, not significantly remobilized, primarily reflect the depositional history rather than the diagenetic history of accumulation as exemplified by manganese. These results conform with evidence of post-depositional migration of manganese from coastal marine sediments (Thompson et al., 1975), lake

sediments (Robbins and Callendar, 1975), and salt marsh soils (McCaffrey, 1977; Lord, 1980).

Slightly elevated concentrations of iron in nearsurface layers in freshwater and intermediate marsh soils (Table B1) suggest significant post-depositional mobility. This behavior is in agreement with the above chemical constraints as applied to sulfide poor, reduced soil systems, such as typify less saline marsh environments in Barataria Basin.

(b) Flux.

The rate of accumulation of sedimentary iron and manganese (Table 5) increased seaward, parallel to the dry weight concentration of both metals, and mineral sediments. The accumulation ratios for iron (1.4:1.0:1.9:7.0) and manganese (2.3:1.0:2.0:6.0) in freshwater, intermediate, brackish and salt marshes respectively reflect the positive deviation of the freshwater marsh sites from the overall pattern. Locally higher manganese concentrations in the freshwater marsh may be related to agricultural enrichment of adjacent upland soils. Kemp (1975) documented enhanced nutrient levels in runoff waters entering the same hydrologic unit. Salt marsh iron ( $29 \pm 6 \text{ g/m}^2/\text{y}$ ) and manganese ( $.14 \pm .03 \text{ g/m}^2/\text{y}$ ) accumulation rates calculated in the present study are comparable to those of DeLaune et al. (1981) ( $20 \text{ g/m}^2/\text{y}$ ;  $.13 \text{ g/m}^2/\text{y}$ ) in Barataria Basin, and the manganese influx rate of McCaffrey (1977) in a New England *Spartina patens* marsh ( $.22 \pm .06 \text{ g/m}^2/\text{y}$ ).

Markedly higher manganese concentrations in surface layers than at depth in soil profiles indicate significant

post-depositional remobilization. Using the approach developed for phosphorus (section 3b), it appears that 20-70% of the manganese deposited on the marsh surface is remobilized at depth and lost to floodwaters. Manganese profiles, indicate greatest subsurface depletion in streamside areas, and a general increase in the extent of depletion from saline (18%) through brackish (35-40%), intermediate (44-47%), and freshwater (50-69%) marshes. Manganese loss rates in streamside and inland areas of freshwater, intermediate, brackish, and salt marshes were 0.27 and 0.04, 0.13 and 0.02, 0.19 and 0.03, and .04 respectively. These values compare with McCaffrey's (1977) of  $0.1 \pm .04 \text{ g/m}^2/\text{y}$ .

The greater depletion of manganese in highly organic soils is consistent with a model of uniform remobilization and export superimposed upon a varying substrate concentration. However such a model is limited by the assumption that surface (0-2 cm) layer concentrations represent incoming sediments, whereas the results of DeLaune et al. (1981) suggest that this may not be the case. Surface concentrations represent the interaction of continuing sedimentation, plant uptake, oxidative precipitation, and diffusive loss. Because the relative magnitude of each of these processes may vary with hydrology, such cross environmental comparisons are strictly invalid. Despite this approximation, manganese flux parameters ( $F$  and  $L$ ) calculated in this study are comparable with McCaffreys' (1972) values from a New England S. patens marsh ( $F = 0.46$ ;  $L = 0.1 \pm .04 \text{ g/m}^2/\text{y}$ ). In situ chamber measurements of Lord (1980) suggest that maximum instantaneous flux rates may approach  $4 \times 10^3 \mu\text{g/cm}^2/\text{y}$  or 250 times the integrated rate calculated from the sedimentary record. Using these results, Nixon (1980) has shown that

inorganic processes rather than plant uptake constitutes the primary mechanisms by which this export is achieved.

### C. PHYSICOCHEMISTRY

#### 1. Chlorinity and Specific Conductance

Interstitial water chlorinity (Fig. 11a) decreased from the Gulf Coast landward in Barataria Basin as anticipated from previous studies (Chabreck, 1972). This result is used here to illustrate the gradient in marine influence along the experimental transect. Physical and chemical gradients associated with marine intrusion have been identified as being of primary importance in defining soil (nutrient) characteristics in coastal Louisiana (Rainey, 1979).

The specific conductance of interstitial waters showed a primary dependence on chlorinity, showing no correlation with extractable metal concentrations. This result contradicts Brannon's (1973) use of conductance measurements to identify seasonal variations in extractable iron and manganese concentrations in salt marsh soils.

#### 2. Eh

##### (a) Variation with marsh type.

Eh varied significantly between marsh environments (Fig. 11b), decreasing predictably from the freshwater marsh (+ 13 mV) through intermediate (-81 mV) to similarly reduced values in the brackish (-162 mV) and salt marshes (-156 mV).

Although measurement of Eh is simple in principle, its interpretation in multicomponent aqueous soil systems involves complex

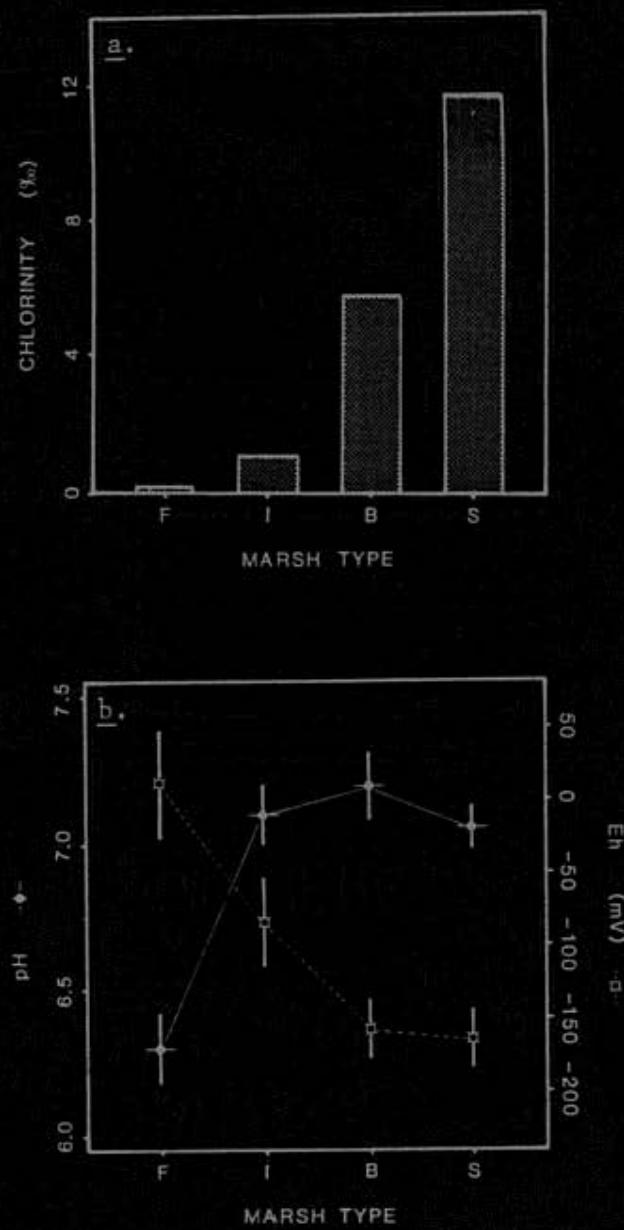


Fig. 11. Variation of physicochemical soil properties with marsh type.  
(a) Chlorinity.(b) Eh and pH. Error bars represent 95% confidence limits.

problems. Where several redox couples are simultaneously active at the electrode surface, interpretation in terms of the Nernst equation is impossible (Stumm and Morgan, 1970). Theoretical considerations aside, the interaction of physical and chemical gradients (soil porosity, tortuosity, Eh buffering capacity and microbial processes) in Barataria Basin are sufficiently complex so as to evade simple correlation with Eh.

It should be emphasized at this point that Eh is a measure of the relative activity of reduced and oxidized electroactive species. Except for the  $\text{Fe}^{2+}/\text{Fe}^{3+}$  couple and perhaps the  $\text{Mn}^{2+}/\text{Mn}^{4+}$  couple, most soil redox couples including  $\text{SO}_4^{2-}/\text{S}^{\circ}$  are not electroactive. But electroactive species may nevertheless influence Eh indirectly by reacting with electroactive species such as iron (1) thus altering the electrode potential (2).



$$E_{\text{Fe}^{2+}/\text{Fe}^{3+}} = E_{\text{Fe}^{2+}/\text{Fe}^{3+}}^{\theta} + \frac{RT}{nF} \ln \frac{\text{Fe}^{3+}}{\text{Fe}^{2+}} \quad (2)$$

Control of Eh by the accumulation of nonelectroactive species, such as sulfides, by nonequilibrium, microbially mediated processes is thus theoretically justified. Such an hypothesis is consistent with qualitative similarities in the sulfide (Fig. 13) and Eh (Fig. 11) patterns in Barataria Basin, and would provide an interesting problem for further investigation.

### 3. pH

#### (a) Variation with marsh type.

Whereas intermediate, brackish, and saline substrates were typically of neutral pH, porewaters in the freshwater marsh were slightly acidic, averaging 6.3 to a depth of 45 cm (Fig. 11b). This trend of increasing acidity from saline (mineral) soils to highly organic substrates of less saline environments has been identified in previous studies (Table 3) and may be related to Eh and/or the distribution of mineral sediments. In general, the colloidal complex of organic colloids, when saturated with hydrogen, develop a lower pH than acid mineral clays similarly charged (Brady, 1978), so that even at the same percentage base saturation, peat is somewhat more acid than mineral soil. However, mineral sediment distribution does not adequately explain the pH phenomenon in the Barataria Basin. The slightly acidic condition of the freshwater marsh soil is consistent with the relatively oxidized (or less reduced) condition of that system as compared to more saline environments.

