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Sedimentation Processes in Selected Coastal Wetlands From the Gulf of Mexico and Northern Europe.

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SEDIMENTATION PROCESSES IN SELECTED COASTAL WETLANDS FROM THE GULF OF MEXICO AND NORTHERN EUROPE

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ABSTRACT

Sediment cores were collected from coastal wetlands from the Gulf coast of North America and northern Europe to study accretion rates and heavy metal accumulation. There was a significant decrease in vertical accretion rates from low to high marsh for Gulf coast samples. Contrary to previous results, these low tidal range sites did not have negative accretion balances. Northern European samples demonstrated the utility of Chernobyl ^{137}Cs as a sediment marker. There were large differences in sediment characteristics and accretion rates between Polish and western European samples. Vertical accretion rates based on ^{210}Pb were lower than ^{137}Cs rates for most cores. The ^{210}Pb rates included effects of compaction and decomposition, and profiles of bulk density and organic content confirmed this. All European sites had positive accretion balances. Multiple regression analyses of a large data set showed that vertical accretion rates were best described by a regression using relative sea level rise, surface organic content, sediment bulk density, position within the marsh, and the interaction between relative sea level rise and position. The significance of the interaction term indicated that low marsh sites responded differently to increases in relative sea level rise than middle and high marsh sites. There was a much higher correlation between vertical accretion and organic accumulation than between vertical accretion and mineral accumulation. A computer model, using an annual cohort approach, successfully simulated sedimentation processes, including surface sediment deposition, below-ground production, decomposition, and compaction. Sensitivity analyses indicated that the most important factors affecting model-generated accretion

rates were: pore space, mineral matter deposition, initial elevation, sea level rise, and below-ground production. Chronologies of sediment heavy metal concentrations for high and low marsh cores from northern Europe showed very good agreement, indicating that sediment profiles represent historic inputs of heavy metals. Some of the sediments had very high heavy metal concentrations, with peak sediment concentrations up to five times greater than found in the oldest sediment samples. Metal concentrations have recently decreased in the cores from St. Annaland and Stiffkey Marshes but remained high throughout the upper part of the cores from the Oder River.

INTRODUCTION

SEDIMENT ACCRETION

In the last decade, wetlands have come under increasing protection as the general public becomes more aware of their ecological importance (National Wetlands Policy Forum 1988). Despite legal protection, the long term survival of tidal wetlands is not a simple issue due, in part, to the complex ecological processes which naturally maintain tidal marshes. Because of their location at the land/ocean interface, coastal wetlands must maintain an elevation within the tidal range, or they will cease to function as wetlands (Baumann et al. 1984, Salinas et al. 1986). If the surface of the marsh becomes too low, plants can become stressed, and the marsh may disappear (Mendelssohn et al. 1981). Conversely, if the marsh collects excessive sediment, its elevation may become so high that upland species will colonize and may outcompete wetland plants.

The relative elevation of coastal wetlands is a function of a variety of factors, including eustatic sea level rise, subsidence (which in itself includes many factors), and the vertical accretion of mineral and organic sediments. Present estimates of eustatic sea level rise range from 0.12 cm/yr to 0.24 cm/yr (Gornitz et al. 1982, Peltier and Tushingham 1989, Woodroffe 1993). Recent estimates of future eustatic sea level rise vary from 20 to 115 cm by the year 2100 (Woodroffe 1993), and these increases are likely to have enormous impacts on coastal wetlands world-wide (Titus 1986, 1991). Tidal marshes have developed during periods of relatively low sea level rise and rapid accumulation of mineral and organic sediments. They can withstand

some increases in sea level rise; however, at present there is no clear understanding of the ability of marshes to compensate for increased rates of sea level rise (Orson et al. 1985).

In recent years there have been many studies of accretionary processes in coastal wetlands. Much of this work has been done in the Mississippi River deltaic plain, where vertical accretion rates are large, but there are also very high rates of subsidence and coastal land loss (DeLaune et al. 1983, Hatton et al. 1983, Salinas et al. 1986). In other areas of the United States and Europe, where data have been collected on accretion rates, a wide range of results has been found, with accretion rates varying from 0 to 1.5 cm per year. Many of these marshes are accreting at a rate fast enough to compensate for present rates of sea level rise (Harrison and Bloom 1977, Griffin and Rabenhorst 1989, Patrick and DeLaune 1990), but some are being inundated and converted to open water habitats (Stevenson et al. 1985).

SYNTHESIS OF RESULTS FROM ACCRETION STUDIES

Despite the large number of recent studies evaluating sediment accretion rates, very little analysis has been done to synthesize the results from these different studies in order to determine if there are common factors from the various studies that are important in affecting accretion rates. Many factors have been proposed to be important in affecting accretion rates at a given site, but most of these proposed factors are intuitive, and have not been tested. Some of these factors include: plant community and density of vegetation (Richard 1978, Gleason et al. 1979, Stumpf 1983), tidal elevation (Harrison and Bloom 1977, Bricker-Urso et al. 1989), sediment

input from riverine, estuarine and marine sources (Salinas et al. 1986), proximity to the sediment source (French and Spencer 1993), total organic input from local marsh production (McCaffrey and Thomson 1980, Hatton et al. 1983, Bricker-Urso et al. 1989), and relative sea level rise (Redfield 1972, Gehrels and Leatherman 1989). In a recent review of salt marsh accretion rates, Stevenson et al. (1986) have shown that there is a strong correlation between tidal range and accretionary balance (the difference between accretion and subsidence). However, this was the only variable addressed in the study, and it is clear from other work that tidal range is not the only factor that influences accretionary balance.

There are a number of reasons that have made the integration of results from various studies so difficult, including: 1) studies of coastal wetlands have been conducted by scientists with a wide range of interests and perspectives; 2) different techniques have been used measure accretion rates and these techniques have resulted in many different types of data; 3) the vocabulary used in these studies has often been confusing and ambiguous; and 4) the processes that affect sediments vary over both time and depth simultaneously, and this complex interaction leads to results that are difficult to interpret. I will cover each of these issues in more detail below.

Different approaches

Studies of wetland sediments have been completed by researchers from many different fields with widely different interests and viewpoints. These studies have been done to evaluate accretion rates in coastal wetlands, but often they have also focused on other issues, including: sediment stratigraphy, heavy metal analysis,

historic shoreline analysis, or plant ecology. For example, geologists often address marsh stability from a long-term sedimentary viewpoint, studying sediment stratigraphy (Nichols et al. 1991) and erosion rates (van Eerdt 1985). Hydrologists focus on the transport of material in and out of the marsh, or other issues, such as tidal asymmetries (Pethick 1980, Healey et al. 1981). Ecologists are interested in many different topics related to wetland sediments, from habitat loss (Templet and Meyer-Arendt 1988) and plant stress (Mendelssohn et al. 1981), to accretion balances (Baumann et al. 1984). Policy analysts have also studied the effects of sea level rise on wetlands, determining rates of loss and potential societal and economic impacts (Titus 1986, 1991). With all of these different viewpoints related to the same issue, there is a large potential for misunderstanding between researchers from different fields.

Terms/vocabulary

In addition, communication between the various groups of researchers has been poor because common terms have only been vaguely defined or have multiple meanings depending on who is using the term. What one person understands as "accretion" is "accumulation" to another researcher. "Subsidence" has been used to mean many different processes, from the oxidation of surface peats to compaction of deep Pleistocene sediments. In order to increase our understanding of these issues, researchers should adopt a common vocabulary. At the very least, particularly loaded words, such as subsidence, should be used with a single meaning by all researchers.

I have attempted to define most of the basic terms in the "Proposed definitions" section.

Methods used to measure accretion

To further complicate the problem, many methods have been used to measure accretion rates, and each method gives a result relative to a different time period (DeLaune et al. 1978, McCaffrey and Thomson 1980, Cahoon and Turner 1989, Boumans and Day 1993). There are two basic types of measurements that have been made for accretion studies. Most measurements use some variation of a marker horizon, however the time period over which rates have been measured varies from a few months or years (feldspar markers) to 20-30 years (^{137}Cs), 100 years (^{210}Pb), or 1000's of years (^{14}C). The second group of techniques evaluate changes in relative elevation. These techniques include surveying (Anderson et al. 1981) and sedimentation-erosion tables (Boumans and Day 1993). They measure net changes in sediment surface elevation relative to a benchmark of some type, not just accretion above a marker, so some components of subsidence may be a part of these measurements, depending on the type of benchmark that is used.

Additional techniques, such as the evaluation of geomorphic features (Allen and Rae 1988), pollen analysis (Clark and Patterson 1985) and radiocarbon dating (Pethick 1981), have also been used to cover much longer periods. Pollen analysis and radiocarbon dating are simply long-term marker horizon techniques, but are not used as commonly as the other techniques outlined above. Each of these different methods gives us important information; however, in most cases the data can not be

compared directly to data from another method. The challenge is to combine the information from all of these different methods into a single unifying understanding of sediment dynamics.

Relationship of time and depth

The time period covered by a particular dating procedure is very important in interpreting results of accretion studies because as material accumulates on the surface of the marsh, other below-ground processes (including organic production, physical compaction, and decomposition of organic material) occur and affect the absolute level of the marsh. All of these processes are occurring simultaneously, and it is difficult to determine from a single measurement the importance of each of the various processes for a given site. Marker horizon studies give information on short-term accretion rates, which probably only reflect accumulation events on the surface of the marsh. Whereas, more long-term measurements (radioisotope dating, such as ^{137}Cs and ^{210}Pb) integrate rates over time, reflecting accumulation at the surface of the marsh, as well as processes that occur within the sediment column (Stevenson et al. 1986). The results from these measurements are average rates relative to a given time period. Processes at depth are continually occurring and are integrated throughout the depth of the sediment over time. A technique that covers a longer time period, such as ^{14}C, will usually produce lower calculated accretion rates because of sediment compaction and decomposition of sediment organic material. Because of these complex interactions of time and depth it is not possible to directly compare accretion measurements that have been made with different techniques, and the

relative magnitude of each process can not be identified without modelling the interaction of time, depth and below-ground processes.

PROPOSED DEFINITIONS

In order to clarify the various processes that are involved in sedimentation dynamics within coastal wetlands, I will define the major terms that are commonly used in this field. Many of these definitions are adapted from a study done by Penland et al. (1988) which focused on processes in the Mississippi Deltaic Plain. The factors that they reviewed are important in any coastal area and cover many of the basic processes which I have outlined in the preceding section. I have regrouped and modified their factors (Penland et al. 1988) into the following categories:

1) **Eustatic sea level rise**, or global changes in sea level due to changes in the mass of water in the oceans, ocean volumes and temperature changes. Current estimates of eustatic sea level rise are based mostly on tidal gauge analysis and range from 0.12 to 0.24 cm/yr (Gornitz et al. 1982, Peltier and Tushingham 1989, Woodroffe 1993). Predicted increases in eustatic sea level rise range from 20 to 115 cm by the year 2100 (Woodroffe 1993).

2) **Compaction of deep sediments** (everything except Holocene sediments) includes both tectconic downwarping of basement sediments and compaction of Tertiary, Pleistocene, and older sediments. Tectonic downwarping is probably only a factor in major depositional areas such as large deltas, where the load from massive amounts of sediments cause the bottom of the sedimentary basin to warp. In most

studies of modern sediments, very little is known about this factor, so it is often treated as an unknown or something outside of the system.

3) **Compaction of Holocene sediments** includes dewatering of sediments (primary consolidation), rearrangement of mineral structure of the sediment and subsequent loss of volume (secondary consolidation), and the decomposition of organic matter in the sediment. This process is probably the most important process affecting subsidence in most coastal areas; however, it is also a very difficult process to evaluate in sediments because it is the combination of the three factors listed above.

In most coastal wetland sediments, decomposition and primary consolidation are the most important factors that affect compaction. The relative importance of the two probably depends on the type of sediment involved. For example, in highly organic sediments, organic matter may be most important in creating the structure of the sediment (McCaffrey and Thomson 1980, Hatton et al. 1983). In this case decomposition is probably extremely important, while in more mineral soils, the dewatering of unconsolidated sediments is probably most important.

4) **Tectonic activity** can affect wetland sediments via growth faults in deltaic areas (Penland et al. 1988), and other faults in tectonically active areas (Reed 1989, Darienzo and Peterson 1990). Very little work has been done in this area, and this effect is extremely difficult to evaluate because of its unpredictable and irregular nature.

5) **Fluid withdrawals and other direct anthropogenic effects** can be important locally, and the most common effect is usually due to the withdrawal of

either hydrocarbons or water. This results in abnormally high compaction rates in deep sediments (Poland 1969). I have also included in this factor the term from Penland et al. (1988), "localized consolidation," which they defined as consolidation "due to the weight of minor land forms and engineering structures."

6) **Accretion** is the vertical accumulation of material on the surface of a marsh or mudflat. This includes the accumulation of both organic and mineral matter. In order to clarify different types of measurements, accretion should be used specifically for vertical measurements; whereas, **accumulation** should be used for mass-based rates.

In addition to the basic terms above, the following general terms also need clarification:

Subsidence has been used to mean many different processes by different researchers. Generally it refers to a decrease in land elevation relative to some relative elevation (benchmark or water level), but the importance of the various sub-processes differ according to particular usages. Geologists generally use it to mean processes that occur deep in the sediment column, similar to factor 2 above. However, some may also include tectonic activity in subsidence. Other researchers use subsidence to denote surface processes, especially the oxidation of organic sediments or sediment dewatering. I feel that the best use of this term is in the broadest sense of Penland et al. (1988), as the "downward displacement of the . . . [coastal] surface with respect to a vertical datum." The processes above can be used

to describe specific factors, and subsidence is used as the sum effect of factors 2 through 5 above. Using this terminology:

relative sea level change = subsidence + eustatic sea level rise - accretion

Positive values for relative sea level change indicate an increase in the average flooding depth or average relative sea level; whereas, negative values indicate a decrease in relative sea level and an increase in the elevation of the marsh.

Submergence is also a term that has been used commonly to mean a variety of processes. Some authors use it synonymously with subsidence; however, this is not appropriate. Submergence should be used to refer to a negative accretion balance, i.e., a sediment surface that is being flooded to greater and greater depths.

Accretion balance refers to the net change in relative sea level at a particular site over time. If a coastal wetland is to remain at the same elevation over time, then accretion must be equal to subsidence plus eustatic sea level rise.

HEAVY METAL ACCUMULATION IN WETLAND SEDIMENTS

As sediments accumulate on the surface of the marsh, they also accumulate other material with them, including heavy metals and other pollutants. Previous work with wetland sediments from Connecticut (McCaffrey and Thomson 1980), Rhode Island (Bricker 1993), the Netherlands (Zwolsman 1993) and other areas has documented the usefulness of wetland sediments for developing heavy metal chronologies.

Presently, anthropogenic inputs of heavy metals to the environment far exceed natural inputs (Nriagu and Pacyna 1988), and these inputs may pose large health risks

in areas where metals accumulate. Impacts of heavy metal pollution to coastal and estuarine areas could be substantial because of a variety of inputs to these areas. Sources include riverine inputs, local runoff, atmospheric deposition, and coastal waters. Local runoff and riverine inputs can carry municipal sewage (Galloway 1979) and industrial effluent (Gross 1978). Both of these inputs have been shown to carry high levels of heavy metals in many areas (Helz 1976). Coastal waters are usually relatively clean and probably do not have a major effect on heavy metal concentrations in most estuarine situations. Recent interest in protecting coastal areas and waters has sharply decreased heavy metal concentrations in coastal waters and increased monitoring of pollutants in these environments (O'Connor and Ehler 1991, Valette-Silver et al. 1993).

In addition to monitoring present inputs of pollutants into coastal areas, there is great interest in determining historic inputs and accumulation rates in coastal environments. Sediments have been used extensively as indicators of chronological inputs to coastal areas, including both subtidal sediments (Santschi et al. 1984, Schmidt and Reimers 1991, Swartz et al. 1991, Valette-Silver 1993) and wetland sediments (Griffin et al. 1989, Bricker 1993). Recent studies in coastal wetlands have successfully developed chronologies of metal inputs (McCaffrey and Thomson 1980, Bricker 1993, Zwolsman et al. 1993). One of the consistent findings of many of these studies is the recent decrease in heavy metal concentrations in sediments (Macdonald et al. 1991), especially for lead (Evans 1982, Trefry et al. 1985, Dörr et

al. 1991). These results have also been confirmed with decreases in metals from snow deposits from Greenland (Boutron et al. 1991).

Salt marshes are sinks for sediments and other materials, and they are excellent areas to study the pollution chronology of coastal and estuarine systems because usually there is little bioturbation of salt marsh sediments (Stumpf 1983). In studies of areas where data are available on historic inputs to the coastal environment, the chronology that is found in the sediments usually closely matches historical inputs (McCaffrey and Thomson 1980, Bricker 1993, Zwolsman 1993). However, in many areas, very little information exists on histories of metal inputs, and in these cases, we must rely upon sediment chronologies to attempt to reconstruct the history of pollution for a given area (Alongi et al. 1991).