#### (b) Variation with depth.

Relatively acidic conditions characterized the uppermost (0-12 cm) layers as compared to soils at depth (Fig. 12a), in agreement with the vertical Eh gradient: pH and Eh were significantly correlated in the intermediate ( $R = -0.66^{**}$ ), brackish ( $R = -0.35^{**}$ ), and salt marsh soils ( $R = -0.43^{**}$ ). Oxidative reactions of sulfur, iron, manganese, and organic matter (forming organic acids) at the soil-water interface probably contribute to this effect:

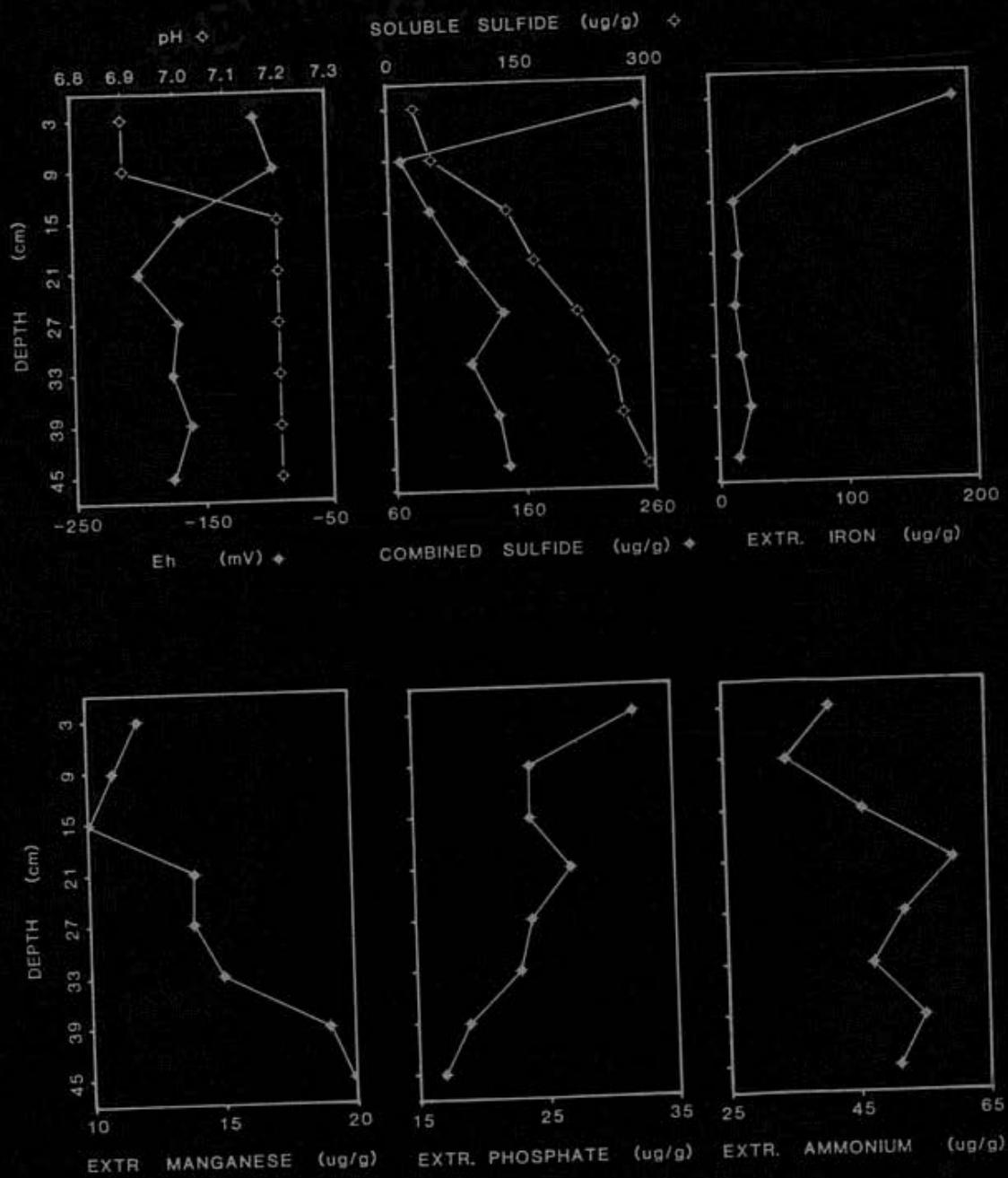
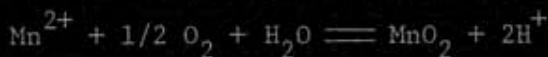
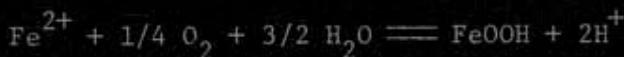
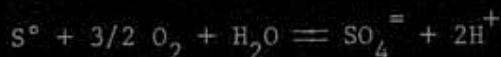


Fig. 12. Physicochemical, sulfide and extractable nutrient profiles.  
 (a) Eh and pH (b) Sulfides (c) Iron (d) Manganese (e) Phosphate  
 (f) Ammonium.



#### D. SULFIDES AND EXTRACTABLE NUTRIENTS

##### 1. Sulfides

###### (a) Variation with marsh type.

The bulk concentration of all forms of sulfide followed a significant seaward increase from freshwater through saline marsh environments (Fig. 13). Qualitative similarities between the distribution of sulfides and of saline waters (Fig. 11a) support the concept of a sulfate-limited sulfide distribution in Barataria Basin.

Combined sulfides showed a secondary dependence on the distribution of mineral sediments with markedly lower concentrations in highly organic peats as compared with high density mineral salt marsh soils (Table C2). Precipitation of sulfides as  $FeS$ ,  $Fe_3S_4$  and  $FeS_2$  by residual sedimentary Fe oxides is the major mechanism for the immobilization of sulfides in flooded salt marsh soils (Lord, 1980). The predominance of soluble sulfides over combined forms in typically organic substrates is thus explained by the paucity of iron and the large water content (Table C2). An apparent reversal of this trend in the freshwater marsh may thus be explained by the scarcity of sulfides relative to iron (Fig. 9).

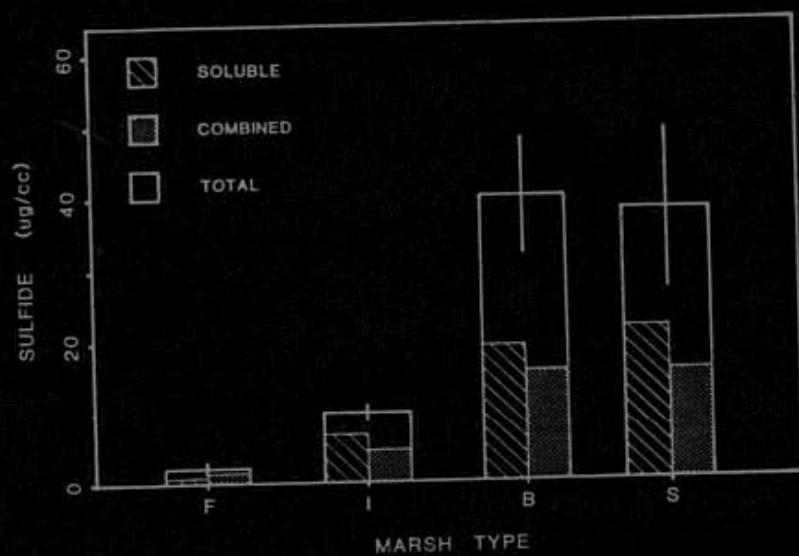


Fig. 13. Variation of sulfides with marsh type.  
Error bars represent 95% confidence  
limits.

## (b) Variation with depth.

Expressed on a dry weight basis, soluble sulfides showed a minimum in surface layers, increasing to a depth of 15-20 cm, with uniformly higher levels at greater depth (Fig. 12b, Table C1). This pattern, predicted by Eh ( $R = -0.38^{**}$ ), probably reflects the depletion of dissolved sulfide by reaction primarily with surface deposits of iron oxides:



Such reactions result in the quantitative release of ferrous iron ( $\text{Fe}^{2+}$ ) into solution (Fig. 12c).

Combined sulfide profiles were characterized by abruptly higher concentrations in the uppermost 6 cm soil layer than in the immediately subadjacent section (6-12 cm), with gradually increasing levels at greater depth (Fig. 18b). This pattern closely resembles non-pyrite sulfur distribution in salt marsh cores from Delaware (Lord, 1980): near-surface accumulations appear to represent both ferrous monosulfide ( $\text{FeS}$ ) and greigite ( $\text{Fe}_3\text{S}_4$ ), metastable intermediates in the formation of pyrite ( $\text{FeS}_2$ );  $\text{FeS}$  was not significantly represented in deposits at depth which must therefore constitute mainly  $\text{Fe}_3\text{S}_4$ . Metastable sulfides thus accumulate in a layer immediately subadjacent to the oxide-rich layer resulting in an apparent co-existence of high levels of oxides and sulfides, processes of oxidation and reduction proceeding simultaneously in adjacent microenvironments.

## 2. Extractable Iron and Manganese

### (a) Variation with marsh type.

With the notable exception of the freshwater marsh, the availability of extractable iron (Fig. 14) and manganese (Fig. 15) showed a general increase seaward along the marsh transect, but with considerable overlap between adjacent marsh types. Parallel trends in bulk density (Fig. 3) and total iron and manganese distributions (Fig. 8) support a dependence of extractable iron and manganese on both bulk exchange capacity and abundance of parent minerals.

Expressed on a dry-weight basis, the environmental trend of iron remained the same, with highest concentrations in the salt marsh environment (Table D2). By contrast, the manganese gradient was reversed indicating a general increase inland from the Gulf Coast. This result further illustrates an important difference in the behavior of iron and manganese in this marsh system, which was demonstrated initially by the distribution of total iron and manganese.