RESEARCH APPROACH

My approach to evaluating sedimentation processes in coastal wetlands was to look closely at accretion rates from two areas, the northern Gulf of Mexico and northern Europe, and to use two methods (multiple regression analyses and simulation modelling) to synthesize the results from these field studies and from other published studies of wetland sedimentation. In addition, I used the cores from northern Europe to investigate changes in historical levels of heavy metals in the sediments.

I chose to look at the northern Gulf of Mexico because extensive research has been completed evaluating accretion rates in the Mississippi River deltaic plain; however, very little research has been done outside of this area. By measuring accretion rates from other areas along the Gulf, I have been able to answer general

questions about accretion processes within the Gulf coast region and to evaluate accretion rates under a low tidal regime. The samples from northern Europe were collected to determine the useful of ^{137}Cs from the 1986 Chernobyl nuclear accident as a sediment marker. In addition, I used data from one of these cores in the calibration of the simulation model.

I have used two techniques in my attempt to synthesis the results from studies of accretion rates in coastal wetlands. Chapter 3 outlines a statistical analysis of accretion processes from a variety of sites, including my own data and results from previously published studies. Multiple regression analyses were used to predict accretion rates using sediment characteristics and other variables. Also, organic and mineral accumulation rates were analyzed to see if there are consistent trends in accretion processes from all of the sites which were included in the analysis. In Chapter 4, I used a computer model to simulate processes in the sediment column. The model addresses assumptions about sedimentation processes, and the results from the sensitivity analyses were used to identify important variables that affect accretion processes.

Finally, the cores from Europe were evaluated for heavy metal concentrations versus depth, and the depths were converted to dates, based on the ^{137}Cs dating of these cores. I compared profiles from the high and low marsh cores to verify the chronologies from a given site. General trends in the chronologies were evaluated in order to get an index of the trends in heavy metal pollution over time in these coastal

wetlands and to see if the sediments document recent decreases in heavy metal pollution.

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CHAPTER 1

SEDIMENT ACCRETION RATES FROM FOUR COASTAL WETLANDS ALONG THE NORTHERN GULF OF MEXICO: ARANSAS NWR, SAN BERNARD NWR, BILOXI BAY, AND THE FLORIDA KEYS

INTRODUCTION

Extensive data have been collected over the last decade concerning sediment accretions rates in coastal marshes because of the potential impacts of increases in sea level rise on these habitats. Recent estimates of future eustatic sea level rise vary from 20 to 115 cm by the year 2100 (Woodroffe 1993). There are some predictions that future increases will not be as great as previously suggested (Meier 1990). Most of the coastal wetlands that have been studied for sediment accretion rates are keeping pace with current rates of sea level rise (Griffin and Rabenhorst 1989; Patrick and DeLaune 1990, Anderson et al. 1992, Thom 1992, Craft et al. 1993); however, some are experiencing submergence (DeLaune et al. 1983; Baumann et al. 1984; Stevenson et al. 1985). Additionally it is not clear how great of an increase coastal wetlands will be able to withstand, if rates of sea level increase in the future, as predicted (Orson et al. 1985). Despite the growing number of studies of wetland sedimentation rates that have been completed, there are still many unknowns concerning the relationship of accretion rate, sea level rise, subsidence, and other variables.

The maintenance of the relative elevation of the marsh is a complex process, and many factors may be involved, including variables on a variety of scales. Some

of the key factors that have been proposed to affect accretion rates include: relative elevation (Letzsch and Frey 1980; Pethick 1981), tidal range (Harrison and Bloom 1977), proximity of the sediment source and creek system (Stoddart et al. 1989), plant community and density of vegetation (Richard 1978, Gleason et al. 1979, Stumpf 1983), and local rates of subsidence (Redfield 1972). In a review of accretion processes and sea level rise, Stevenson et al. (1986) showed a positive linear relationship between tidal range and net accretion rate; however all of the sites with small tidal ranges that they studied were experiencing relatively high rates of subsidence and, as a result, had very low (in fact all were negative) rates or net accretion. This lead us to ask the question - are low net accretion rates really characteristic of low tidal range sites.

The northern Gulf coast is an ideal place to study accretion rates under a low tidal regime. There are an abundance of coastal wetlands, and the entire coast has a low tidal regime. Many studies have been completed of accretion rates in Louisiana coastal wetlands (DeLaune et al. 1978, Hatton et al. 1983, Knaus and Van Gent 1989, Nyman et al. 1990). These areas have low tidal ranges but are also characterized by extremely high rates of subsidence (and hence relative sea level rise). Very few studies have been completed on the Gulf coast outside of Louisiana (Lynch et al. 1989). In order to evaluate sediment accretion processes in low tidal regime marshes, I chose to measure accretion rates from four different coastal wetlands in the Gulf of Mexico. Our sites include three tidal salt marshes from Aransas National Wildlife

Refuge (NWR), TX; San Bernard NWR, TX; and Biloxi Bay, MS, and a group of tidal mangrove wetlands in the Florida Key (Figure 1.1).

In addition to addressing the question of tidal range and accretion rates, this data set has allowed us to evaluate additional questions concerning marsh accretion processes, including some of the relationships that have been proposed by other researchers and discussed above. I am specifically interested in the relationships between vertical accretion, organic matter accumulation, and mineral matter accumulation rates. These correlations have been used previously to help interpret the roles of organic and mineral matter in coastal marshes (Nyman et al. 1993; Bricker-Urso et al. 1989), and I hoped to see if similar trends could be found from sites across the Gulf of Mexico. Finally, vertical accretion data from these coastal wetland will identify areas which may be vulnerable to potential increases in global sea level.

METHODS

Site locations

At each of the four sites below, six cores were collected. At the first three sites, cores were collected along two transects from the low marsh to high marsh. Our low marsh station was situated approximately 10 M from the creek bank, in order to avoid abnormally high rates that have been found in "streamside" habitats (Hatton et al. 1983). The mid and high marsh stations were each located 20 M farther into the marsh. In the Keys, I was not interested in evaluating changes along transects through the wetland, but focused on areas dominated by Red and Black mangrove

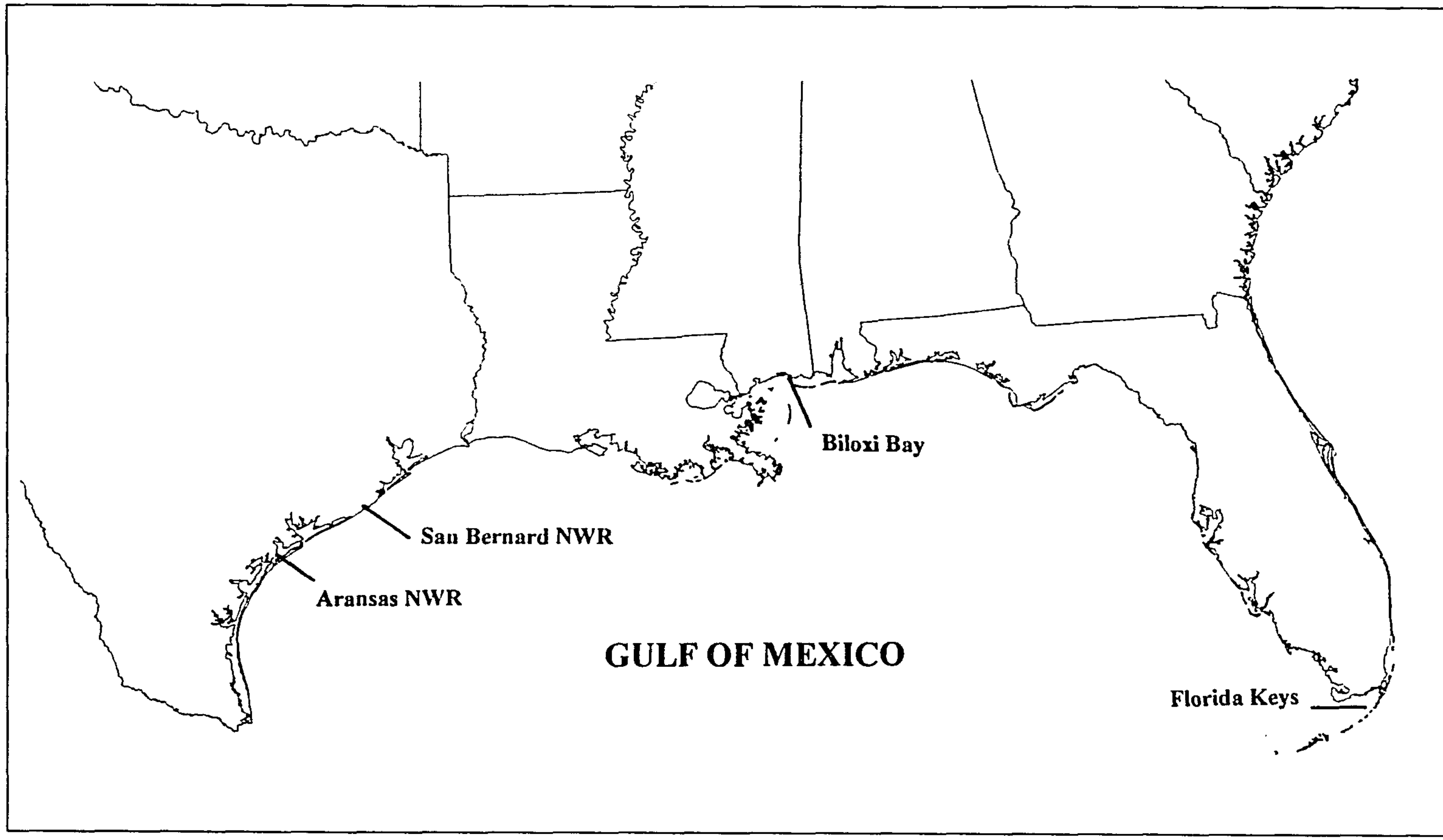

Figure 1.1. Location of sampling sites along the northern Gulf of Mexico.

species. Because of this, I collected three samples from different areas dominated by each of these species. Our study sites were:

a. **Aransas National Wildlife Refuge, TX** - This old barrier island marsh is located near Austwell, TX. Both transects were collected from the seaward side of Mustang Lake, just north of False Live Oak Point. The low marsh stations here were dominated by *Spartina alterniflora*, with very little *Grindelia* sp., and *Batis maritima.* Mid and high marsh stations consisted of *S.alterniflora*, *Salicornia virgnica*, and *Grindelia* sp.

b. **San Bernard National Wildlife Refuge, TX** - Samples were collected from the backside of the barrier peninsula, southwest of Raccoon Point (transect #1), and from the edge of the mainland, near the Cedar Lakes Backwater on the western edge of the refuge (transect #2). The vegetation at the low marsh stations on both transects consisted mostly of *S. alterniflora*, with some *B. maritima.* The mid and high marsh stations were *Distichlis spicata*, *S. alterniflora*, and *B. maritima.*

c. **Biloxi Bay, MS** - Samples were collected from Weeks Bayou, near the mouth of Biloxi Bay (transect #1), and from upper Biloxi Bay (transect #2). As is typical in marshes from this area of the gulf (Eleuterius 1972), *Juncus roemerianus* and *S. alterniflora* were found in the low marsh, and *Spartina patens* and *S. alterniflora* in the mid and high marsh. This was the only marsh that I sampled that had extensive amounts of *J. roemerianus.*

d. **Florida Keys, FL** - Samples were collected from Lignum Vitae Key, Plantation Key, and North Key Largo. These samples were collected by Jack

Meeder, National Audubon Society, Florida. At each Key, one sample was collected from the fringe area, approx. 10 M from open water in areas dominated by red mangroves (*Rhizophora mangle*). A second core was taken in the same vicinity, but in areas dominated by black mangroves (*Avicennia germinans*).

In 4 of the 24 cores, ^{137}Cs profiles did not yield a distinct peak, and rates of accretion could not be calculated. Cores that were not dated came from:

1) low marsh station from transect #2 at Aransas NWR

2) mid marsh station from transect #2 at Aransas NWR

3) low marsh station from transect #1 at Biloxi Bay

4) red mangrove station at Lignum Vitae Key

Because of these omissions, two of our six transects were incomplete. These transects have been dropped from the calculation of averages for each site - and from the analysis of changes along the transects; however, the remaining samples from these sites were included in the correlation analyses of sediment characteristics and sedimentation rates.

Collection methods and sample analyses

Cores were collected in 15 cm diameter aluminum cylinders to a depth of 30-50 cm. Cores were sectioned every 2 cm, and individual sections were dried at 80 deg C, weighed, and crushed with a mortar and pestle. ^{137}Cs activity of the bulk sediment was counted with a Lithium Drifted Geranium detector and multi-channel analyzer. ^{137}Cs is a product of nuclear weapons testing and does not occur naturally. Significant levels of this isotope first appeared in the atmosphere in the early 1950's

with the peak quantities detected in 1963/64 (Ritchie and McHenry 1990). Sediment profiles are dated based on the 1963 peak in ^{137}Cs, giving an average sedimentation rates for the period from 1963 to the present (Delaune et al. 1978).

In addition to the ^{137}Cs analysis, 5 g subsamples were taken from each section for combustion at 400 deg C in order to determine organic content (loss-on-ignition). Using the organic/mineral content of the sediment and the bulk density, linear-based accretion rates were converted to mass based rates and divided into organic and mineral components. Profiles of organic content and sediment bulk density were constructed in order to evaluate changes in these parameters with depth. These profiles were also converted to volume distribution profiles, using specific gravity conversions for organic matter and mineral matter of 1.14 and 2.61 g/cm^3 respectively.

Statistical analyses were performed using SAS 6.04. I used a two-way ANOVA analysis without replication in testing for differences between different stations along the transects. The interaction term was used as the error term, and the Tukey test for non-additivity was used to determine the significance of the interaction term (Sokal and Rohlf 1981).

RESULTS

Sediment characteristics and profiles

Large differences were found in sediment characteristics between the different sites (Table 1). Cores from mangroves in Florida had the lowest bulk densities and

Table 1.1. Sediment characteristics for cores from transects in 3 salt marshes and from mangrove areas in the Florida Keys. Values given are for the top 10 cm of the core. Standard deviations are given in parentheses.

Site	Bulk Density (g/cm^3)	Organic Content (percent)
Aransas NWR	0.70 (0.226)	14.4 (5.20)
San Bernard NWR	0.90 (0.165)	9.1 (1.84)
Biloxi	0.28 (0.039)	24.3 (3.98)
Florida Keys		
Red Mangroves	0.16 (0.013)	55.5 (10.01)
Black Mangroves	0.15 (0.024)	61.3 (8.86)

the highest organic content. The sediments from the coastal salt marshes in Aransas and San Bernard both had very high bulk densities; whereas Biloxi Bay sediments were mid range. Organic content was negatively correlated with bulk density (Figure 1.2); although the relationship between these two variables was very different for mangrove sediments than for the salt marsh sediments from Aransas, San Bernard and Biloxi Bay. Previous studies of soil bulk density and organic content have shown similar relationships for sediments from Louisiana coastal marshes (Gosselink, et al. 1984). There were no consistent trends in organic content or bulk density across the marsh transects.

Most cores exhibited decreases in organic content and increases in bulk density with depth (Figures 1.3a and 1.3b). These changes are paralleled in the volume distribution profiles, with relatively less space occupied by organic matter and pore space and more space being occupied by mineral matter as the depth increases. These changes in sediment characteristics with depth are due to below-ground organic matter production, decomposition and compaction throughout the sediment profile. Rae and Allen (1993) have documented similar trends, although it is difficult to separate the relative importance of the three factors above because all are occurring simultaneously. A computer model was used to simulate these processes over both depth and time (Chapter 4).