### (b) Variation with depth.

Pronounced near-surface maxima in extractable iron and manganese characterize soil profiles throughout the study area (Fig. 12c, d). Previous studies in salt marshes (Lord, 1980) indicate that such accumulations represent high levels of both soluble and exchangeable forms of these metals, closely associated with deposits of residual and antigenic oxides. Presumably the reaction of sulfides ( $\text{HS}^-$ ) with metal oxides results in the quantitative release of  $\text{Fe}^{2+}$  and  $\text{Mn}^{2+}$  into solution, with a consequent depletion of soluble sulfides in surface

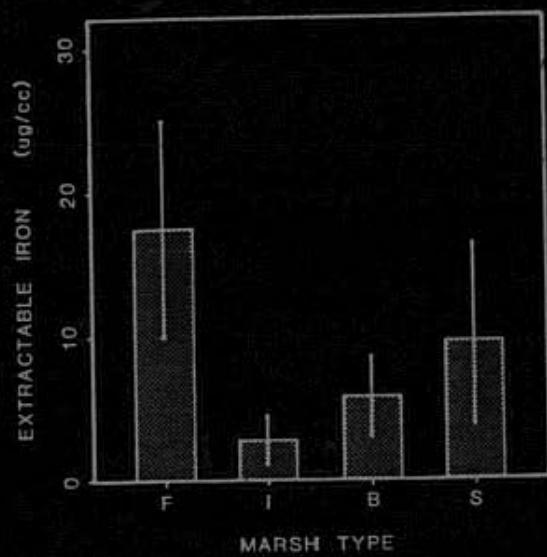


Fig. 14. Variation of extractable iron with marsh type.  
Error bars represent 95% confidence limits.

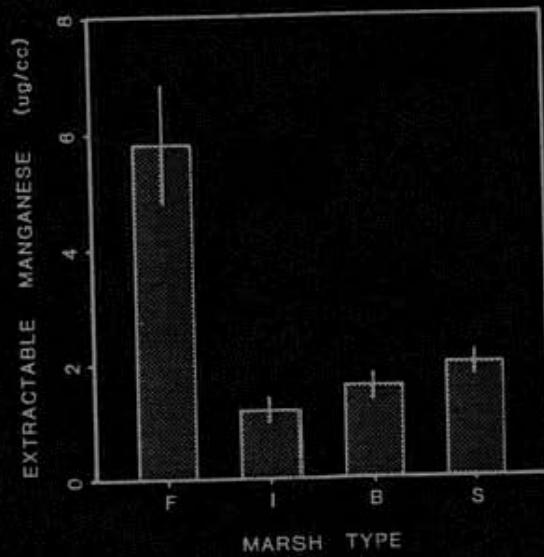


Fig. 15. Variation of extractable manganese with marsh type.  
Error bars represent 95% confidence limits.

layers (Fig. 12b). Dissolution is further enhanced by the relatively acidic conditions in surface layers. However, it is possible that much of the extractable fraction is actually amorphous oxide phase, dissolved by pH 4.5 acetate extraction. Higher concentrations of manganese at depth are thought to represent dissolution of anthiginic minerals such as  $MnCO_3$  and  $MnHPO_4$ , soluble in mildly acidic solutions.

### 3. Extractable Phosphate

#### (a) Variation with marsh type.

Similar bulk extractable phosphate concentrations were identified in all marsh substrates sampled along the transect (Fig. 16) except for particularly high levels in exceptionally mineral soils in the salt marsh (Table C2). Brannon (1973) and Rainey (1979) established the direct correlation between bulk density and the availability of phosphates expressed on a volume basis. Although this relationship is not readily apparent from the present study, its existence is qualitatively demonstrated where large bulk density gradients are involved, i.e., in the comparison of typically organic, with highly mineral salt marsh soils.

Expressed on a soil dry weight basis extractable phosphates increased predictably inland from the salt marsh through brackish and intermediate soils to attain a maximum in the freshwater marsh, in contrast to Chabreck's (1977) results derived from air-dried samples from Barataria Basin. This result was not expected considering the predominantly mineral association of phosphates: phosphate adsorption is thought to occur principally by association with iron (Patrick and

Mahapatra, 1968). Nixon (1980) has suggested that such gradients, also manifest in total phosphorus distribution, may result from a more rapid remobilization of phosphates from intertidal as compared to supratidal, and subtidal marine deposits.

(b) Variation with depth.

Abruptly higher concentrations of extractable phosphate were consistently identified in surface layers, coincident with high levels of both iron and manganese (Fig. 12). Iron and phosphates were positively correlated in freshwater ( $R = 0.40^{**}$ ), brackish ( $R = 0.27^{**}$ ) and salt marsh soils ( $R = 0.46^{**}$ ). The results of Lord (1980) suggest that porewaters do not contribute to the higher surface concentrations, which are thus a direct reflection of iron oxide-phosphate adsorption. Authigenic and residual iron oxide precipitates appear to constitute an efficient trap for upward migrating phosphates, thus limiting diffusional loss to supernatant floodwaters (Lord, 1980).

4. Extractable Ammonium

(a) Variation with marsh type.

With the exception of the intermediate marsh, ammonium availability increased seaward in the basin reaching a maximum in the salt marsh environment (Fig. 17). Together with soil profiles (Table C1), these results indicate a primary dependence on bulk density. This trend is reversed when expressed on a dry weight basis (Table D1, D2), consistent with the greater exchange capacity, and the preferential

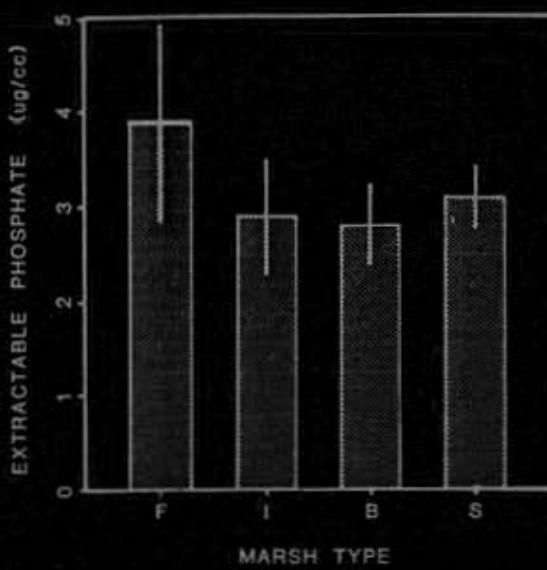


Fig. 16. Variation of extractable phosphate with marsh type.  
Error bars represent 95% confidence limits,

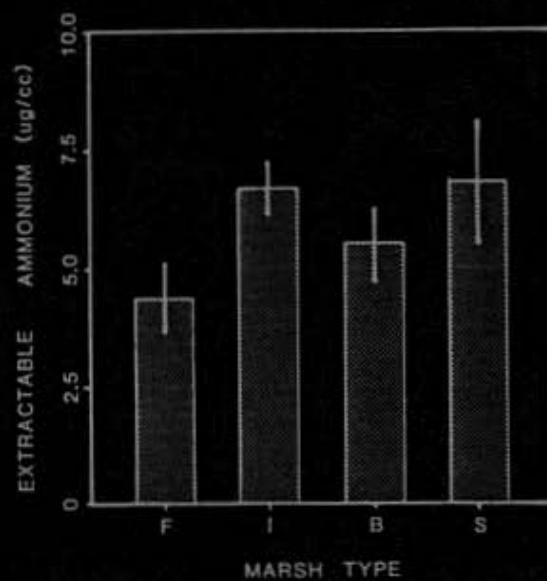


Fig. 17. Variation of extractable ammonium with marsh type.  
Error bars represent 95% confidence limits.

adsorption of ammonium by organic matter as compared to inorganic sediments. The contribution of microbial processes to the environmental trend remains to be evaluated.

(b) Variation with depth.

Ammonium showed a slight increase with depth in the upper 20 cm of soil profiles with a local minimum (inconsistently) apparent in the 6-12 cm Section. This effect, predicted by Eh, has been attributed to the supply of oxygen from the soil surface and through rhizosphere oxidation (Casselman, 1979).

### 5. General Considerations

Results presented here suggest that mineral sediments are of primary importance in defining bulk extractable nutrient concentrations. Because of the limited data available, this analysis has been largely qualitative. However the causal relationships identified in this study are in accord with Rainey's (1979) conclusions with respect to phosphates and extractable metal ions in Louisiana marshes.

When expressed in units of dry weight, extractable nutrient concentrations reveal a complex situation which does not conform to any broad generalizations. A considerable number of largely unknown factors potentially contribute to the observed environmental variations. These include (i) differential adsorption of specific cations by the sediment, (ii) selectivity of specific soil types for specific ions, (iii) degree of fixation by the sediment, (iv) selective utilization of nutrients by vegetation, (v) base saturation capacity and pH, (vi) Eh, (vii) nature and abundance of sedimentary minerals,

(viii) hydrologic gradients, (ix) varying microbial activity especially involving nitrogen transformations, and (x) anthropogenic activity. Thus explanations offered in the preceding discussion are by necessity, largely speculative.

The unusually high levels of iron and manganese, and to a lesser extent phosphate, in the freshwater marsh may be partially explained by the slightly acidic, less reduced condition of that soil, and the paucity of sulfides which otherwise serve to immobilize metals as insoluble precipitates. Sims and Patrick (1978) have recorded substantial increases (30-50%) in the water-soluble concentrations of iron and manganese as a result of a change in pH from 7.5 to 6.0, showing that large changes in the availability of these and associated elements may be induced by relatively small changes in acidity. In addition to pH, nutrient enrichment of this freshwater hydrologic unit by agricultural runoff has been recorded by Kemp (1975).

## SUMMARY AND CONCLUSIONS

### A. MARSH ACCRETION

1. Whereas the supply of mineral sediments was of primary importance in defining local variations in marsh accretion and subsidence, this relationship did not hold for "cross environmental" comparisons. Similar vertical accretion rates were observed in all comparable marsh environments, independent of the seaward gradient in mineral sediment supply and regional subsidence; values ranged from 3.1 mm/y in inland areas to 16.9 mm/y in natural levee deposits. Mechanisms for the maintenance of marsh elevation appear to vary between marsh types, related to the supply of mineral sediments and the vegetation type.

2. In situ variability severely limits extrapolation of vertical accretion rates computed from low intensity studies, to marshes as a whole; vertical accretion rates should be accompanied by records of spatial coordinates and an index of in situ variance.