Vertical accretion rates

Rates of vertical accretion from the individual cores ranged from 0.18 to 0.89 cm/yr. At all of the three sites where samples were collected on a transect through

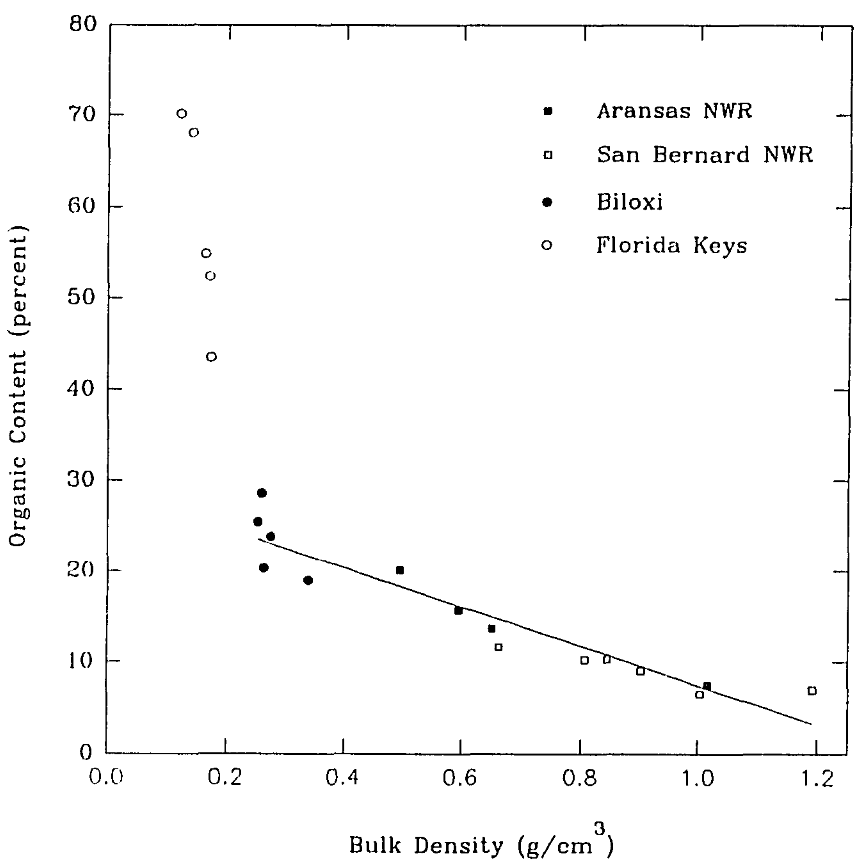

Figure 1.2. Organic content and sediment bulk density for the top 10 cm of cores collected from all sites. Solid line indicates correlation for salt marsh sites only (r^2 = 0.87).

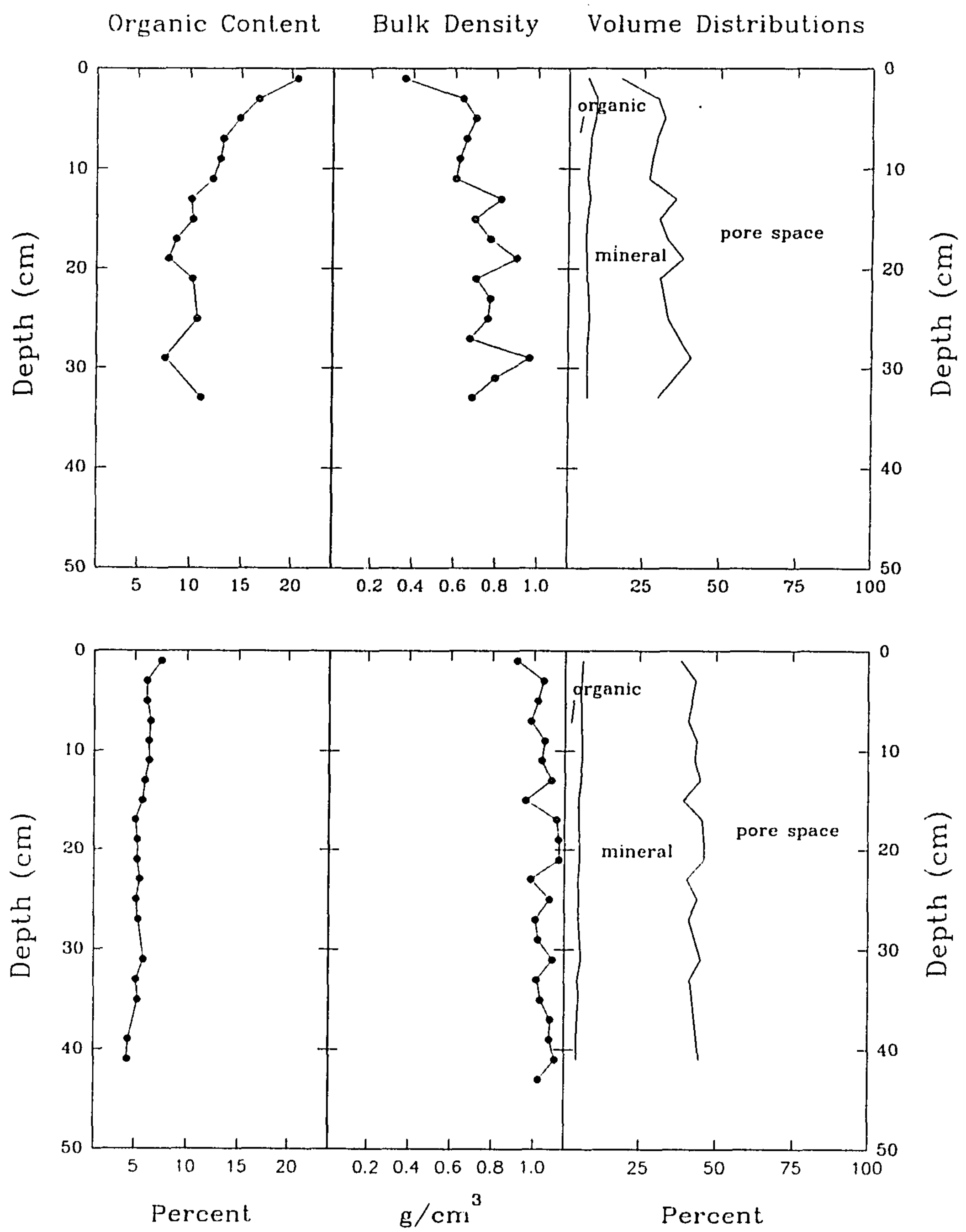

Figure 1.3a. Profiles of organic content, bulk density, and sediment volume distributions for cores from the low marsh at Aransas NWR (top) and San Bernard NWR (bottom).

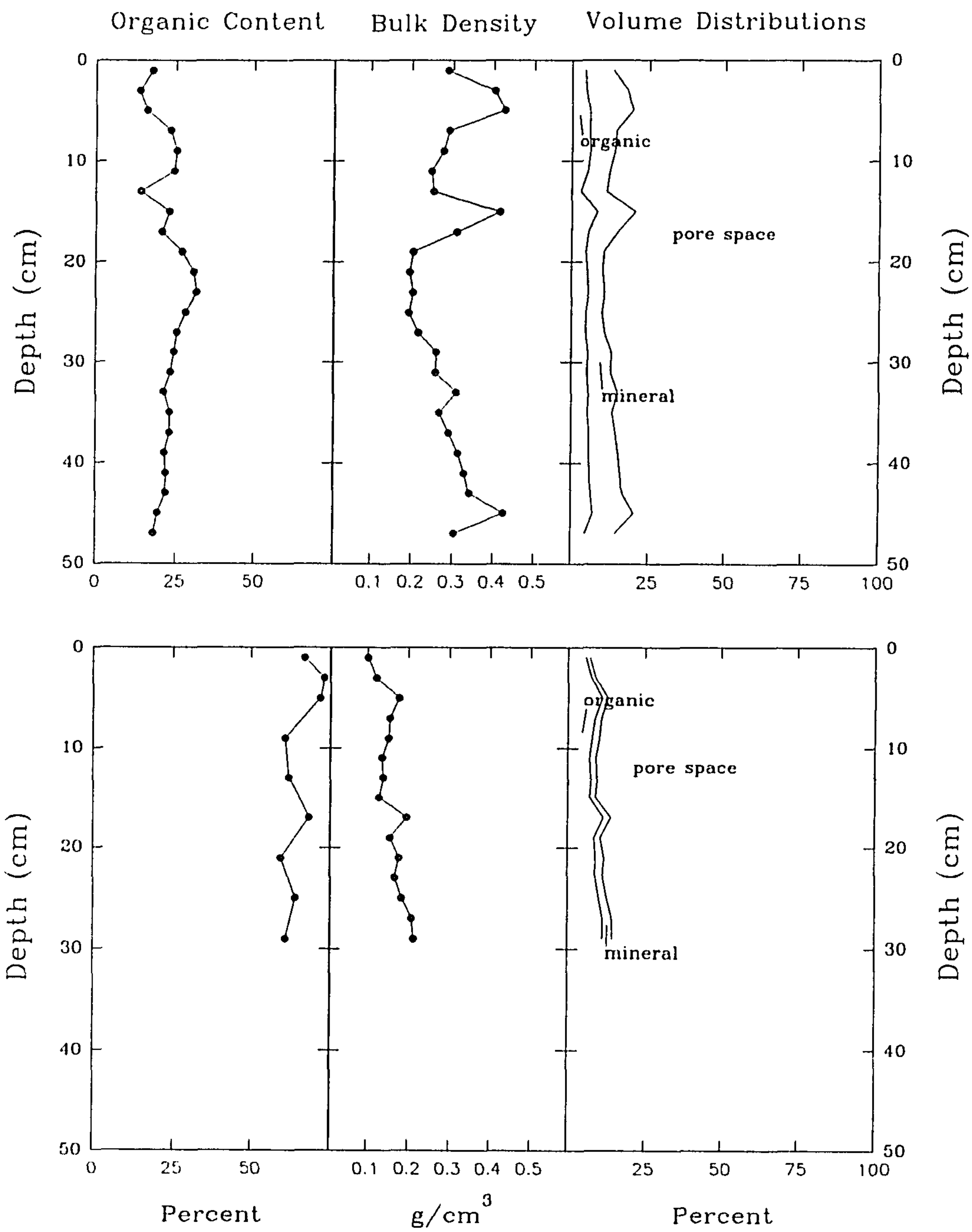

Figure 1.3b. Profiles of organic content, bulk density, and sediment volume distributions for cores from the low marsh at Biloxi Bay (top) and from the black mangrove station at North Key Largo (bottom).

the marsh (Aransas NWR, San Bernard NWR, and Biloxi Bay), accretion rates decreased from the low to high station (Figure 1.4). There were significant differences between high, mid, and low marshes ($p = 0.001$; Duncan's post ANOVA test for differences between stations, $p = 0.05$). Using average rates from the transects only, rates of vertical accretion were highest at Biloxi Bay and lowest at Aransas NWR (Table 2).

Vertical accretion rates from the incomplete transects, and individual rates for cores from the mangrove sites are given in Table 3. The three samples from the incomplete transects are similar to those found along the complete transects, with the exception of the high rates found at the high station on Biloxi Bay transect #1. This was probably due to the fact that this sample, although being at the upper end of the marsh, was also very near to an adjacent tidal creek that may have delivered material and drained this area, causing it to function as a low marsh station, rather than a high marsh station. This exception indicates some of the variability that is typical across any wetland and which makes assessing rates for a given area very difficult.

A similar evaluation of rates across the wetland was not possible for the mangrove samples since samples were not collected along transects; however, average rates were higher for the red mangrove stations (0.40 cm/yr vs. 0.27 cm/yr for black mangrove stations). This agrees with the trend found along the salt marsh transects because red mangroves grow along the lowest edge of the wetland, with black mangroves in the interior of the mangrove swamp (Odum et al. 1982). These rates of

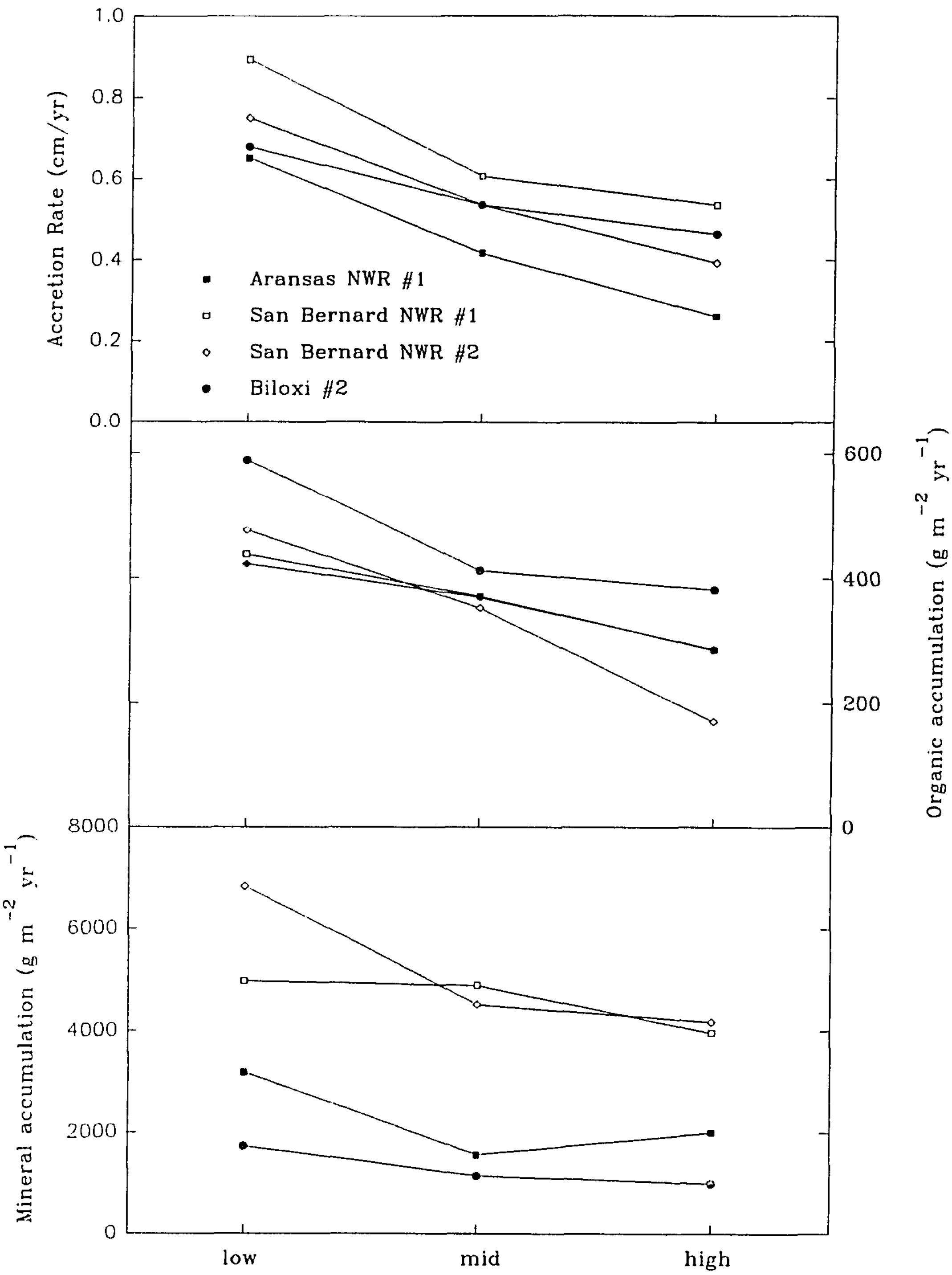

Figure 1.4. Rates of vertical accretion (top), organic accumulation (middle), and mineral accumulation (bottom) from transects across the marsh. Data are for complete transects from salt marsh sites only.

Table 1.2. Average vertical rates of accretion. Standard deviations are given in parantheses. Rates of relative sea level rise are from Penland and Ramsey (1990).

	Vertical Accretion (cm/yr)		Relative Sea Level Rise (cm/yr)	NET ACCRETION (cm/yr)
Aransas NWR	0.44	(0.16)	0.31	0.13
San Bernard NWR	0.62	(0.15)	0.63	-0.01
Biloxi	0.56	(0.09)	0.15	0.41
Florida Keys				
Red Mangroves	0.40	(0.01)	0.22	0.18
Black Mangroves	0.27	(0.11)	0.22	0.05

Table 1.3. Vertical accretion, organic accumulation and mineral accumulation rates for cores from incomplete transects and for mangrove cores from the Florida Keys.

Site and Core Location	**Vertical Accretion** (cm/yr)	**Organic Accumulation** ($g/m^2/yr$)	**Mineral Accumulation** ($g/m^2/yr$)
Aransas #2 high	0.48	368.4	2626.3
Biloxi #1 mid	0.32	205.4	669.5
Biloxi #1 high	0.61	350.3	1304.3
Florida Keys - red mangroves			
core #3	0.42	331.6	143.5
core #6	0.39	304.4	171.5
Florida Keys - black mangroves			
core #2	0.42	361.0	303.7
core #4	0.19	115.4	123.3
core #5	0.19	156.3	64.9

vertical accretion are higher than have been shown for other mangroves swamps in Florida (Lynch et al. 1989).

The gross vertical accretion rates can be compared to rates of relative sea level rise for each site in order to estimate a "net accretion balance" (i.e., changes in the relative elevation of the wetland sediment surface over the last 30 years) which indicates whether or not the wetland is keeping up with current rates of sea level rise or is in danger of submergence. Data for relative sea level rise are from tide gauge analysis by Penland and Ramsey (1990). Three of the four sites have vertical accretion rates greater than recent rates of relative sea level rise (Table 2). San Bernard NWR is the only site with a negative accretion balance. In fact, the rate of relative sea level rise that was used for this site may be too large, because the only nearby tide gauge station that was available was from Galveston, where rates of subsidence are relatively high and are probably higher than at San Bernard NWR.