3. Successful application of the <sup>137</sup>Cs dating method in this marsh system confirm the suitability of the technique for use in highly reduced, typically organic substrates.

### B. BULK DENSITY, AND TOTAL NUTRIENT DISTRIBUTION AND FLUX

1. The abundance of mineral sediments in Barataria Basin marshes varied inversely as the distance from the Gulf Coast. Inorganic sediments were highest in salt marsh soils composing a progressively smaller fraction of soil solids in less saline marsh environments.

2. Similar bulk organic matter concentrations were identified in all environments such that bulk density reflects directly the abundance of mineral sediments. A predictable curvilinear relationship was established between bulk density and dry weight organic carbon content.

3. Bulk (Kjeldahl) nitrogen concentrations, like carbon did not change significantly between marsh types. Nitrogen and carbon were linearly related over all environments, yielding a mean C:N ratio of 18.2. Accumulation of these elements thus proceeds at a roughly uniform rate in all marsh environments studied.

4. Bulk concentrations of phosphorus, iron and manganese decreased inland from the Gulf Coast, as predicted by the distribution of mineral sediments. The total nutrient potential of marsh soils and the accumulation rates in respect of these minerals thus decrease from the salt marsh through less saline environments, to the freshwater marsh.

5. Phosphorus, carbon, and nitrogen appear to constitute a progressively greater fraction of soil solids inland from the Gulf Coast. The reverse was true for iron, and for manganese with the exception of the freshwater marsh. Freshwater marsh soils were exceptionally rich in manganese and slightly enriched in phosphate. The sedimentary concentration of iron was essentially defined by the abundance of mineral sediments. This relationship did not hold for manganese which shows only a weak correlation with iron. Phosphorus varied independently of mineral sediments and a weak association with

organic matter was implied from statistical correlations. Interpretation of manganese and phosphorus associations was complicated by effects due to remobilization processes.

6. Whereas there is a net accumulation of phosphorus and manganese soil profiles showed clearly that these elements are subject to significant post-depositional remobilization and loss from marsh soils throughout the basin. Phosphorus depletion was relatively uniform in all marshes: calculations based on soil profiles indicate a loss of approximately 30% (22-55%) of sedimentary phosphorus through chemical and biochemical pathways, at a rate of  $0.16\text{-}2.32 \text{ g/m}^2/\text{y}$ . Manganese loss ranged from 18% to 69% of the influx, loss rates ranging from  $0.16\text{-}2.32 \text{ g/m}^2/\text{y}$ . Greatest manganese depletion was apparent in low density freshwater and intermediate marsh substrates. Phosphorus and manganese remobilization was more pronounced in streamside than in inland deposits.

#### C. PHYSICOCHEMISTRY, SULFIDES AND EXTRACTABLE NUTRIENTS

1. Soils were progressively reduced along the transect from freshwater to salt marshes. Theoretical considerations do not allow precise explanation of this result.

2. With the exception of the freshwater marsh, soils were typically of neutral acidity. pH in the freshwater marsh was slightly acidic averaging 6.3. This result may be explained by the redox condition of these soils.

3. Sulfides were most abundant in saline marshes, decreasing with salinity inland. This result is consistent with a

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3. Sulfides were most abundant in saline marshes, decreasing with salinity inland. This result is consistent with a

sulfate-limited model for the distribution of sulfides in the basin. Combined sulfide accumulation was defined secondarily by the abundance of residual and antihigenic iron oxides and as such was limited by the mineral sediment supply. Soluble sulfides varied inversely as Eh in soil profiles increasing with depth below the soil surface.

4. The bulk availability of extractable nutrients (including iron, manganese, phosphates, and ammonium) increased seaward with bulk density from freshwater through saline environments. The importance of bulk density and mineral sediment supply in controlling nutrient availability was reiterated in all nutrients studied. This relationship was clearly defined in the case of iron and manganese but inconsistent for phosphates and ammonium. Freshwater marsh soils yielded significantly higher levels of iron, manganese, and phosphates, which may be related to physicochemical (Eh, pH) peculiarities in that system, and to authropogenic contamination.

5. Dry weight extractable nutrient distributions did not consistently conform to theoretical predictions based on cation exchange capacity alone. Iron showed a strong association with mineral sediments whereas the reverse was generally true for manganese phosphate and ammonium on the environmental transect. The causal relationships of these phenomena are potentially complex, and poorly understood in this system.

6. Significantly higher dry weight concentrations of extractable iron, manganese and phosphates were consistently identified in near-surface layers than at depth in soil profiles, a pattern which

is apparently explained by the distribution of reductant soluble (anthigenic and residual) metal oxides.

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Appendix A.  $^{137}\text{Cs}$  data tables and figures.

Core Designation:

Marsh type	1 (A2)	40	R1	Repetition
	Site	Distance from adjacent water body (m).		
1 = Freshwater				
2 = Intermediate				
3 = Brackish				
4 = Salt				

Table A1.  $^{137}\text{Cs}$  activity profiles. (a) Freshwater marsh.

DEPTH (cm)	ACTIVITY (pCi/section)						
	1(A1) 10	1(A2) 40	1(B1) 81	1(B-1) 30	1(B+) 30	1(B2) 60	1(C1) 80
1	13.5	139.3	40.5	19.6	94.3	45.8	72.5
3	73.3	243.3	101.8	45.0	75.5	17.1	81.9
5	105.7	281.5	103.7	66.9	115.2	0.0	101.4
7	92.5	267.0	86.5	84.3	187.4	86.2	185.4
9	91.1	137.9	235.8	124.8	260.9	151.3	228.6
11	117.6	66.6	329.0	208.5	424.4	218.8	491.0
13	222.6	43.0	187.0	176.9	309.3	180.9	429.6
15	263.0	18.7	377.5	179.4	286.7	183.4	321.9
17	281.0	0.8	419.6	212.9	218.2	141.1	201.7
19	229.9	0.0	220.7	226.9	106.8	133.3	279.5
21	101.8	0.8	48.3	84.3	41.6	51.4	14.0
23	39.7	0.0	46.8	56.3	32.9	40.8	33.5
25	0.0	49.4	58.9	54.2	41.4	41.4	25.1
27	0.0	24.0	0.0	19.1	31.6	4.2	19.5
29	7.6	0.0	0.0	5.7	12.4	14.3	3.7
31	0.0	0.0	0.0	0.0	0.0	0.0	0.0
33	0.0	0.0	0.0	0.0	0.0	0.0	0.0
35	0.0	0.0	0.0	0.0	0.0	0.0	0.0
37	0.0	0.0	0.0	0.0	0.0	0.0	0.0
39	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Table A1 (continued). (b) Intermediate Marsh.

DEPTH (cm)	ACTIVITY (pCi/section)									
	2 (A55) 2 (A1) 20	2 (A2) 20C	2 (H55) B1	2 (H55) B2	2 (H1) 20	2 (H2) 20	2 (C1) 20	2 (H3) 20	2 (H4) 20	2 (H5) 20
1	11.8	85.8	18.5	57.5	49.9	10.7	40.5	125.2	25.2	48.9
44.3	79.5	34.0	51.8	72.5	92.1	117.0	243.2	41.1	95.6	68.1
5	90.6	206.5	29.6	95.0	68.2	69.1	137.5	320.8	76.5	226.4
7	99.8	236.8	41.3	91.2	94.3	101.2	151.3	321.9	117.7	492.0
9	96.0	195.1	66.6	50.2	92.8	125.5	205.5	234.1	449.5	163.6
11	86.8	239.7	92.2	84.2	93.6	119.6	189.6	162.8	382.9	146.6
13	89.9	173.4	105.3	104.0	102.3	102.3	546.6	163.8	257.8	840.2
15	101.4	160.9	195.4	177.0	188.7	621.0	68.1	96.2	167.7	245.3
17	209.9	151.0	289.5	167.2	220.7	597.4	812.7	62.5	132.6	66.1
19	209.9	160.4	353.6	283.1	319.6	115.5	182.3	52.9	64.2	41.1
21	253.6	68.6	410.4	433.6	547.5	24.6	21.7	30.1	21.6	14.0
23	455.5	87.2	358.8	182.9	304.9	147.4	283.3	22.3	38.2	22.1
25	325.8	27.3	190.1	104.9	210.9	20.2	0.0	31.3	10.2	7.2
27	287.0	18.0	113.8	191.1	161.0	18.5	3.1	18.2	6.2	5.1
29	255.8	13.5	62.0	95.1	51.4	20.9	0.0	7.8	7.8	17.9
31	96.7	19.0	41.0	58.5	58.5	10.6	0.0	2.4	7.8	22.1
33	59.2	11.8	40.0	72.6	72.6	11.9	0.0	0.0	0.0	0.0
35	0.0	1.2	1.8	27.6	27.6	1.2	0.0	0.0	0.0	0.0
37	29.6	0.0	10.1	10.2	10.2	7.8	0.0	0.0	0.0	0.0
39	16.2	7.0	21.2	21.8	21.8	0.3	0.0	0.0	0.0	0.0
41	24.0	2.1	24.0	2.1	24.0	7.1	0.0	0.0	0.0	0.0
43	2.1	3.1	2.1	3.1	2.1	0.0	0.0	0.0	0.0	0.0
45	3.1	3.1	3.1	3.1	3.1	0.0	0.0	0.0	0.0	0.0

Table Al (continued), (c) Brackish Marsh.