Mass based accumulation rates

Organic matter accumulation declined consistently across the marsh ($p = 0.0027$), with significant differences between all stations based on Duncan's post ANOVA test with $p = 0.05$ (Figure 1.4). Mineral matter accumulation rates also declined across the marsh ($p = 0.032$); however, significant differences were only found between the low marsh station and the other stations using the same Duncan test (Figure 1.4).

Average organic accumulation rates at the 5 sites ranged from 210 to 380 $g/m^2/yr$, with a much larger variation in average rates of mineral matter

accumulation (Table 4). San Bernard NWR and Aransas NWR had the highest rates of mineral matter accumulation; whereas rates from the mangroves in the Florida Keys were extremely low (Table 4). The rates of mineral matter accumulation from San Bernard NWR and Aransas NWR are much greater than other published values for most coastal salt marshes (need some citations).

DISCUSSION

Differences along the transects

Previous authors have reported decreases in accretion rates with increasing elevation within the marsh, or with increasing distance from a tidal channel (Richard 1978, Pethick 1981, Oenema and DeLaune 1988). Pethick (1981) stated that the frequency and duration of inundation were most important in affecting rates of accretion across the marsh. Letzsch (1983) used marker horizons in a study of six different "stations" in a Georgia marsh, over a seven year time period and found consistently lower accretion rates from the upper part of the marsh. Stoddart et al. (1989) evaluated changes with distance from tidal channels at a macrotidal site along the Norfolk coast in Britain. It was assumed that the channels were the main source of mineral matter for the marsh, and they found lower accretion rates with increasing distance from the channel.

Our data with ^{137}Cs dating confirm these trends over a 30 year time scale. However the results of the mineral matter accumulation estimates indicate that the cause may not be simply a decrease in the delivery of mineral sediments to the upper

Table 1.4. Average rates of organic and mineral accumulation. Standard deviations are given in parentheses.

	Organic Accumulation Rate ($g/m^2/yr$)		Mineral Accumulation Rate ($g/m^2/yr$)	
Aransas NWR	333.4	(125.67)	2248.4	(686.97)
San Bernard NWR	413.0	(91.71)	4895.2	(937.84)
Biloxi	359.6	(56.17)	1288.1	(318.86)
Florida Keys				
Red Mangrove	318.0	(13.62)	157.5	(13.97)
Black Mangrove	210.9	(107.46)	164.0	(101.64)

marsh, due to the shorter duration of flooding at higher elevation, as has been proposed previously. If this were the case, we would expect a sharp decrease in mineral matter accumulation rates across the marsh. However, our results showed that vertical accretion rates across the marsh decreased significantly, but mineral matter accumulation rates were only significantly greater at the low station (Figure 1.4). It appears that this was due to decreases in organic accumulation rates, which also were significantly different across all three stations, not the delivery of mineral matter to the marsh surface (Figure 1.4).

A "tidal subsidy" is possible at these sites, even if it is not in the form of mineral matter delivery. Increased drainage and above and below-ground plant production at the low end of the marsh may have been the cause for increased rates of organic matter accumulation in these lower stations. Wiegert et al. (1983) have shown a significant increase in above-ground production with increased drainage and soil water movement. Other studies have also documented trends in production across the marsh (DeLaune et al. 1979). The differences in organic matter accumulation could also be due to differences in below-ground decomposition rates across the marsh, although most evidence indicates that decomposition rates would be higher where drainage is better because of increased aerobic decomposition rates.

Tidal range vs. net accretion rates

On a larger scale than the importance of relative elevation and creek proximity within a given marsh, the tidal input of material has been held to be very important because of the relationship between tidal range and accretion rates from different

marshes that has been documented in the past. In a study of five high marsh sites along the Connecticut coast in Long Island Sound, Harrison and Bloom (1977) found that gross accretion rates along the sound increased with increases in mean tidal range. Harrison and Bloom (1977) offered a series of potential explanations for this and indicated that the most probable explanation was the input of mineral material (and not organic matter or the availability of mineral matter at different sites).

This concept was also proposed in a review of accretion processes by Stevenson et al. (1986), using data from 13 sites. They found a significant positive correlation of net accretion rates with tidal range for the 13 non-riverine sites. However, the only low tidal range sites that had been measured at that time were also sites with high rates of local subsidence (Lousiana and Chesapeake Bay) and Stevenson et al. (1986) pointed out that additional data from low tidal range sites were necessary. Our data indicate that this trend is probably not as simple as has been shown in the past (Figure 1.5). The four data points that I added to Stevenson et al.'s data decreased the r^2 of the regression from 0.73 to 0.42, although the regression remained significant ($p = 0.0037$).

Other recent work from two areas on the Atlantic coast also does not support this relationship. Craft et al. (1993) also found positive accretion balances for three of the four microtidal sites that they studied on the North Carolina coast. They measured rates in regularly and irregularly flooded marshes, and found a deficit of 0.1 cm/yr in the regularly flooded backmarsh site, while other areas had positive accretion balances ranging from 0.05 to 0.17 cm/yr (Craft et al. 1993). Additionally,

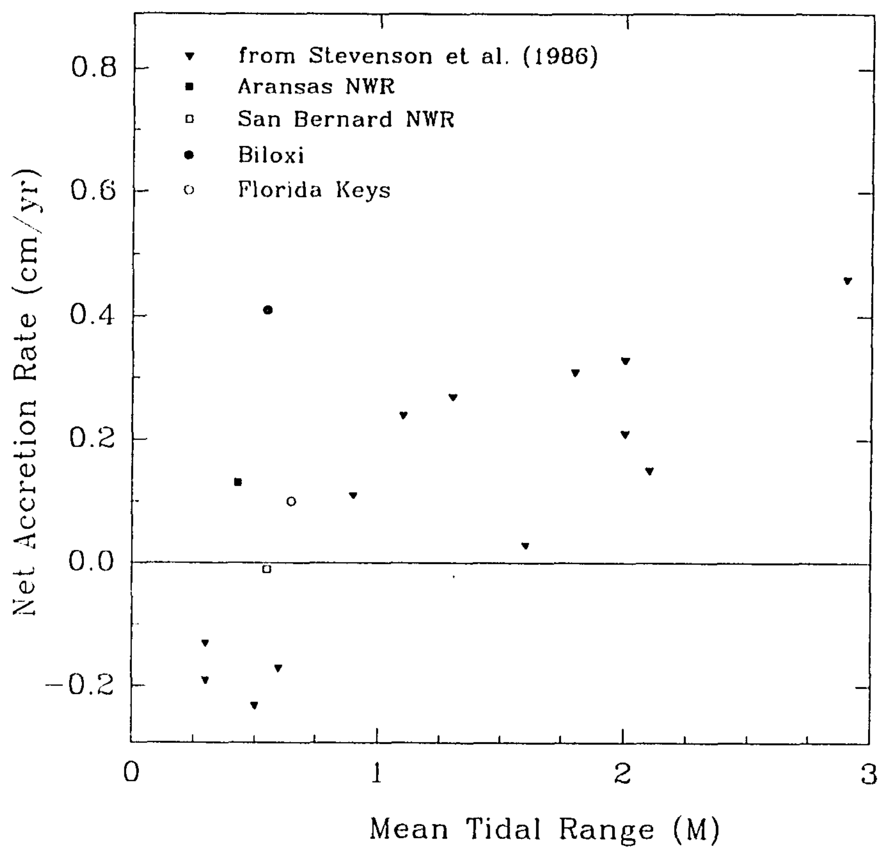

Figure 1.5. Relationship of mean tidal range and net accretion rate for data from the sites sampled and from other coastal wetlands (additional data from Stevenson et al. 1986). Tidal range values for Gulf coast sites are from NOAA (1993).

Wood et al. (1989) studied short term accretion rates at 26 sites in Maine, with tidal ranges varying from 2.5 to 5.5 M, and found no relationship between mean tidal range and accretion balance for these sites alone (see Figure 8 from Wood et al. 1989). Our data and these other studies do not imply that there is no relationship at all between tidal range and accretion processes; however, they do indicate that tidal range alone can not predict the accretion balance of a particular site. Many more variables are probably involved. In order to further investigate this complex issue, I also used a multi-variate approach to analyze potential relationships between variables such as tidal range, sediment organic content, and bulk density with gross and net accretion rates (Chapter 3).

Correlations of vertical accretion, organic and mineral accumulation rates

Given the fact that there appears to be more involved in the vertical accretion rates at these sites than just the delivery of mineral matter, the question arises - what is controlling the accumulation of sediment in coastal wetlands along the northern Gulf of Mexico. Using data from all of our 20 cores, rather than just the transect data, I evaluated the correlation between vertical accretion, mineral accumulation and organic accumulation rates. As other authors (Nyman et al. 1993) have pointed out, it is to be expected that these rates are correlated, since the estimation of the mass based rates uses vertical accretion in their calculation. However, the evaluation of these trends can still shed some light on the overall process of accretion in coastal wetlands.

For the wetlands that I evaluated, vertical accretion is much more highly correlated with organic accumulation rates than mineral accretion rates (Figure 1.6). The r^2 value for vertical accretion and organic accumulation rates was 0.895; whereas, it was 0.458 for vertical accretion and mineral accumulation rates. This reinforces the findings that I found along the transects (and is to be expected since I used some of the same data in this analysis). Nyman et al. (1993) also found a similar relationship for coastal wetlands in Louisiana . The correlation that I found for vertical accretion and organic accumulation rates is very similar to the relationship that Nyman et al. (1993) found and indicates that there may be a similar relationship for marsh accretion across the northern Gulf coast region. Other researchers have also looked at the relationship of organic matter and mineral matter accumulation rates in hopes of interpreting sedimentation processes (McCaffrey and Thomson 1980; Bricker-Urso et al. 1989). Our data adds further support to the hypothesis that organic matter is the key to the development of marsh sediment (McCaffrey and Thomson 1980, Nyman et al. 1993).

In order to evaluate processes on a regional scale, it is very instructive to evaluate the relationship between organic and mineral matter accumulation rates, based on both individual cores and as average values for each site (Figure 1.7). I have discussed above the difficulties in assigning an average value for a given site, however I feel that the use of the transect approach is the best alternative. Most of the variability in the rates of organic matter accumulation is due to changes across the transects (or within a given mangrove area), and that when data are averaged for

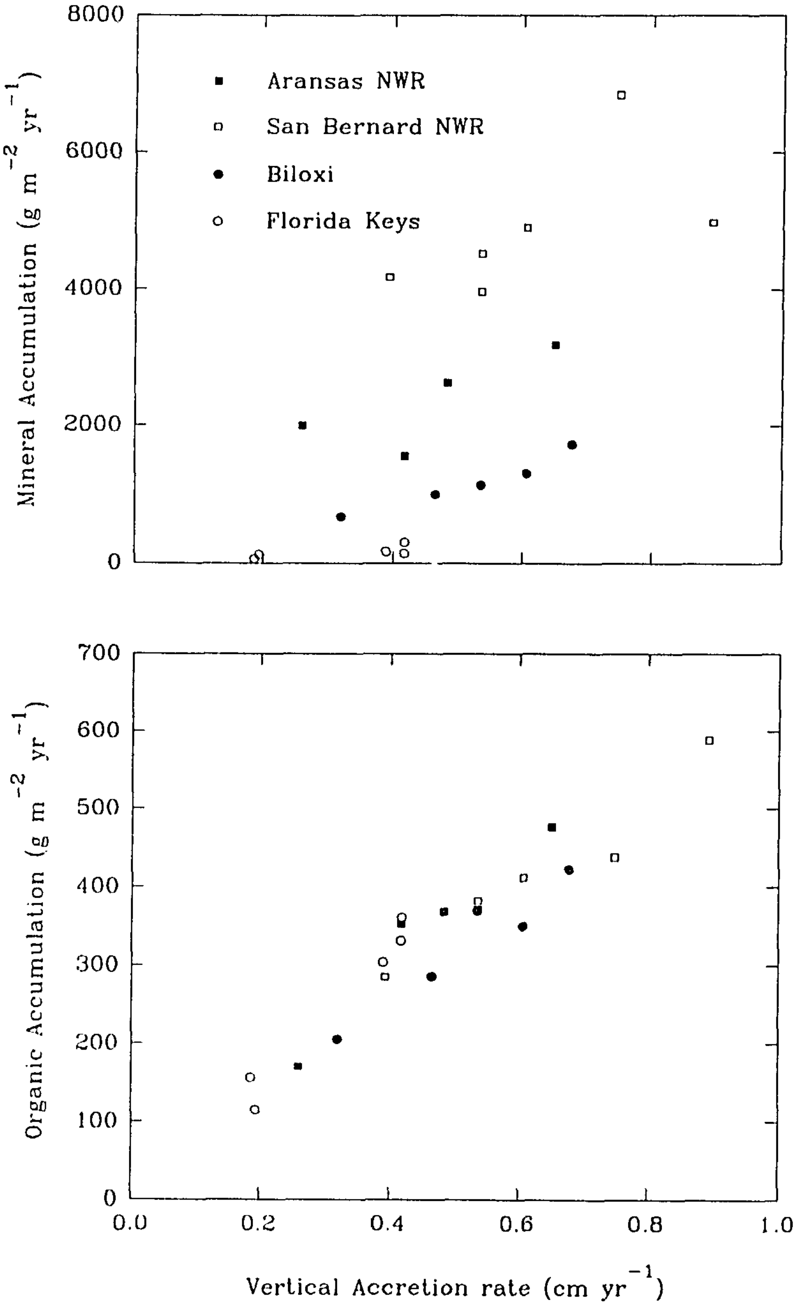

Figure 1.6. Correlation of vertical accretion rates with mineral accumulation rates (top) and organic accumulation rates (bottom) for all cores.

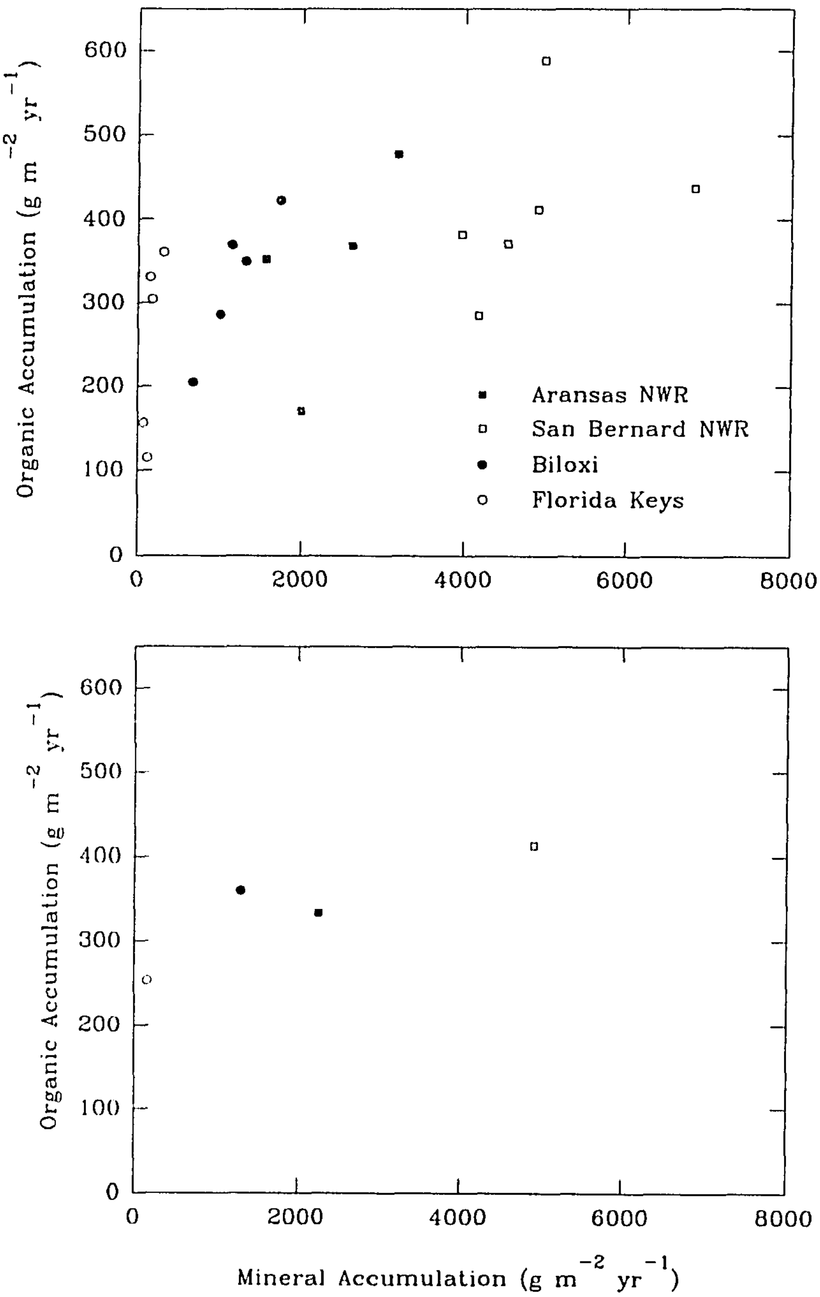

Figure 1.7. Correlation of mineral and organic accumulation rates for all cores (top) and using average values for each site (bottom).

individual sites, average organic accumulation rates from all of the sites are very similar (Table 4). Bricker-Urso et al. (1989) have suggested that there may be a limit to the organic accumulation rates in coastal marshes (see their Figure 12) based on their own data and data from other sites along the Atlantic and Gulf coasts. The average data from our sites also shows that there is a very flat relationship between mineral matter and organic matter accumulation rates. At our sites, average mineral matter accumulation rates varied from approx. 160 to 5000 $g/m^2/yr$, while organic matter accumulation rates varied from approx. 200 to 400 $g/m^2/yr$. If in fact there were a "limit" to the maximum rates of organic matter accumulation, this could have importance consequences for the ability of marshes to compensate for future increases in sea level rise.

The fate of these wetlands in light of sea level rise

There does not appear to be any relationship between subsidence rates (relative sea level rise rates) and accretion rates. The r^2 for the 20 cores is 0.19 ($p = 0.055$). Although it has been suggested that there is a positive relationship between relative sea level rise and accretion (Redfield 1972), the lack of a statistical relationship has been shown by other researchers (Stevenson et al. 1986, Wood et al. 1989).

In addition, these coastal wetlands do not appear to be in danger of flooding in the near future (Figure 1.8). The one site that may be in trouble is San Bernard NWR. Interestingly this site has the highest rates of mineral matter accumulation. Also, subsidence rate here may be too high, as mentioned earlier. In comparison to the sites that have been studied in Louisiana, these Gulf sites are behaving very

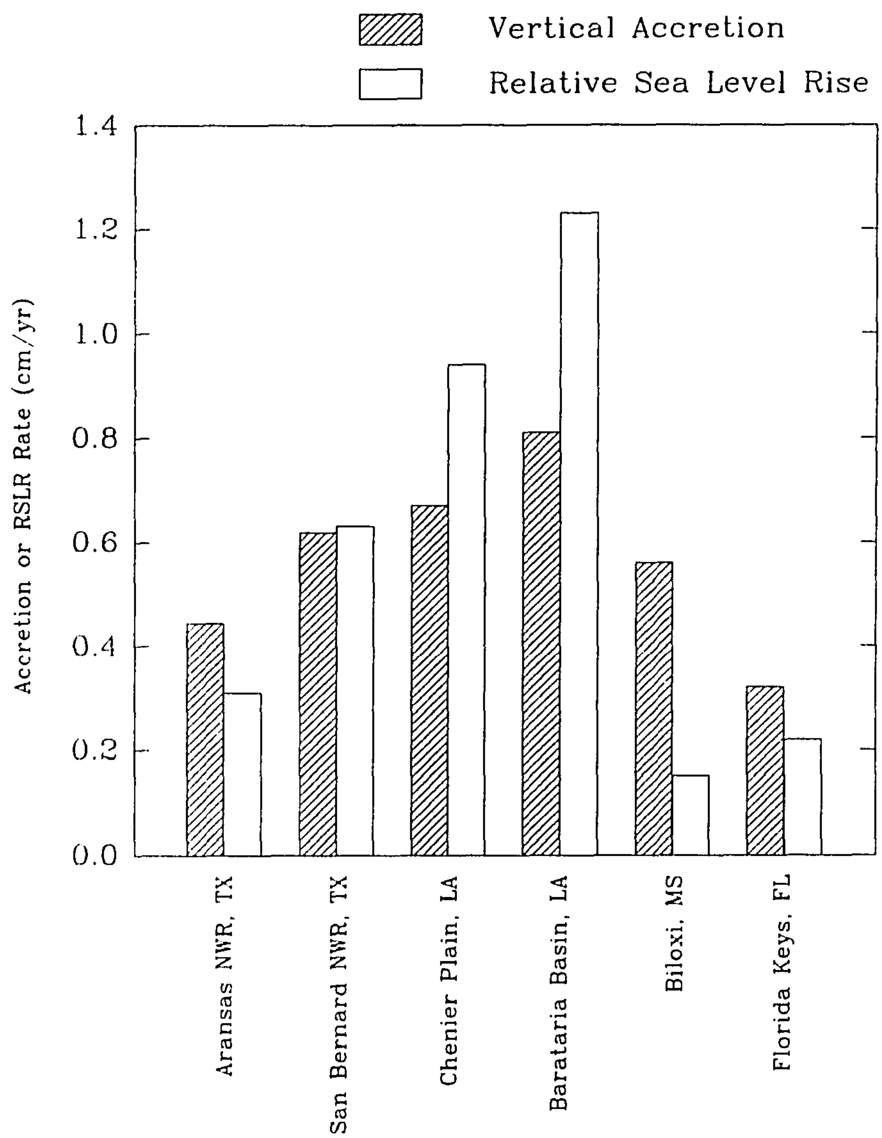

Figure 1.8. Vertical accretion rates and rates of local sea level rise (from Penland and Ramsey 1990) for our sites from the Gulf of Mexico, as well as two sites in Louisiana (Data from DeLaune et al. 1983 and Hatton et al. 1983).

differently (Figure 1.8). Several studies (DeLaune et al. 1983, Salinas et al. 1986, Baumann et al. 1984) have documented the relationship between negative accretion balances and the loss of coastal marsh acreage in coastal Louisiana, but this does not appear to be the case in most other areas in the northern Gulf of Mexico. There is a Gulf wide trend in subsidence rates with higher rates near the influence of the Mississippi River delta (Penland and Ramsey 1990), but sites away from this delta are currently not in danger of submergence.

Turner (1991) has indicated some of the problems with using tide gauge data for estimating rates of relative sea level rise for marsh accretion studies. In particular, Turner (1991) noted that rates of subsidence are probably not the same in tidal channels, where tide gauges are usually installed, as they are in coastal wetlands. Because of the processes, such as decomposition and compaction, which occur within the sediment column in coastal wetlands (Chapter 4), it is likely that rates of sea level rise could be greater in the wetland. If this is the case, positive values for accretion balance may not necessarily mean that the relative elevation of the marsh is increasing. Rather, if the differences between processes at the tide gauge and within the wetland (decomposition and compaction) are also considered, slightly positive values of accretion balance may indicate that the wetland is simply keeping pace with measured increases in relative sea level rise.

Besides actual land loss, one potential impact of an increase in sea level rise that could be very important is the conversion of upper marsh to low marsh, due to lower vertical accretion rates in the upper marsh (Figure 1.4). This type of a change

in vegetation communities has been documented in New England tidal marshes over the past 50 years, and through peat core analysis (Warren and Niering 1993). There could be a similar change in coastal wetland plant communities along the Gulf coast, and this loss of upper marsh habitat could have important consequences for wildlife and other marsh functions.

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CHAPTER 2

SEDIMENT ACCRETION RATES FROM SELECTED NORTHERN EUROPEAN COASTAL WETLANDS USING ^{137}Cs AND ^{210}Pb PROFILES

INTRODUCTION

The long term stability of a coastal wetland depends on its elevation relative to mean sea level (Redfield 1972, Orson et al. 1985). This relative elevation is increased by the accumulation of mineral and organic sediments and decreased by eustatic sea level rise (Gornitz 1982) and subsidence (DeLaune et al. 1983). Mineral and organic material is trapped on the surface of the marsh (Gleason 1979, Stumpf 1983), and organic material also accumulates below ground from root and rhizome production (Bloom 1964, McCaffrey and Thomson 1980, Oenema and DeLaune 1988). Some of the organic material that accumulates in the sediment column is lost over time by decomposition. In addition, compaction can change the volume occupied by mineral and organic sediments and contribute to overall subsidence rates (Bloom 1964). All of these factors are very important in the dynamics of surface sediments of coastal wetlands, and without the accumulation of new material, coastal wetlands would be inundated and converted to intertidal mudflats or subtidal areas.

Potential increases in global temperatures will cause increases in eustatic sea level that may greatly increase the current rate of flooding in coastal wetlands (Gornitz 1982, Woodroffe 1993). Because the only way that these wetlands will be able to compensate for changes in sea level is through the accretion of sediments,

determining current rates of deposition is extremely important in predicting the future survival of marshes under a scenario of increased sea level rise. Additionally, if management decisions are made that will affect the stability of coastal wetlands, these decisions should be made with the most complete information available (Titus 1986). By systematically evaluating changes in marshes over time, we can compile some of this necessary information. I studied a series of coastal wetlands in Northern Europe (Figure 2.1) in order to document accretion rates in these areas and to evaluate changes in organic content and sediment bulk density due to decomposition and compaction. These sites are of particular interest because of the potential use of a new ^{137}Cs sediment marker from the 1986 Chernobyl disaster. I chose to look at a large number of sites in order to address the usefulness of the spike of ^{137}Cs from the Chernobyl nuclear accident in April, 1986. The plume of radioactive fallout was patchy from this incident because it did not get into the upper atmosphere and came down most intensely with local rain (Cambray et al. 1987). Because of the patchy nature of this fallout, it seemed unlikely that the Chernobyl spike would be found in cores from all northern European locations. Recent cores that have been collected in the U.S. have not found evidence of this peak.

METHODS

Sampling locations

a. **Stiffkey Marsh, UK** - This site was located in a nature preserve just northwest of the town of Stiffkey. This area is influenced by storm surges from the

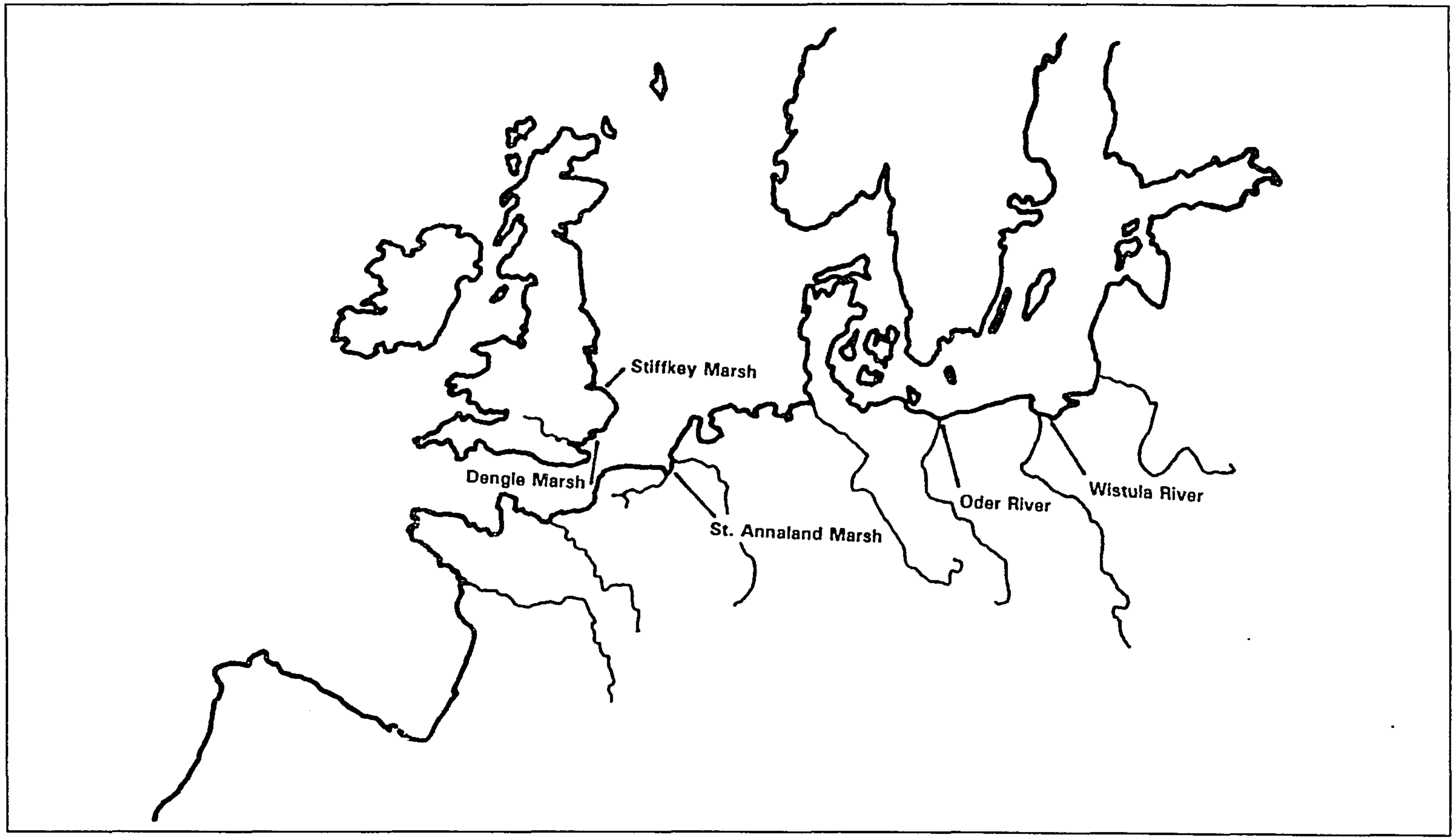

Figure 2.1. Location of sampling sites along northern European coastline.

North Sea and has a mean spring tidal range of 6.4 M, with a mean neap tidal range of 3.4 M. This marsh is very similar to nearby Scolt Head Island where extensive studies of marsh sedimentation rates have been completed by Steers (1938), Stoddart et al. (1989) and French and Spencer (1993). The sample area from the low marsh at Stiffkey was vegetated by *Spartina anglica*, and the high marsh was mostly *Armeria maritima*, *Limonium vulgare*, and *Triglochin maritima.*

b. **Dengie Marsh, UK** - Samples were collected from the central section of Dengie Marsh. A 0.25 km wide swath of marsh has developed here, just seaward of a large flood protection levee. This area is open to the North Sea and is potentially impacted by large storm surges. The site is between the Rivers Blackwater (to the north) and Crouch (to the south) and is also influenced by the plume of the River Thames, which flows North along this part of the English coast. The low marsh sample here was taken in an area dominated by *Halimione portulacoides.* A small area of marsh below this was colonized by *Salicornia sp.* and *Spartina anglica.* It appeared that this area was newly vegetated and would not be suitable for either ^{137}Cs or ^{210}Pb dating. High marsh sites were also vegetated with *Halimione portulacoides.*

c. **St. Annaland Marsh (Eastern Schelde), Netherlands** - The site was located just east of the town of St. Annaland in a small marsh of approximately 1.7 km^2. The area has a strong tidal influence, although the tidal range within the entire Eastern Schelde estuary has been reduced due to a series of large scale projects that were implemented for flood control in this part of the Netherlands (Smaal and Nienhuis 1992). The main impact of the projects on this site was from the

construction of a moveable storm surge protection barrier in 1986 which allows muted tidal influence to the area. This area is affected by fresh water input from the Rhine River, but fresh water input has decreased in the last few decades due to flood control projects in the area (Heip 1989, Smaal and Nienhuis 1992). Fresh water inputs from the Schelde have been limited to the Western Schelde since the 1860's. The low marsh site here is dominated by *Spartina anglica*, and the high marsh site is dominated by *Halimione portulacoides* and *Puccinelia maritima*.

d. **Oder River, Poland** - The site was located within Szczeciń Bay, approximately 8 km southeast of the town of Świnoujście. As with the site from the Wistula River described below, the area has a very low tidal range and low water salinities. This area is at the north end of Szczecń Bay (Zalew Szczeciński), along the main channel of the Old Świna river, a distributary of the Oder River. Cores were collected from the edge of a small island (Wispa Karsiborska Kępa) which is bordered by levees. The center of the island is used for cattle grazing, as are many other small islands within the Bay. The marsh here is a monoculture of *Phragmites communis*.

e. **Wistula River, Poland** - This site was located in the eastern part of the historic delta of the Wistula River. No natural marshes exist at the current mouth of the Wistula River, which is in a highly impacted area within the city of Gdansk. The cores were collected from an area of the delta that has been diked off from river influence and is now primarily used for farming. At the eastern edge of this area, the former delta is influenced by the Baltic via the Wistula Lagoon (Wisłany Zalew).