DEPTH (cm)	ACTIVITY (pCi/section)									
	3(A1)20	3(B1)30	3(B2)50	3(C1)150	3(BB)200	3(C1)20	3(C2)60	3(m)20	3(55)P1	3(55)P2
1	60.2	39.6	51.4	49.1	71.8	64.2	16.4	92.7	83.3	69.4
3	65.2	94.2	57.0	67.8	39.7	67.9	63.0	54.9	65.5	50.7
5	111.3	81.1	145.0	123.7	63.9	93.6	86.2	82.0	64.8	60.0
7	161.3	113.5	146.6	141.9	89.7	99.2	109.3	114.7	95.9	95.9
9	224.0	216.3	82.6	131.8	81.1	159.1	122.3	92.0	66.3	72.5
11	231.6	330.2	35.2	88.9	32.7	214.3	201.6	117.2	113.8	95.8
13	138.6	223.0	19.9	17.1	20.2	264.2	219.0	110.4	104.5	109.9
15	101.7	89.7	11.2	7.0	27.1	87.9	153.5	82.3	120.1	137.2
17	24.8	62.2	5.6	1.2	33.0	55.2	51.4	45.5	145.0	134.1
19	24.8	41.0	17.6	0.0	0.0	24.9	18.5	46.1	180.9	166.1
21	17.0	66.7	0.0	5.3	18.7	72.3	0.9	41.4	135.7	155.2
23	9.0	47.2	0.0	0.0	15.2	23.0	40.1	48.9	148.2	169.2
25	11.6	65.9	0.0	0.0	8.4	19.1	13.4	131.8	131.8	167.7
27	32.6	37.2	0.0	0.0	11.7	3.7	32.7	189.5	74.1	237.9
29	8.1	40.2	0.0	0.0	23.7	16.2	53.9	305.7	60.0	122.4
31	11.6	39.0	0.0	0.0	1.4	0.0	35.8	293.2	74.8	180.3
33	36.8	0.0	0.0	0.0	19.5	0.0	28.0	193.4	72.5	39.0
35	34.1	0.0	0.0	0.0	0.3	0.0	8.7	128.7	43.3	39.0
37	31.2	0.0	0.0	0.0	0.0	0.0	0.0	129.4	48.3	48.3
39	0.0	0.0	0.0	0.0	0.0	0.0	0.0	141.9	31.3	18.7
41	0.0	0.0	0.0	0.0	0.0	0.0	0.0	21.6	25.8	20.2
43	0.0	0.0	0.0	0.0	0.0	0.0	0.0	81.9	10.9	29.6
45	0.0	0.0	0.0	0.0	0.0	0.0	0.0	71.6	19.9	48.3

Table A1 (continued). (d) Salt Marsh.

DEPTH (cm)	ACTIVITY (pCi/section)				
	#(A1)20	#(A2)40	#(A1)20	#(A1)20	#PAL20
1	57.0	23.6	28.9	65.2	33.5
3	84.3	39.3	45.8	72.3	68.6
5	109.7	29.1	67.7	89.5	58.9
7	107.6	73.1	61.7	96.0	158.3
9	188.2	157.8	83.6	156.3	207.8
11	163.4	191.6	117.6	187.2	170.0
13	96.7	63.3	193.8	215.9	151.3
15	52.4	41.5	154.1	238.6	96.7
17	41.8	37.5	163.0	227.7	38.2
19	13.7	28.6	20.2	97.5	13.4
21	3.5	3.5	5.3	56.9	20.1
23	26.0	26.3	11.8	26.1	17.1
25	29.4	6.7	26.0	61.0	20.2
27	8.1	0.0	24.2	21.2	8.4
29	21.5	7.8	1.0	0.0	19.6
31	3.3	0.0	9.0	1.0	21.8
			8.4	14.0	15.8
					3.1

Table A2. Mean and mean peak  $^{137}\text{Cs}$  activities by marsh type.

Marsh Type	Activity			
	pCi/Section		pCi/g	
Mean	Peak	Mean	Peak	
Freshwater	122.1	350.6	4.7	13.7
Intermediate	121.1	429.3	3.5	11.0
Brackish	71.7	206.4	1.3	3.8
Salt	35.9	193.0	0.9	2.6

Figure A1. Sedimentary Cs-137 activity profiles: graphical representation. Activity expressed in units of pCi per 2 cm core section.

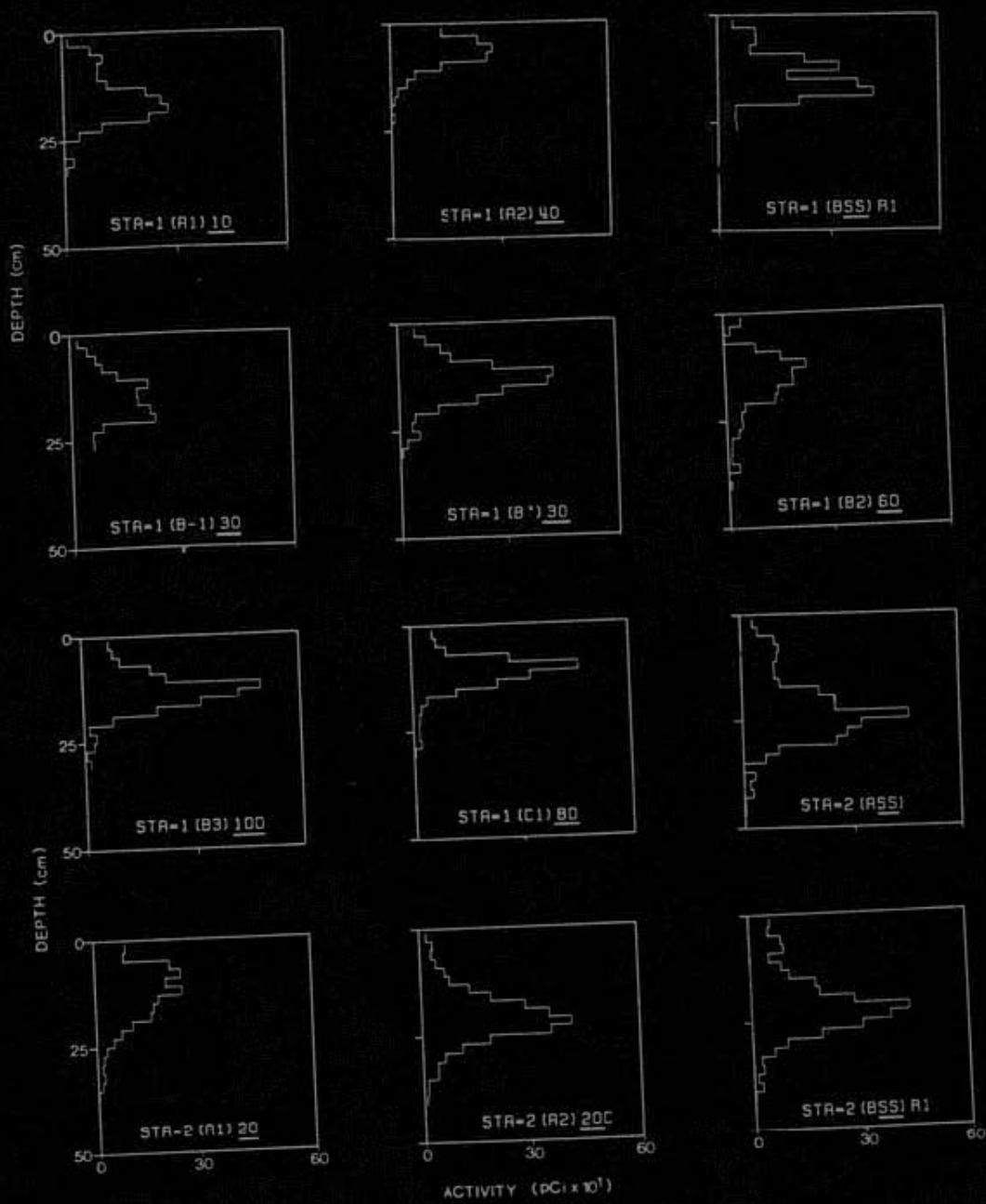


Figure A1 (continued)