The tidal range at this site and at the other site from the Baltic is extremely low due to the limited influence of the North Sea tides on the Baltic. In addition, water salinities in the Baltic are very low. Because of these conditions, coastal fringe marshes in the Baltic are dominated by *Phragmites communis* rather than more saline tolerant species, such as are found in coastal wetlands bordering the North Sea. Cores were collected from a small marsh dominated by *Phragmites* and bordering the Cieplicówka branch of the Nogat (a distributary of the Wistula), near the small village of Cieplice.

Core collection and analyses

Two cores were collected for analysis at each site, one from the low marsh and one from a mid to high marsh area. I collected samples from low sites that were at least 10 m away from the lowest edge of the marsh (avoiding a stream-side effect) and were located in vegetation typical of the "low marsh" of that particular area, usually either *Spartina anglica* or *Phragmites*. High marsh sites were located in the same marsh but at higher elevations. Because of time and shipping constraints, it was not possible to collect multiple cores at a given sampling area.

Cores were collected in 15 cm diam. aluminum cylinders to a depth of 50 cm. Cores were extruded with a wooden plunger and were sectioned every 1 cm from the top 10 cm and every 2 cm from 10 to 50 cm. Compaction was minimal and was estimated in the field by measuring the depth to the sediment surface from the top of the coring tube, inside and outside of the core. Measurements were also made of the length of the sediment core before and after extruding in order to measure any

compaction due to extruding the sample from the coring tube. One cm sections were used in the top of the cores in order to give better precision in locating the Chernobyl peak. Individual sections were placed into ziplock bags, weighed wet, dried at 80°C, weighed dry, and crushed with a mortar and pestle. ^{137}Cs activity of the bulk sediment was counted with a Lithium Drifted Geranium detector and multi-channel analyzer.

^{137}Cs is a product of nuclear weapons testing and does not occur naturally. Significant levels of this isotope first appeared in the atmosphere in the early 1950s with the peak quantities detected in 1963. Sediment profiles are dated based on the 1963 peak, and average sedimentation rates are obtained for the period from 1963-present. Additionally, a significant amount of ^{137}Cs was released from the April 1986, accident at the Chernobyl nuclear power plant in the Ukraine and may be useful as a sediment marker. The ^{137}Cs was part of a large cloud of radioactive material that was dispersed across much of Europe. The plume initially moved in a N/NW direction over the former Soviet Union, northeastern Poland, and Scandinavia. The cloud then moved to the S/SW and travelled over much of Europe (Islam and Lindgren 1986).

Five of the cores were used for ^{210}Pb analysis. I only evaluated five cores, because the other cores were too short to used for ^{210}Pb analysis, based on the accretion rates obtained from ^{137}Cs. ^{210}Po was measured by alpha spectrometry, assuming secular equilibrium with ^{210}Pb (Flynn 1968). A 1 g subsample was taken from sections at 6 cm intervals. Organic matter was removed using concentrated

hydrogen peroxide and heat, and the samples were then digested with HF, HNO_3, and HCl, following the method given in Flynn (1968). ^{209}Po was used as a tracer to estimate the efficiency of the digestion and plating process. Before plating, the sample was treated with hydroxylamine hydrochloride and sodium citrate, and the pH was adjusted between 1 and 2 with ammonium hydroxide. Samples were plated onto silver disks at room temperature for 7 days. After plating, the disks were heated to near boiling in concentrated HCl for one hour to remove iron deposits that interfere with ^{210}Po and ^{209}Po alpha emissions (Benoit and Hemond 1988). Samples were counted for one to three days (depending on ^{210}Po activity) using a surface-barrier detector and a multi-channel analyzer. Supported ^{210}Pb activity was based on average measurements of ^{210}Pb activity in the lowest samples from each core, where ^{210}Pb activity became constant. Vertical accretion rates were estimated from the excess (unsupported) ^{210}Pb profiles for each core, using the constant initial concentration method (Goldberg et al. 1977). Linear regression analysis was used to solve for (λ/s) in the log transformed equation for radioactive decay:

$$\ln A_x = \ln A_0 - (\lambda/s) * x$$

where: x = depth of section (cm)

A_x = excess activity of ^{210}Pb at depth x (dpm/g)

A_0 = excess activity of ^{210}Pb at surface (intercept in the regression)

λ = decay constant for ^{210}Pb = 0.03114/year

s = vertical accretion rate (cm/yr).

The slope of the regression in the ^{210}Pb profiles is equal to -(λ/s), and more negative slopes indicate smaller vertical accretion rates.

In addition to the ^{137}Cs and ^{210}Pb analyses, 1 g subsamples were taken from each section for combustion at 400°C in order to determine organic content (loss-on-ignition). Using the organic and mineral content of the soil and the bulk density, linear-based accretion rates were converted to mass based rates and divided into organic and mineral components. Profiles of organic content and sediment bulk density were constructed in order to evaluate changes in these parameters with depth. These profiles were also converted to volume distribution profiles, using assumed specific gravity conversions for organic matter and mineral matter of 1.14 and 2.61 g/cm^3 respectively.

RESULTS AND DISCUSSION

Chernobyl ^{137}Cs peak

The peak from Chernobyl input was found in 7 of the 10 cores (Table 2.1). Chernobyl peaks were found in all of the cores from the Wistula and Oder River, as well as in the cores from St. Annaland and one core from Stiffkey (Figures 2.2a, 2.2b, 2.2c). In most cases the original intensity of this peak (considering corrections for decay since the ^{137}Cs was deposited), was about 80% of the original intensity of the 1963 peak (Table 2.1). Peaks from 1986 were sharper than those from the 1963 peaks. In the core from the high marsh at the Wistula River, the corrected intensity from the Chernobyl peak was stronger than the 1963 peak. Additionally, at the low

Table 2.1. Relative intensity of the ^{137}Cs peaks from 1963 and 1986. Peak intensites were corrected for decay to original intensities.

		1963 peak (pCi/g)	1986 peak (pCi/g)	ratio 86/63
ENGLAND				
Stiffkey Marsh	low marsh	8.31		
	high marsh	1.86	1.76	0.95
Dengie Marsh	low marsh	2.08		
	high marsh	2.35		
NETHERLANDS				
St. Annaland	low marsh	1.77	1.73	0.98
	high marsh	1.80	0.97	0.54
POLAND				
Oder River	low marsh	6.69	5.52	0.83
	high marsh	6.34	5.44	0.86
Wistula River	low marsh		15.47	
	high marsh	3.88	7.37	1.90

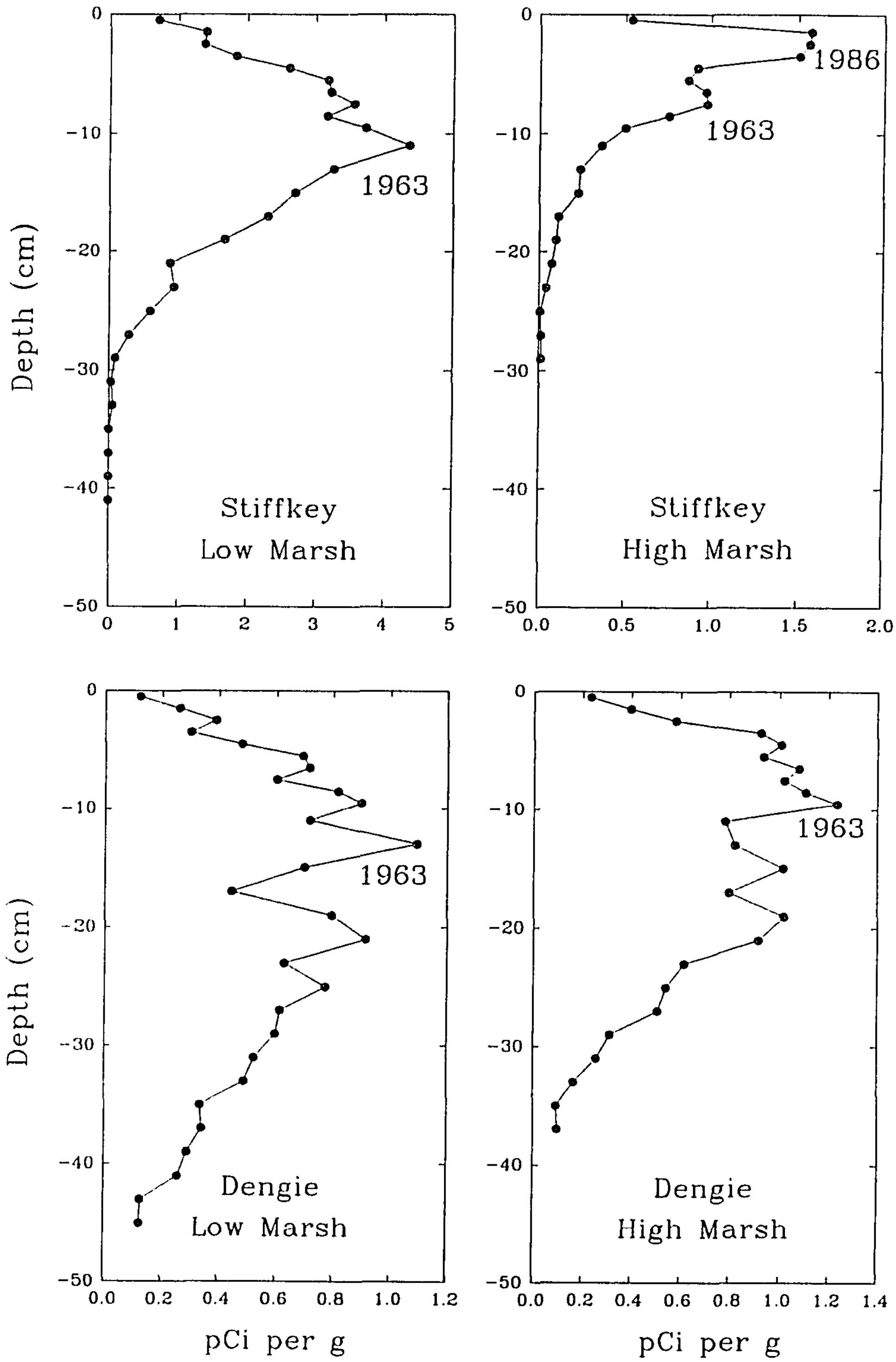

Figure 2.2a. ^{137}Cs profiles for cores from Stiffkey and Dengie Marshes, UK.

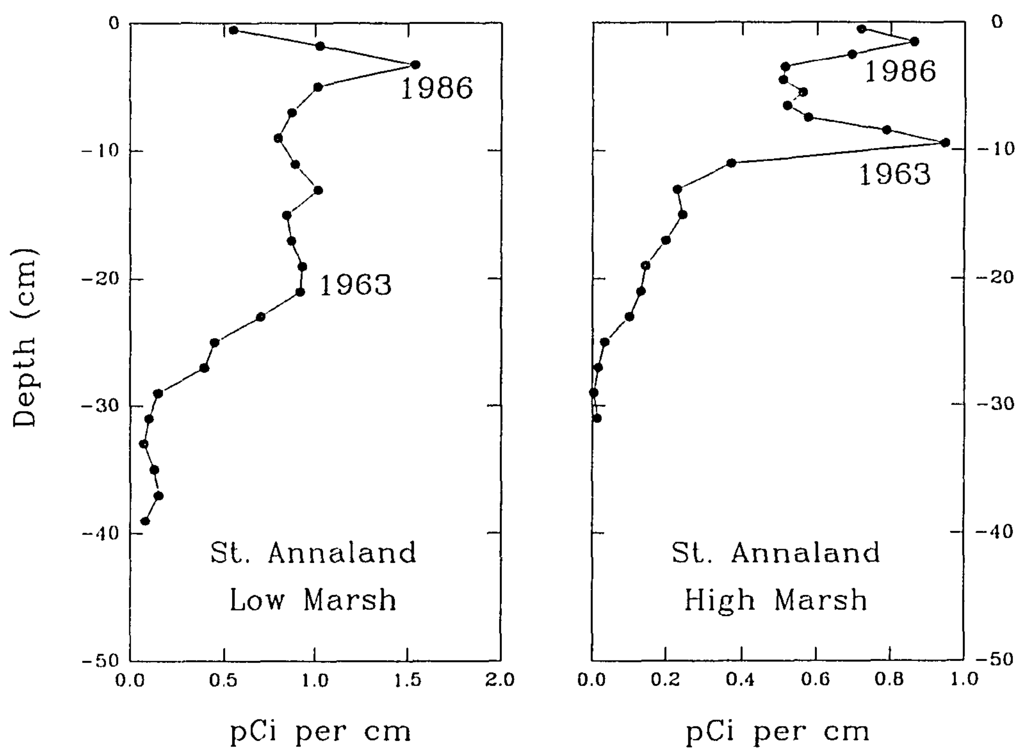

Figure 2.2b. ^{137}Cs profiles for cores from St. Annaland, the Netherlands.

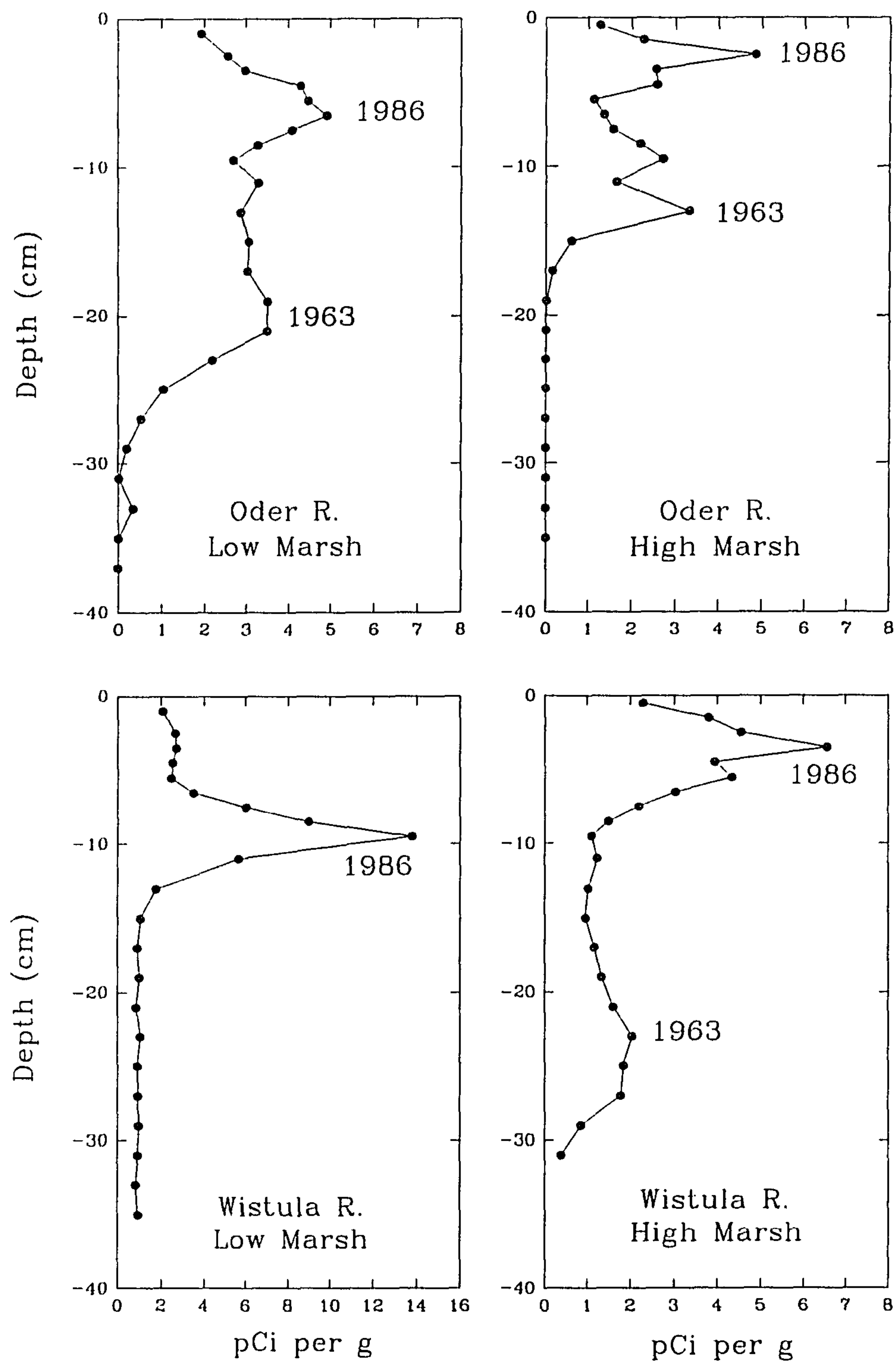

Figure 2.2c. ^{137}Cs profiles for cores from Oder River and Wistula River, Poland.

marsh site from the Wistula River, no 1963 peak was found for comparison; however, the intensity of the Chernobyl peak was extremely high in this core. The highest levels of ^{137}Cs activity from the Chernobyl peak were found in samples from the Wistula River, with lower levels to the west, as would be expected based on levels of fallout immediately after the accident (Table 2.1).