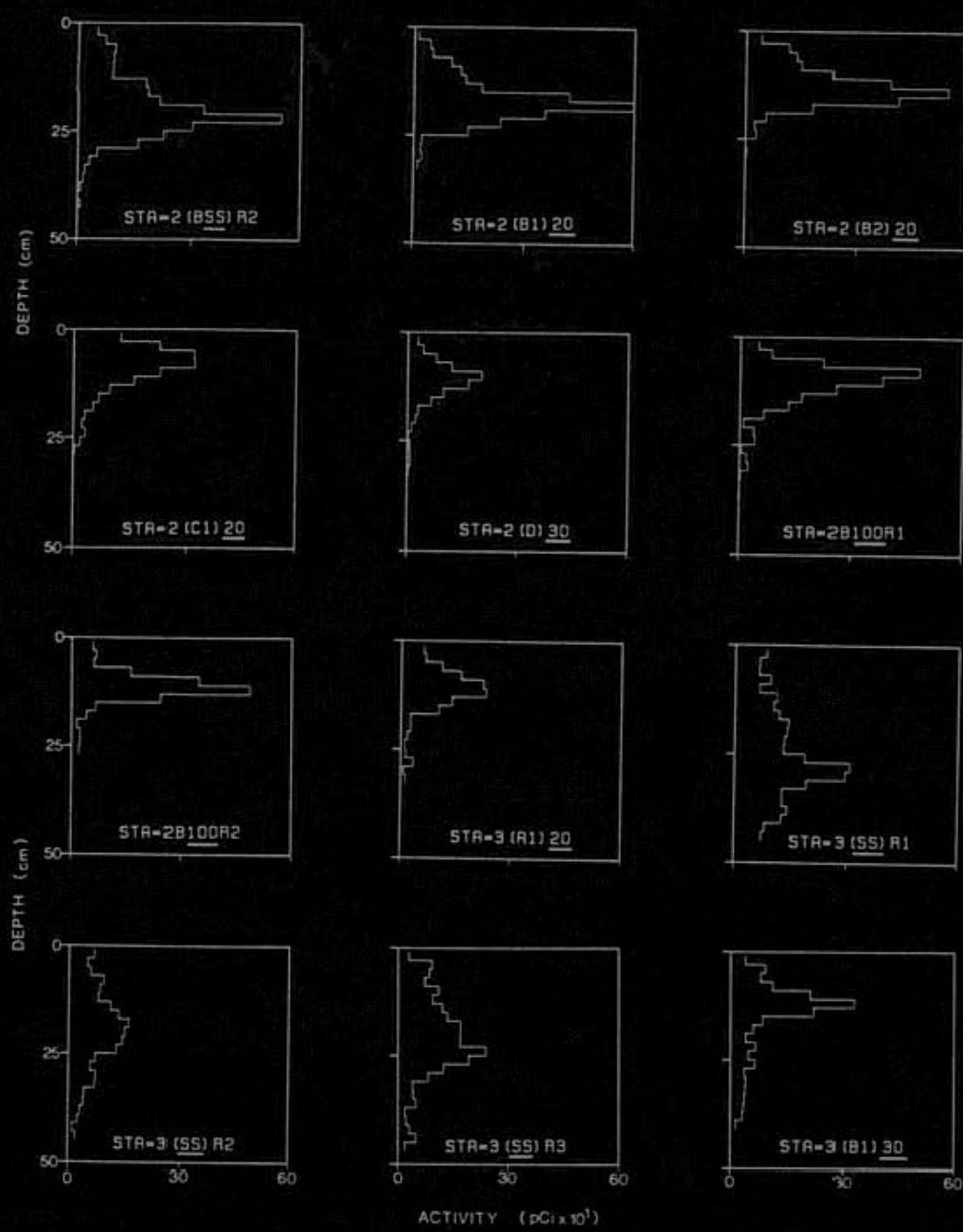
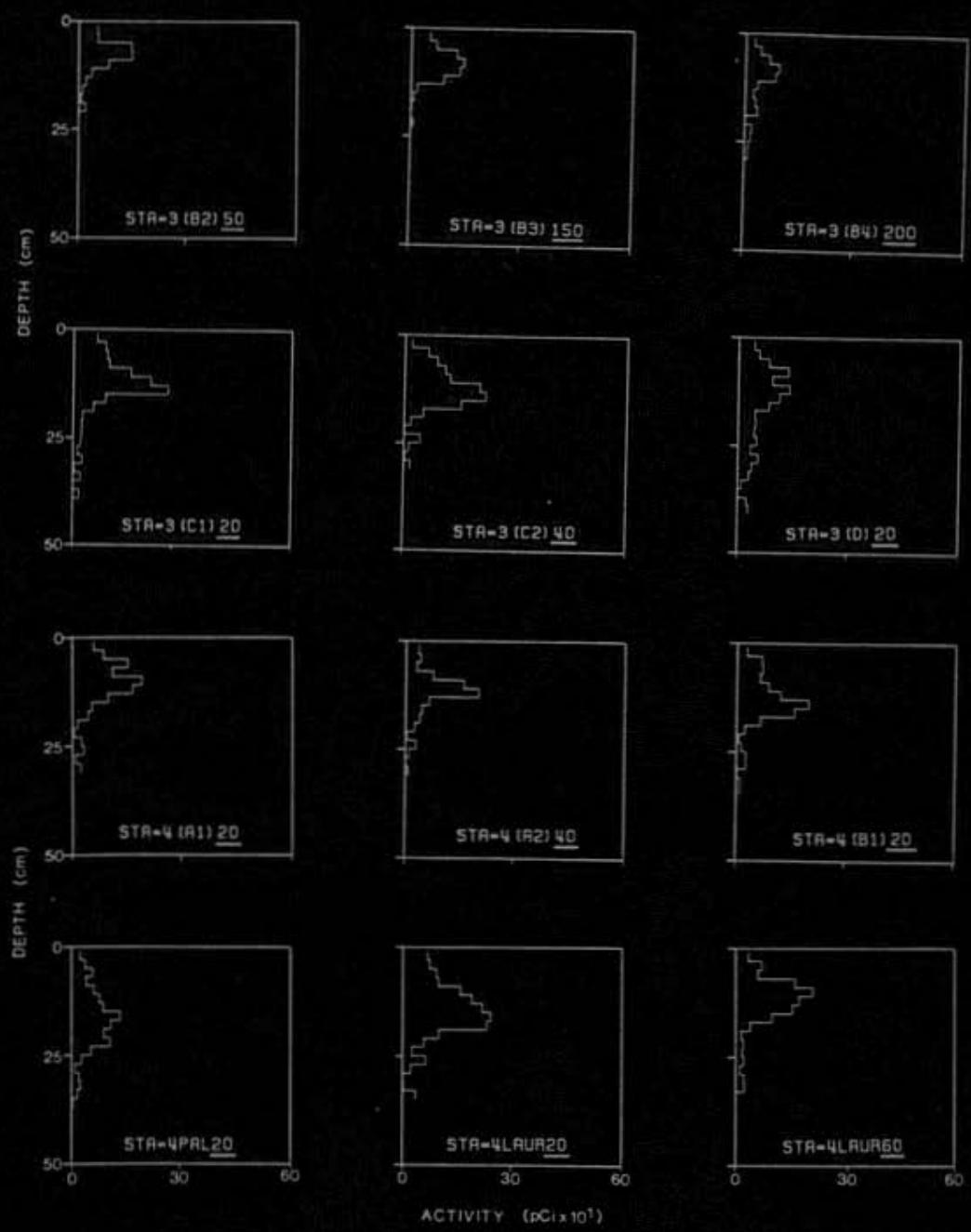


Figure A1 (continued)



Appendix B. Data tables for bulk density, organic carbon, and (total) nitrogen, phosphorus, iron and manganese.

Variable	Symbol	Unit
Bulk density	D	g/cc
Total organic carbon	C	%
Total (Kjeldahl) nitrogen	N	%
Total phosphorus	P	ug/g
Total iron	Fe	ug/g
Total manganese	Mn	ug/g

Table B1. Profiles of bulk density, organic carbon, and (total) nitrogen, phosphorus, iron and manganese.

Marsh Environment	Depth (cm)	D (g/cc)	C (%)	N (%)	P (ug/g)	Fe (ug/g)	Mn (ug/g)
FRESHWATER STREAMSIDE	1	0.07	22.5	1.42	1100	15000	353
	5	0.10	21.3	1.31	1196	12500	148
	9	0.08	29.7	1.99	1221	7353	94
	13	0.11	20.1	1.46	1083	11111	100
	17	0.07	24.4	2.22	1063	8594	114
	21	0.06	35.8	2.16	1043	9286	118
	25	0.09	29.0	1.45	833	14103	115
	29	0.18	16.6	0.94	694	20833	125
	33	0.16	16.2	0.94	391	29219	129
	37	0.14	17.7	1.10	641	24286	133
FRESHWATER INLAND	1	0.05	33.0	2.00	1386	7678	169
	5	0.07	33.6	1.99	1116	5632	82
	9	0.07	33.7	2.18	1079	5811	75
	13	0.05	17.0	2.14	1075	5110	93
	17	0.05	37.4	2.22	990	5520	88
	21	0.08	16.3	2.33	806	8049	104
	25	0.06	32.2	1.76	838	12104	121
	29	0.10	25.0	1.42	674	13653	126
	33	0.19	11.0	0.84	714	18137	127
	37	0.37	16.2	0.93	744	18623	148
INTERMEDIATE STREAMSIDE	1	0.16	18.6	1.19	883	21760	113
	5	0.18	16.2	1.16	769	19231	92
	9	0.25	14.6	1.66	538	18196	72
	13	0.16	15.1	1.27	669	19711	58
	17	0.29	15.0	1.05	387	17453	59
	21	0.21	14.5	0.67	510	20142	60
	25	0.18	13.9	0.96	588	23079	73
	29	0.13	22.7	1.36	720	24152	59
	33	0.11	27.1	1.43	721	11441	56
	37	0.09	27.4	1.52	702	11513	52
INTERMEDIATE INLAND	1	0.06	28.5	1.47	1012	12125	89
	5	0.10	24.1	1.61	764	14421	74
	9	0.08	27.9	1.70	726	12567	63
	13	0.09	28.6	1.79	684	12675	61
	17	0.07	29.5	2.15	725	8430	59
	21	0.06	35.6	1.09	694	7616	44
	25	0.07	33.6	2.18	744	6186	45
	29	0.08	37.5	2.25	784	7177	55
	33	0.08	29.6	2.22	755	8229	54
	37	0.04	31.3	2.32	716	10434	54
BRACKISH STREAMSIDE	1	0.25	14.1	0.73	1184	18275	150
	5	0.26	17.6	0.74	759	18989	85
	9	0.25	12.1	0.69	570	20311	104
	13	0.29	8.8	0.64	604	21615	90
	17	0.13	10.0	0.63	611	20779	83
	21	0.24	14.7	0.65	548	20192	90
	25	0.28	11.0	0.52	540	18159	91
	29	0.29	11.9	0.64	520	24230	103
	33	0.22	11.4	0.73	529	22489	100
	37	0.26	13.4	0.67	369	23113	104
BRACKISH INLAND	1	0.12	17.5	1.15	895	15789	100
	5	0.16	16.3	1.07	722	16337	95
	9	0.17	18.4	0.47	650	15716	89
	13	0.13	24.5	1.26	645	11377	68
	17	0.11	26.6	1.42	718	9654	61
	21	0.11	25.9	1.30	619	9637	62
	25	0.12	26.9	1.36	576	10130	66
	29	0.12	22.7	1.24	647	11294	75
	33	0.12	25.0	1.43	594	8595	69
	37	0.11	32.9	1.01	567	7816	71
SALT INLAND	1	0.15	11.3	0.86	833	18666	97
	5	0.22	11.1	0.77	733	18679	82
	9	0.26	13.2	0.72	695	17303	77
	13	0.27	11.4	0.57	642	16629	83
	17	0.23	13.3	0.70	665	15442	73
	21	0.31	14.3	0.61	671	16488	74
	25	0.30	10.7	0.64	657	17169	87
SALT	29	0.39	11.5	0.77	631	15238	84
	33	0.34	8.1	0.51	640	14891	83
	37	0.45	5.4	0.40	511	14410	80

Table B2. Environmental means and statistical analysis of bulk density, organic carbon, and (total) nitrogen, phosphorus, iron and manganese.

Variable	Environment	Mean	N	Standard Error	Grouping <sup>a</sup>	Source	Analysis of Variance		
							DF	SS	MS
D (g/cc)	F	.11	41	.02	A	Model	6	1.39	14.65**
	I	.08	40	.00	A	Error	194	3.06	.07
	B	.14	40	.01	A	Total	200	4.44	
	S	.29	50	.03	B				
E (z)	F	29.6	41	1.6	A	Model	6	10825.6	1804.34**
	I	23.1	40	2.1	A	Error	190	9843.5	.52
	B	23.7	40	1.0	B	Total	196	20668.0	
	S	11.4	46	.8	C				
N (z)	F	1.80	40	.09	A	Model	6	45.66	7.61**
	I	2.00	37	.08	B	Error	179	30.89	.17
	B	1.31	40	.05	C	Total	185	76.56	
	S	.69	39	.05	D				
P (ug/g)	F	914	40	.41	A	Model	6	2644850	441641.60**
	I	763	40	.17	B	Error	163	10103188	.552083
	B	664	40	.20	B	Total	189	11753039	
	S	668	40	.40	B				
Fe (ug/g)	F	9956	40	1024	A	Model	6	231649900.3	384146650.17.54**
	I	10078	39	811	A	Error	182	400709994.8	24017032
	B	11829	40	986	A	Total	188	632409935.2	
	S	10856	40	436	B				
Mn (ug/g)	F	114	40	6	A	Model	6	9574.7	15958.15.93**
	I	60	39	4	B	Error	181	18134.3	1001
	B	77	40	3	C	Total	187		
	S	83	40	4	C				

\*Highly significant at  $P < .01$ .

<sup>a</sup> Means with the same letter are not significantly different.

Table B3. Correlation of bulk density, organic carbon, and (total) nitrogen, phosphorus, iron and manganese.

	C	N	P	Fe	Mn
D	-.64 **	-.62 **	-.25	.45 **	.05
C		.87 **	.36 **	-.81 **	-.19 **
N			.34 **	-.74 **	-.20 **
P				-.39 **	-.20 **
Fe					.33 **

\*\* Highly significant at  $P < .01$ .

Appendix C. Data tables for physicochemical soil properties, sulfides and extractable nutrients expressed on a volumetric basis.

Legend		
Variable	Symbol	Unit
Depth	Dept	cm
Eh	Eh	mv
pH	pH	
Specific conductance	SPC	μmhos
Chlorinity	CL	°/oo
Percent water content	H <sub>2</sub> O	% (g/g)
Bulk density	D	g/cc
Soluble sulfides	SS	μg/cc
Combined sulfides	CS	μg/cc
Total sulfides	TS	μg/cc
Extractable Fe	Fe	μg/cc
Extractable Mn	Mn	μg/cc
Extractable PO <sub>4</sub>	PO <sub>4</sub>	μg/cc
Extractable NH <sub>4</sub>	NH <sub>4</sub>	μg/cc

Table C1. Profiles of physicochemical soil properties, sulfides and extractable nutrients expressed on a volumetric basis.

MARSH	DEPTH (cm)	PH	SPC (umhos)	Cl <sub>x</sub> (%) <sub>so</sub>	H <sub>2</sub> O (%) <sub>so</sub>	D (g/cc)	SS (g/cc)	CS (g/cc)	PE (g/cc)	NR (ug/cc)	POL (ug/cc)	NH <sub>4</sub> (ug/cc)
INTERMEDIATE	3	7.6	6.2	0.71	0.14	89.7	0.07	0.2	1.1	1.2	4.8	3.9
	9	1.18	6.3	0.58	0.11	89.6	0.08	0.5	1.6	2.6	3.5	3.7
	15	-2.3	6.3	0.51	0.12	90.5	0.07	1.1	2.1	2.6	14.5	5.2
	21	-3.2	6.2	0.55	0.13	90.5	0.08	0.8	1.8	2.1	9.2	4.7
	27	-1.1	6.3	0.56	0.14	83.5	0.17	0.7	1.3	1.7	10.9	3.0
	33	-1.1	6.3	0.67	0.15	82.1	0.17	1.3	2.7	3.2	25.3	6.2
	39	-6	6.3	0.76	0.16	82.1	0.11	0.8	1.3	1.8	10.9	3.1
	45	-31	6.2	0.80	0.19	85.5	0.10	0.7	1.6	2.3	10.1	1.0
	3	3.9	6.7	2.78	0.90	85.9	0.07	1.5	4.4	5.7	10.5	4.1
	9	-5.8	6.9	3.08	0.94	84.9	0.10	3.9	4.4	7.4	4.3	3.5
SALT	15	-11.2	7.0	3.07	1.03	86.0	0.10	6.3	4.0	9.9	1.6	8.5
	21	-9.8	7.0	3.36	1.08	87.1	0.07	7.2	4.3	10.0	1.1	0.8
	27	-9.8	7.1	3.61	1.13	87.8	0.06	7.2	3.6	9.7	0.9	1.6
	33	-11.2	7.2	3.76	1.26	87.7	0.07	7.1	4.0	10.6	1.1	1.2
	39	-10.6	7.1	4.20	1.24	86.7	0.08	8.4	4.4	12.0	1.2	1.9
	45	-10.0	7.1	4.67	1.48	86.6	0.08	9.1	4.3	12.5	1.2	1.9
	3	-107	6.9	18.25	7.18	79.5	0.13	1.9	32.7	46.5	24.4	5.3
	9	-92	6.9	18.81	7.64	79.8	0.14	6.8	10.0	20.9	9.4	3.4
	15	-166	7.2	17.93	7.05	79.8	0.11	15.7	22.6	22.6	1.1	3.9
	21	-199	7.2	17.10	6.48	82.9	0.10	17.9	12.3	3.6	2.0	5.3
SALT ORGANIC	27	-169	7.2	17.80	6.66	84.0	0.11	24.3	16.6	41.8	1.4	2.9
	33	-173	7.2	18.80	7.24	80.8	0.11	29.4	13.8	47.2	2.7	5.8
	39	-161	7.2	18.51	6.66	85.7	0.10	27.4	14.2	46.1	2.2	5.4
	45	-176	7.2	19.96	7.19	85.8	0.10	30.7	15.0	48.2	1.5	5.3
	3	-115	6.6	10.24	10.68	76.2	0.18	3.6	21.6	25.4	42.3	5.3
	9	-72	6.7	29.88	11.65	74.5	0.20	5.1	9.7	18.8	18.1	2.1
	15	-145	6.8	31.30	11.38	73.1	0.23	15.3	12.9	25.1	7.3	4.0
	21	-180	6.8	29.00	11.27	79.8	0.16	12.2	12.1	3.4	1.4	2.5
	27	-180	6.9	30.76	11.58	77.8	0.13	14.0	9.6	23.7	8.8	2.7
	33	-191	7.0	31.18	12.00	76.8	0.18	18.8	13.9	32.7	5.2	4.1
	39	-199	7.1	28.78	10.23	79.1	0.22	37.9	18.8	70.9	7.3	7.3
	45	-196	7.1	28.36	10.46	81.1	0.26	64.2	26.1	90.3	4.8	3.1
	3	-123	6.8	30.10	11.47	76.3	0.20	1.5	26.0	41.3	61.4	3.2
	9	-109	7.0	30.45	12.09	68.3	0.27	6.7	26.8	41.7	44.2	6.2
SALT ORGANIC	15	-136	7.1	31.16	12.37	68.7	0.30	13.9	24.9	30.7	32.6	2.4
	21	-182	7.0	30.29	13.22	56.8	0.50	13.9	21.2	49.8	32.5	9.9
	27	-173	7.0	29.64	21.30	50.2	0.55	18.3	54.7	75.6	43.5	8.3
	33	-203	7.2	28.50	11.74	43.9	0.97	27.1	160.9	195.5	81.3	4.2
	39	-175	7.2	28.25	12.05	33.3	1.17	20.9	190.7	244.8	115.1	3.0
SALT	45	-192	7.3	27.58	11.44	29.1	1.23	12.7	167.3	188.4	103.2	2.5
	45	-192	7.3	27.58	11.44	29.1	1.23	12.7	167.3	188.4	103.2	2.5

Table C2. Environmental means and statistical analysis of physicochemistry.  
 (F = Freshwater Marsh, I = Intermediate Marsh, B = Brackish Marsh, S = Organic Salt Marsh, S' = Mineral Salt Marsh Soils).

Variable	Environment	Mean	N	Standard Error	Grouping <sup>†</sup>	Analysis of Variance				
						Source	D.F.	S.S.	H.S.	F
Eh (mv)	F	-412	40	18.0	A	Environment	4	972417.4	243104.4	32.2 <sup>**</sup>
	I	-81	49	15.3	B	Error	220	1466054.9	7548.6	
	B	-156	48	10.2	C	Total	224	2033072.4		
pH	S	-162	48	9.7	C					
	S'									
SPC (µmhos)	F	6.3	40	.06	C	Environment	4	19.8	4.9	29.1 <sup>**</sup>
	I	7.1	48	.05	A	Error	220	37.4	0.2	
	B	7.2	48	.06	A	Total	224	57.1		
CL ( $\mu$ /cc)	S	6.9	48	.07	B					
	S'	7.1	40	.07	A					
D ( $\mu$ /cc)	F	0.64	40	.03	D	Environment	4	33460.8	8363.2	861.2 <sup>**</sup>
	I	3.56	48	.10	C	Error	219	2127.2	9.7	
	B	18.40	48	.33	A	Total	223	35587.9		
S' ( $\mu$ /cc)	S	29.80	48	.71	A					
	S'	29.50	40	.66	A					
H <sub>2</sub> O (%)	F	0.14	16	.03	D					
	I	1.14	16	.20	D					
	B	5.72	16	.40	C					
S	S	11.16	16	.90	B					
	S'	12.12	16	3.30	A					
S <sub>1</sub>	F	0.10	8							
	I	0.08	8							
	B	0.12	8							
S <sub>2</sub>	S	0.20	8							
	S'	0.66	8							
S <sub>3</sub>	F	87	40	.9	A	Environment	4	36643.4	9160.9	115.20 <sup>**</sup>
	I	87	48	0.4	A	Error	219	1717.9	79.5	
	B	83	48	0.5	B	Total	223	54058.3		
S <sub>4</sub>	S	77	48	0.7	C					
	S'	53	40	2.6	D					

<sup>\*\*</sup>Highly significant at P less than 0.01.