Extensive documentation and modelling of the movement of the cloud of radioactive isotopes over Europe have indicated that the fallout of ^{137}Cs was very patchy. Within Switzerland alone, there was an order of magnitude variation in fallout (Dominik and Span 1992). The plume moved over parts of Eastern Europe and Scandinavia before crossing Switzerland, Italy, Germany, France, Netherlands and England (ApSimon and Wilson 1986, Islam and Lindgren 1986). Deposition rates were highest in areas where the passage of the plume coincided with local rainfall, and atmospheric levels of ^{137}Cs in Europe remained at about four times peak 1963 levels for two to three months (Cambray et al. 1987). Recent studies of ^{137}Cs profiles in sediments from other areas in Europe have documented the peak in salt marshes from the Western Schelde (Zwolsman et al. 1993) and Denmark (Ehlers et al. 1993), and in lake sediments from a Rhine River catchment basin (Beurskens et al. 1993) and Switzerland (Dominik and Span 1992).

Based on the results from these cores, it is clear that easily detectable levels of ^{137}Cs were deposited at the most of sites that I studied. Because of the patchiness of the fallout, it is not clear exactly where else this marker will be found in northern Europe. However, the studies cited above indicate that it could be useful in many

areas in Europe. Given the intensity of the deposits and the half life of ^{137}Cs (30 years), this peak will serve as a valuable sediment marker and will be useable for sedimentation studies for at least 50-60 years. It will be especially useful for short term sediment studies in the near future. Additionally, by using repeated sampling in the same area, this marker along with the 1963 marker, could also be used to identify a distinct section of the sediment column and to document changes in sediment characteristics as it is buried. By following changes to this section, the relative importance of decomposition and compaction could be evaluated. This type of information would be extremely useful for predicting long term sediment processes, as is outlined in my simulation model (Chapter 5).

Sediment accretion rates

As discussed below, there were large differences in accretion rates and sediment characteristics (organic content and bulk density) between the samples from the Baltic Sea wetlands in Poland and those from the three sites in western Europe. The Polish samples were more similar to fresh water marshes, where very high rates of organic matter accumulation are found. Because of these large differences, I will discuss the two groups separately.

Western European sites

At all of the Western European sites, accretion rates based on ^{137}Cs peaks were higher in cores collected from the low marsh (Table 2.2), as expected based on differences in elevation and distance from the sediment source (Richard 1978, Hatton et al. 1983). Rates from the high marsh were 50, 76, and 69 percent of low marsh

Table 2.2. Vertical accretion rates based on ^{137}Cs peaks and ^{210}Pb dating for the ten cores from northern Europe.

		^{137}Cs Peaks 1986	^{137}Cs Peaks 1963 (cm/yr)	^{210}Pb
ENGLAND				
Stiffkey Marsh	low marsh		0.39	1.28
	high marsh	0.30	0.27	0.19
Dengie Marsh	low marsh		0.46	
	high marsh		0.34	not useable
NETHERLANDS				
St. Annaland	low marsh	0.65	0.68	
	high marsh	0.30	0.34	0.26
POLAND				
Oder River	low marsh	1.30	0.71	
	high marsh	0.50	0.46	0.26
Wistula River	low marsh	1.90		
	high marsh	0.70	0.82	

rates at St. Annaland, Dengie, and Stiffkey, respectively. Overall, vertical accretion rates were very similar at all of the high marsh sites, with rates based on ^{137}Cs dating ranging from 0.27 to 0.34 cm/yr. Rates from the low marsh were more variable, ranging from 0.39 to 0.68 cm/yr. Differences between rates based on the 1963 ^{137}Cs peak and the 1986 peak from Chernobyl were very small and within the range of error, based on the 1 - 2 cm width of each section.

Rates from ^{210}Pb dating from the high marsh at Stiffkey (Figure 2.3a) and St. Annaland (Figure 2.3d) were both less than rates based on the 1963 ^{137}Cs peak (Table 2.2). These differences are probably due to changes in decomposition and compaction. The simulation model of sediment processes (Chapter 4) was calibrated for the core from the high marsh at Stiffkey using ^{137}Cs dating, and organic and bulk density profiles. Using these constraints, the model predicted a 100 year accretion rate of 0.19 cm/yr, exactly the same as was measured using ^{210}Pb dating. This agreement confirms that decomposition and compaction are contributing to a decrease in the long term accretion rates at this site, and similar changes probably caused the difference in rates from the St. Annaland high marsh core.

The ^{210}Pb profile from the low marsh at Stiffkey (Figure 2.3b) indicated a very high rate of accretion compared to the rate based on the 1963 ^{137}Cs peak. It is likely that the low marsh has developed from a mud flat and was previously at a lower elevation. As it was developing as a mud flat and colonized by *Spartina anglica*, it probably had a much greater rate of vertical accretion. Stoddart et al. (1989) reported rates of vertical accretion up to 1.5 cm/yr in low marshes at Scolt Head Island, and it

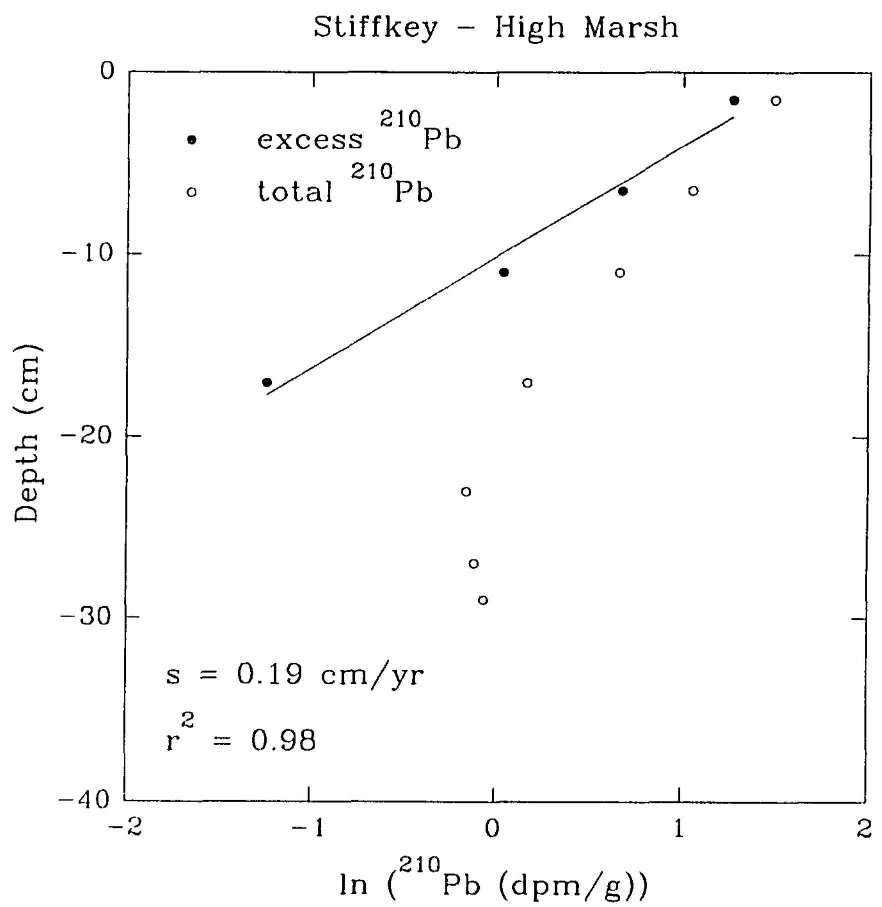

Figure 2.3a. ^{210}Pb profile from the high marsh core, Stiffkey Marsh, UK.

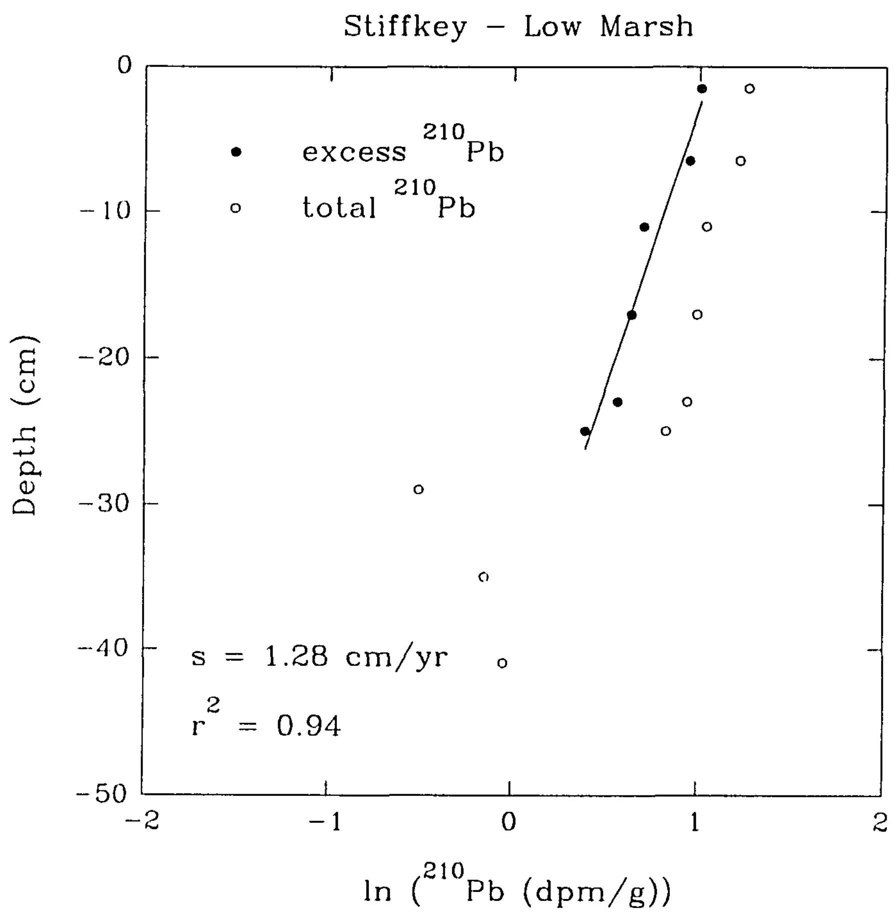

Figure 2.3b. ^{210}Pb profile from the low marsh core, Stiffkey Marsh, UK.

is likely that rates on adjacent mud flats were even greater. Pethick (1981) measured elevation differences in marshes of varying age along the Norfolk coast, and based on this data predicted accretion rates of 1.7 cm/yr for newly forming marshes. The ^{137}Cs peak from 1963 reflects accretion rates when the marsh had reached a higher elevation, and rates decreased. An abrupt change in sediment characteristics at approximately 25 cm also indicates that there was a recent change in the depositional environment (Figure 2.4a). The sediments at depth had high bulk densities and low organic content, which is typical of intertidal mud flats.

The ^{210}Pb profile from the high marsh core at Dengie marsh (Figure 2.3c) did not have the typical log decrease in ^{210}Pb activity with depth, indicating that the core may have been disturbed or had an irregular rate of sediment input. The two ^{137}Cs profiles from this site had distinct peaks (Figure 2.2a), but metal profiles also were unusual from this site (Chapter 5). Reed (1988) evaluated accretion rates in this area using pins to measure changes in the surface of the marsh and found mean accretion rates along transects through the marsh of 1.1 cm/yr near the shoreline and 0.65 cm/yr along the most inland transect.. The maximum accretion rate at a given location over the two year study was 2.1 cm/yr (Reed 1988). However, given these problems with the cores from Dengie marsh, the accretion rates based on ^{137}Cs dating from this site should be taken as tentative results, until confirmation can be made from additional cores from this area. Previous studies of accretion rates in Western European marshes have found similar rates of accretion (Table 2.3).

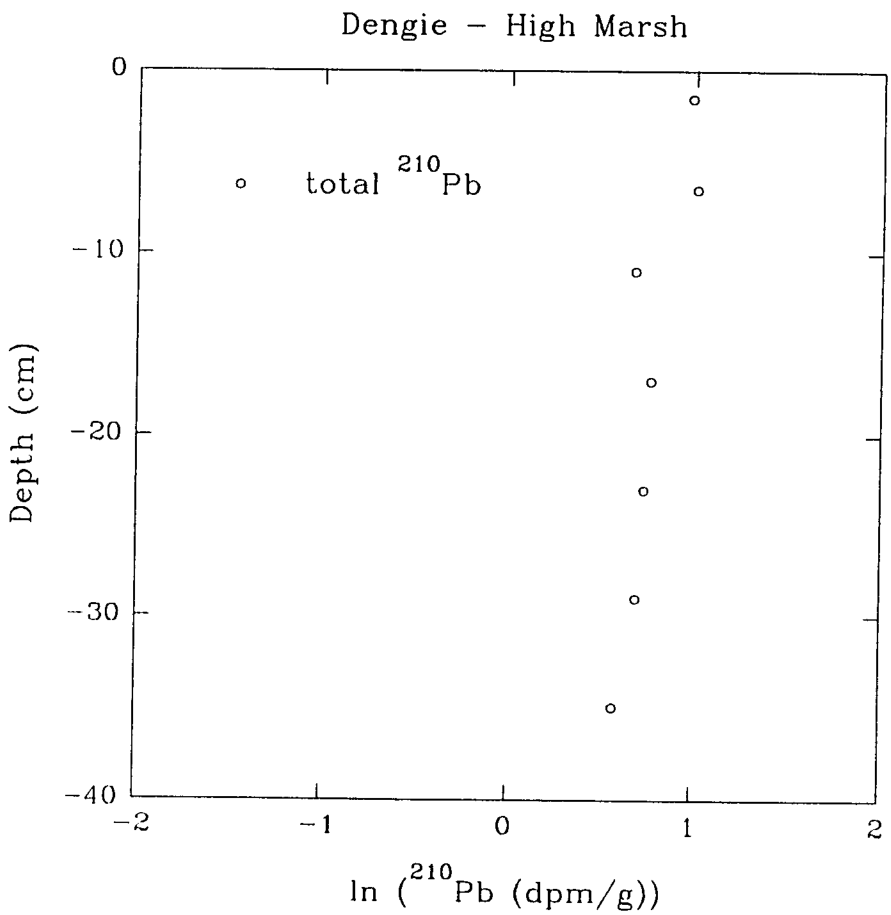

Figure 2.3c. ^{210}Pb profile from the high marsh core, Dengie Marsh, UK.

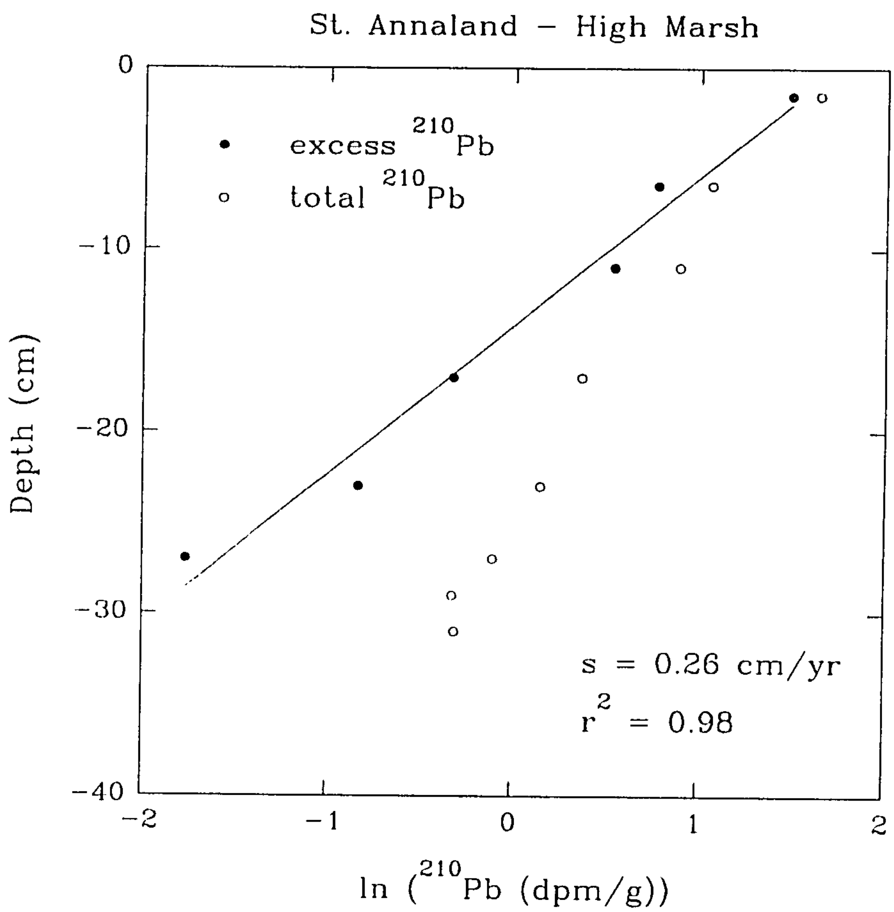

Figure 2.3d. ^{210}Pb profile from the high marsh core, St. Annaland Marsh, the Netherlands.