<sup>†</sup>Means with the same letter are not significantly different.

Table C3. Environmental means and statistical analysis of sulfides and extractable nutrients expressed on a volumetric basis. (F = Freshwater Marsh, I = Intermediate Marsh, B = Brackish Marsh, S = Organic Salt Marsh, S' = Mineral Salt Marsh Soil).

Variable	Environment	Mean	N	Standard Error	Grouping <sup>a</sup>	Source	Analysis of Variance			
							DF	S.S.	M.S.	F
	F	0.0	37	.1	B	Environment	4	11092.8	2773.2	11.0
	I	6.4	41	.5	B	Error	168	47435.6	292.2	
	B	19.3	40	.7	A	Total	192	58508.4		
SS ( $\mu\text{g/cc}$ )	S	21.5	40	.4	A					
	S'	14.4	40	.8	A					
	F	1.0	37	.2	B	Environment	4	18170.3	4542.5	19.5*
	I	4.2	41	.3	B	Error	168	439009.8	2335.2	
	B	15.7	40	.5	B	Total	192	60720.2		
CS ( $\mu\text{g/cc}$ )	S	15.6	40	.7	B					
	S'	84.1	40	1.5	A					
	F	2.1	39	.2	C	Environment	4	362456.7	85614.2	28.7*
	I	9.8	49	.5	C	Error	219	652487.4	2919.4	
	B	38.8	48	.9	B	Total	223	994934.1		
TS ( $\mu\text{g/cc}$ )	S	38.5	40	.7	B					
	S'	111.0	48	1.9	A					
	F	17.4	39	.7	B	Environment	4	120729.8	3037.5	34.4*
	I	2.8	49	.9	C	Error	219	191206.0	873.4	
	B	4.6	1.4	B	C	Total	223	311495.8		
EE ( $\mu\text{g/cc}$ )	S	11.7	40	3.0	B					
	S'	12.1	40	1.3	A					
	F	5.8	39	.5	A	Environment	4	553.7	118.4	50.9*
	I	1.7	49	.1	D	Error	219	595.2	2.7	
	B	2.6	40	.1	C	Total	223	1148.9		
NH <sub>4</sub> ( $\mu\text{g/cc}$ )	S	2.0	40	.2	C					
	S'	2.9	40	.2	B					
	F	3.9	39	.5	B	Environment	4	2663.1	650.8	35.3*
	I	2.9	49	.3	B	Error	219	4038.0	18.4	
	B	2.8	48	.2	B	Total	223	6641.1		
PO <sub>4</sub> ( $\mu\text{g/cc}$ )	S	3.1	40	.3	B					
	S'	11.4	40	1.2	A					
	F	4.4	39	.4	C	Environment	4	160.7	401.2	17.6*
	I	6.7	49	.3	B	Error	219	3000.3	22.0	
	B	5.6	40	.4	C	Total	223	6604.9		
NH <sub>3</sub> ( $\mu\text{g/cc}$ )	S	6.0	40	.6	B					
	S'	12.1	40	1.3	A					

\* Highly significant at P less than 0.01.

<sup>a</sup> Means with the same letter are not significantly different.

Appendix D. Data tables for bulk density, sulfides and extractable nutrients expressed on a dry weight basis.

Legend

Variable	Symbol	Unit
Depth	Dept	cm
Bulk density	D	g/cc
Soluble sulfides	SS	$\mu\text{g/g}$
Combined sulfides	CS	$\mu\text{g/g}$
Total sulfides	TS	$\mu\text{g/g}$
Extractable iron	Fe	$\mu\text{g/g}$
Extractable manganese	Mn	$\mu\text{g/g}$
Extractable phosphate	PO <sub>4</sub>	$\mu\text{g/g}$
Extractable ammonia	NH <sup>4</sup>	$\mu\text{g/g}$

Table D1. Profiles of sulfides and extractable nutrients expressed on a dry weight basis.

MARSH ENVIRONMENT	Dept (cm)	SS ( $\mu\text{g/g}$ )	CS ( $\mu\text{g/g}$ )	TS ( $\mu\text{g/g}$ )	Fe ( $\mu\text{g/g}$ )	Mn ( $\mu\text{g/g}$ )	PO4 ( $\mu\text{g/g}$ )	NH4 ( $\mu\text{g/g}$ )
FRESHWATER	3	1.6	16.2	17.4	367.8	59.0	67.8	54.5
	9	6.1	19.3	21.9	341.3	42.7	81.7	44.8
	15	15.0	27.7	34.3	189.5	46.3	77.3	54.8
	21	10.3	21.8	26.0	110.3	50.9	50.1	56.7
	27	6.9	12.4	15.9	101.5	41.9	27.8	39.2
	33	7.6	15.4	18.6	143.7	61.9	17.9	39.7
	39	7.1	11.9	16.8	91.7	58.6	9.7	31.1
	45	7.2	15.7	22.8	120.3	89.5	10.7	37.3
INTERMEDIATE	3	19.9	58.1	74.4	136.4	13.7	59.5	54.2
	9	39.4	43.9	71.3	42.5	8.4	34.6	55.7
	15	62.7	40.3	58.4	16.2	12.4	39.9	84.2
	21	96.1	58.3	134.5	15.1	11.8	38.7	86.9
	27	104.5	53.4	141.3	14.2	14.0	26.6	93.4
	33	92.9	52.0	137.9	15.4	16.5	24.7	90.6
	39	101.5	53.6	145.0	14.4	19.1	23.5	86.2
	45	110.7	52.3	151.4	14.8	20.7	22.6	85.5
BRACKISH	3	15.1	252.1	358.0	198.3	11.9	32.1	41.3
	9	47.3	69.2	144.4	65.3	11.2	24.0	33.9
	15	133.1	91.5	234.0	17.3	9.7	24.3	45.5
	21	166.2	114.5	292.6	18.9	13.7	27.4	55.8
	27	215.9	147.1	370.2	16.0	13.6	24.3	51.9
	33	254.2	119.5	407.2	19.2	14.5	22.8	47.3
	39	266.2	137.9	448.1	23.8	18.8	19.4	55.0
	45	298.5	146.0	469.7	15.4	20.1	16.6	51.4
SALT (organic)	3	21.3	120.0	141.3	235.3	14.8	19.3	25.4
	9	24.9	47.1	72.0	88.1	10.4	19.5	27.1
	15	65.5	55.4	107.3	31.5	7.3	10.9	24.8
	21	75.9	75.3	151.3	21.5	9.1	14.3	32.2
	27	108.2	74.3	182.6	37.0	12.2	21.3	32.2
	33	100.6	74.4	175.0	27.8	8.1	10.5	39.3
	39	171.8	85.4	321.2	33.1	9.5	14.9	47.1
	45	242.3	98.7	341.1	18.1	11.7	16.4	41.3
SALT (mineral)	3	7.3	125.0	198.9	296.2	15.5	30.1	33.1
	9	24.4	97.4	151.1	160.3	8.3	22.6	35.5
	15	45.8	82.1	166.8	100.8	7.9	15.4	29.3
	21	27.4	41.7	97.9	63.3	4.9	19.0	19.5
	27	32.7	97.9	135.3	77.8	4.6	14.9	15.9
	33	27.7	168.4	199.7	83.1	4.3	14.3	18.1
	39	17.7	162.0	208.0	97.8	2.5	16.3	12.9
	45	10.3	136.0	151.1	83.9	2.0	17.6	15.0

\*  $N_{\text{S}} \neq N_{\text{SS}} + N_{\text{CS}}$

Table D2. Environmental means for bulk density, sulfides and extractable nutrients. (F = Freshwater Marsh, I = Intermediate Marsh, B = Brackish Marsh, S = Organic Salt Marsh, S' = Mineral Salt Marsh Soils).

Environment	D (g/cc)	SS ( $\mu\text{g/g}$ )	GS ( $\mu\text{g/g}$ )	TS ( $\mu\text{g/g}$ )	Fe ( $\mu\text{g/g}$ )	Mn ( $\mu\text{g/g}$ )	PO <sub>4</sub> ( $\mu\text{g/g}$ )	NH <sub>4</sub> ( $\mu\text{g/g}$ )
F	0.10	8.0	17.8	21.7	184.9	55.6	43.7	45.0
I	0.08	78.2	51.4	119.2	33.7	14.5	33.9	80.3
B	0.12	174.6	134.8	340.5	45.6	14.2	23.9	48.3
S	0.20	101.4	78.9	186.5	61.6	10.4	15.9	34.2
S'	0.66	24.6	100.4	172.5	118.5	6.4	19.4	22.3

\*  $N_{TS} \neq NSS, NGS$ .

VITA

Richard S. Hatton was born December 6, 1954 in Bulawayo, Rhodesia. He graduated from Milton High School, Bulawayo, Rhodesia in 1972. In 1973 he completed his military service in the Rhodesian Army.

In 1974 he started a chemical program at the University of Natal, Pietermaritzburg, South Africa. He received his B.S. Degree in December of 1976 and in 1977 he obtained his B.S. Honours Degree in chemistry.

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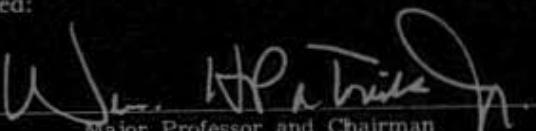
EXAMINATION AND THESIS REPORT

Candidate: Richard S. Hatton

Major Field: Marine Sciences

Title of Thesis: ASPECTS OF MARSH ACCRETION AND GEOCHEMISTRY:  
BARATARIA BASIN, LOUISIANA

Approved:



Major Professor and Chairman



Dean of the Graduate School

EXAMINING COMMITTEE:







Date of Examination:

March 25, 1981