Table 2.3. Previous studies of accretion rates in coastal wetlands in northern Europe, using ^{137}Cs, ^{210}Pb, or other short term techniques.

Study Site	Range of Accretion Rates	Reference
ENGLAND		
Scolt Head Island	0.1 - 1.2	Steers data from Stoddart et al. 1989
Scolt Head Island	0.1 - 1.4	Stoddart et al. 1989
Scolt Head Island	0.1 - 0.8	French & Spencer 1993
Norfolk Coast, UK	0.0 - 1.7	Pethick 1981
Bridge Creek, Dengie Marsh	0.5 - 1.1	Reed 1988
NETHERLANDS		
Western Schelde	0.9 - 1.7	Zwolsman et al. 1993
Ratekaii	1.0 - 1.5	Oenema & DeLaune 1988
St. Annaland	0.4 - 0.9	Oenema & DeLaune 1988
DENMARK		
Isle of Sylt and Nösse Peninsula	0.8 - 1.2	Ehlers et al. 1993

There were large variations in the mineral matter accumulation rates for the cores from these three sites (Table 2.4). The low marsh cores from Dengie marsh had the highest mineral matter accumulation rates of any of the 10 cores (3200 g/m^2 yr). Mineral matter accumulation rates from the low marsh cores at St. Annaland and Stiffkey were also high. For the high marsh cores, rates again were highest at Dengie marsh (2100 g/m^2 yr). Organic matter accumulation rates were very similar at these three sites, ranging from 320 to 440 g/m^2 yr (Table 2.4). Mineral and organic accumulation rates were both higher in the low marsh than the high marsh samples. The variation in organic accumulation rates was much smaller than for mineral matter accumulation rates. It appears that despite the variability in mineral inputs between the sites, that rates of organic matter accumulation are fairly constant at these marshes. These rates are also similar to values of organic matter accumulation that have been measured from other salt marshes (see Chapter 3 for a review of these rates).

Polish sites

The highest vertical accretion rate from any of the cores was from the sample taken from the low marsh at the Wistula River (1.9 cm/yr). Accretion rates were only determined for this core based on the ^{137}Cs peak from 1986, and there is no peak from 1963. Based on the rate for 1986, a peak from 1963 would be at 53 cm, below the sampling depth of this core. The core from the high marsh at this site also had very high rates of accretion. These rates were the highest of any of the high marsh sites that I sampled. Vertical accretion rates from both the low and high marsh at the

Table 2.4. Mass rates of mineral and organic accumulation for the ten cores from northern Europe.

		Mineral Accum. Rate (g/m^2 yr)	**Organic Accum. Rate** (g/m^2 yr)
ENGLAND			
Stiffkey Marsh	low marsh	1793.0	319.2
	high marsh	839.1	304.1
Dengie Marsh	low marsh	3172.2	425.1
	high marsh	2113.4	347.7
NETHERLANDS			
St. Annaland	low marsh	2347.5	441.5
	high marsh	814.4	322.5
POLAND			
Oder River	high marsh	229.1	369.2
	low marsh	119.9	267.6
Wistula River	low marsh	2077.9*	953.5*
	high marsh	1812.1	635.1

*Rates are based on the 1963 ^{137}Cs peak, except for the core from the low marsh at the Wistula River, which is based on the 1986 peak since no peak was found for 1963.

Oder River were also very high compared to the other samples that were collected. Rates from the low marsh at the Oder River based on the 1986 Chernobyl peak were much higher than rates based on the 1963 ^{137}Cs peak; however, rates based on the two peaks from the high marsh cores were very similar.

The accretion rate from the ^{210}Pb profile for the core from the high marsh at the Oder River (Figure 2.3e) was lower than that based on the 1963 ^{137}Cs peak (Table 2.2). As with the high marsh cores from Stiffkey and St. Annaland, the decrease in the estimate from 0.46 to 0.26 cm/yr was probably due to changes that have occurred in the sediment column due to compaction and decomposition. These changes are not as important near the sediment surface, so they would not impact rates from the ^{137}Cs peaks. Given the highly organic nature of these sediments, and the large changes in bulk density and organic content in this core (see discussion below), it is understandable that a large change such as this could occur.

There were large differences in sedimentation rates between the cores from these two sites when they were converted to mass based rates and divided into mineral and organic matter accumulation. Mineral matter rates from the Wistula River cores were 9 (low marsh) and 15 (high marsh) times greater than mineral rates from the Oder River samples. It is not clear what the source of mineral matter is at the Wistula River site. However, the area is heavily farmed, and it is possible that there are high rates of erosion from these adjacent agricultural sites. The organic accumulation rates from the two Wistula River cores were higher than any other cores that I collected. Rates of organic accumulation were also very high fore the Oder

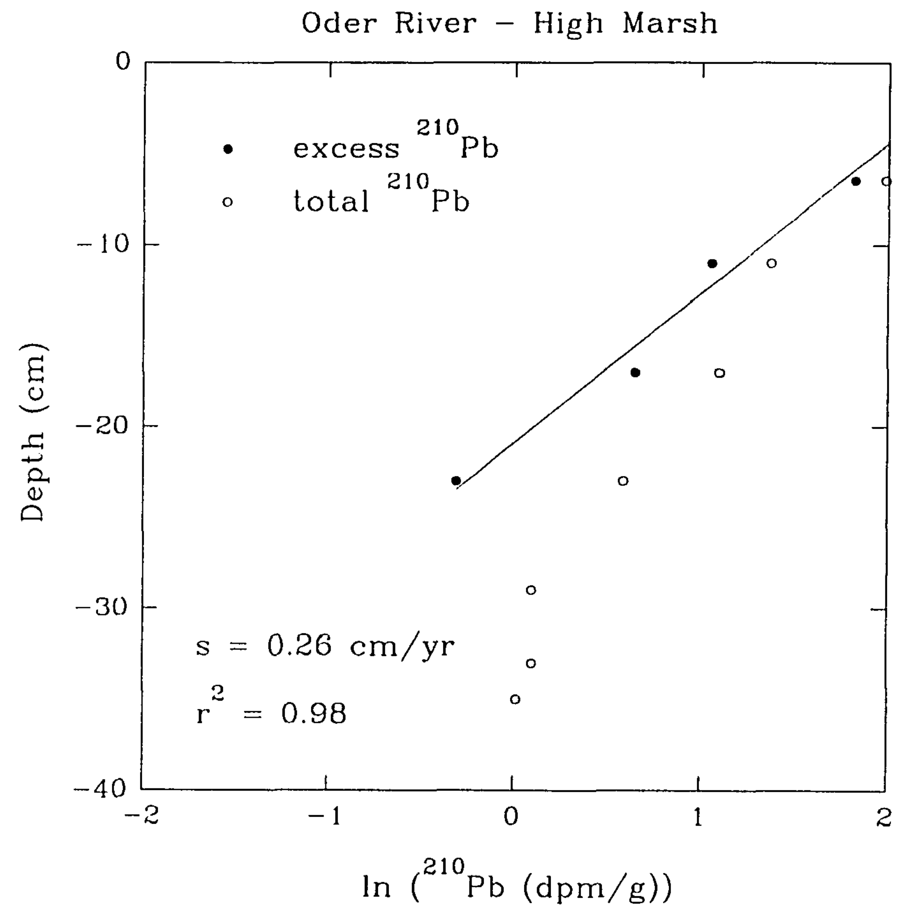

Figure 2.3e. ^{210}Pb profile from the high marsh core, Oder River, Poland.

cores. These areas are primarily accumulating peat material. Similar historic peat deposits have been documented throughout Szczecin Bay (Jasnowski 1962).

Profiles of organic content, bulk density, and sediment volume distribution

Interpretation of organic content, bulk density and volume distribution profiles is very difficult because multiple processes occur simultaneously throughout the sediment column and can affect the profile in different ways. Also, the initial organic content and bulk density of sediments can vary due to irregular deposition, from storms and other natural variability (Stumpf 1983, Reed 1989). This complexity is one of the reasons that I have chosen to use a simulation model to help interpret sediment dynamics (Chapter 4). However, there are some patterns that are apparent from the profiles (Figures 2.4a and 2.4b).

In general, there was a decrease in organic matter content with depth in most of the cores; whereas, bulk density increased with depth (Figures 2.4a and 2.4b). In comparing mean values of organic content from the top and bottom five sections from each core, 7 of the 10 cores had significantly lower organic content in the bottom five sections, based on ANOVA statistics. Differences in organic content with depth can also be demonstrated through regression analysis. In 8 of the 10 cores, there was a significant negative correlation of organic content and depth ($p<0.05$). Both of these results indicate that there was a significant decrease of organic material with depth, and this decrease was most likely due to the loss of organic matter through decomposition.

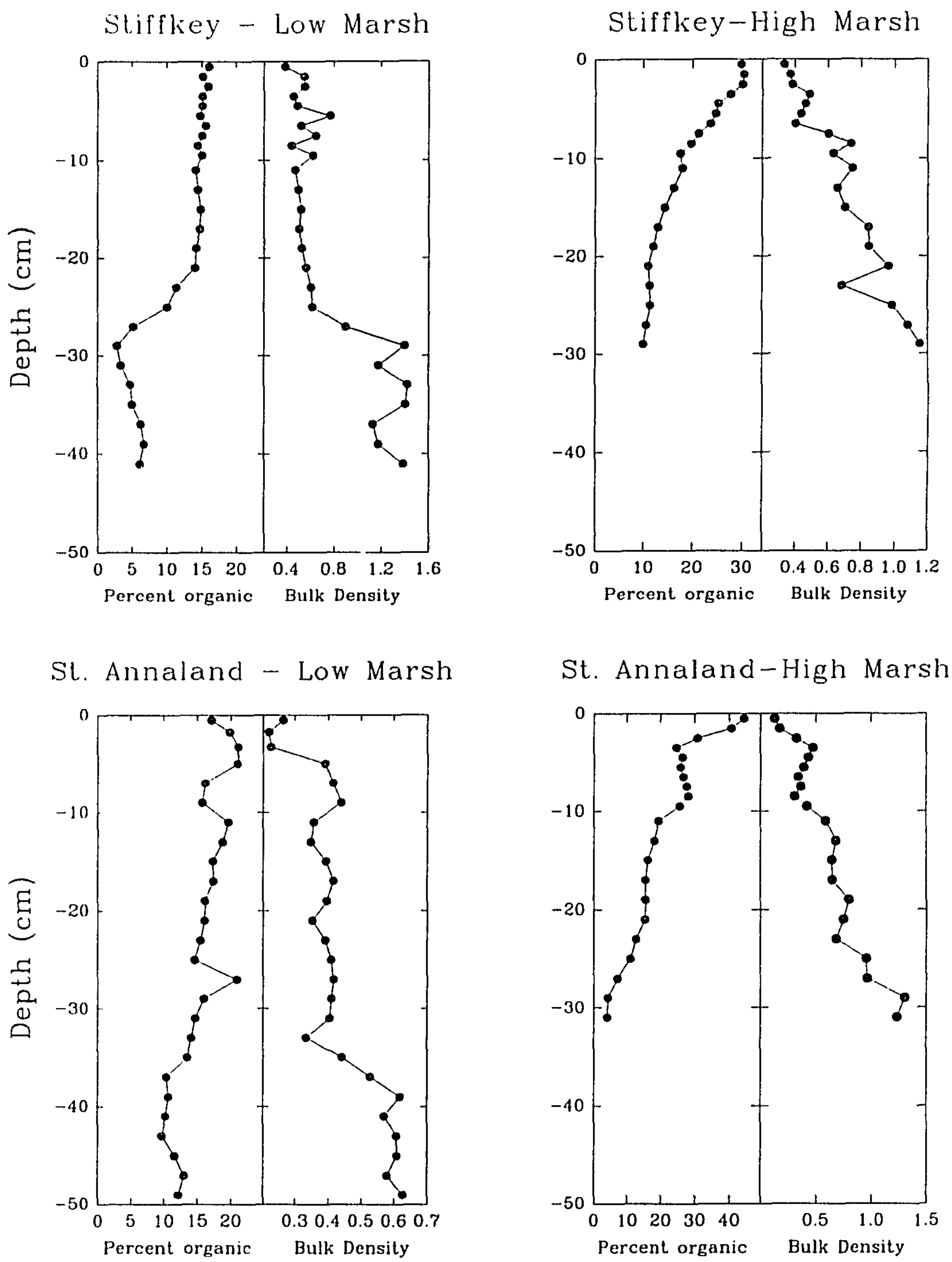

Figure 2.4a. Organic content (percent) and bulk density (g/cm^3) profiles for high and low marsh cores from Stiffkey and St. Annaland Marshes.

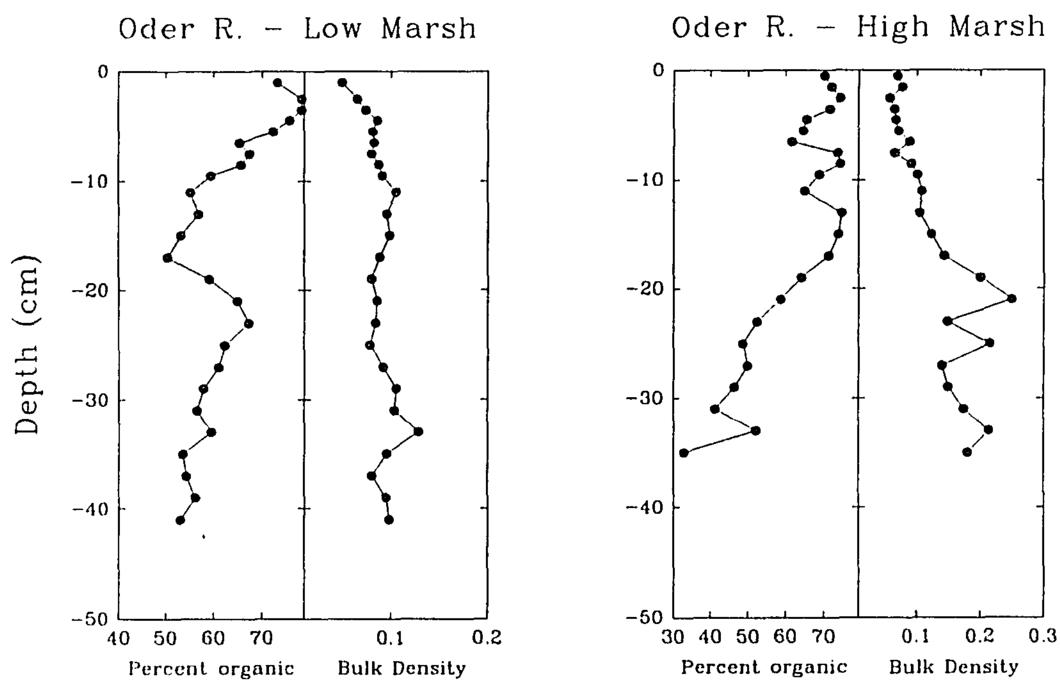

Figure 2.4b. Organic content (percent) and bulk density (g/cm^3) profiles for high and low marsh cores from the Oder River.

Evaluating changes in sediment bulk density, I found that in 7 of the 10 cores, bulk densities from the bottom five samples of each core were significantly greater than bulk densities from the top five samples of the same core. As with organic content, there was a large variation in bulk density with depth; however in 6 of the 10 cores there was a significant positive relationship between bulk density and depth ($p < 0.05$). This trend is apparent in graphs of bulk densities profiles from many of the sites, with a two- to three-fold increase in bulk density over the entire core (Figure 2.4a and 2.4b). The increases in bulk density were due to the loss of very light organic material through decomposition and to compaction (loss of pore space) in these surface sediments.

The effect of compaction on these shallow sediments can also be seen in the decrease in the volume of pore space with depth in the volume distribution profiles. Pore space decreased from 80 to 90 percent by volume near the surface to 50 to 70 percent near the bottom of the core in the samples from western Europe (Figure 2.5a). The cores from Poland had greater pore space near the surface but did not show consistent decreases in pore space with depth (Figure 2.5b). Some cores, such as the high marsh core from Stiffkey marsh, had decreases in organic content by weight (Figure 2.4a) and increases in organic content by volume (Figure 2.5a). This is due to the combined action of decomposition, which causes a decrease in organic content by both weight and volume, and compaction, which does not affect organic content by weight but increases organic content by volume. Because of these

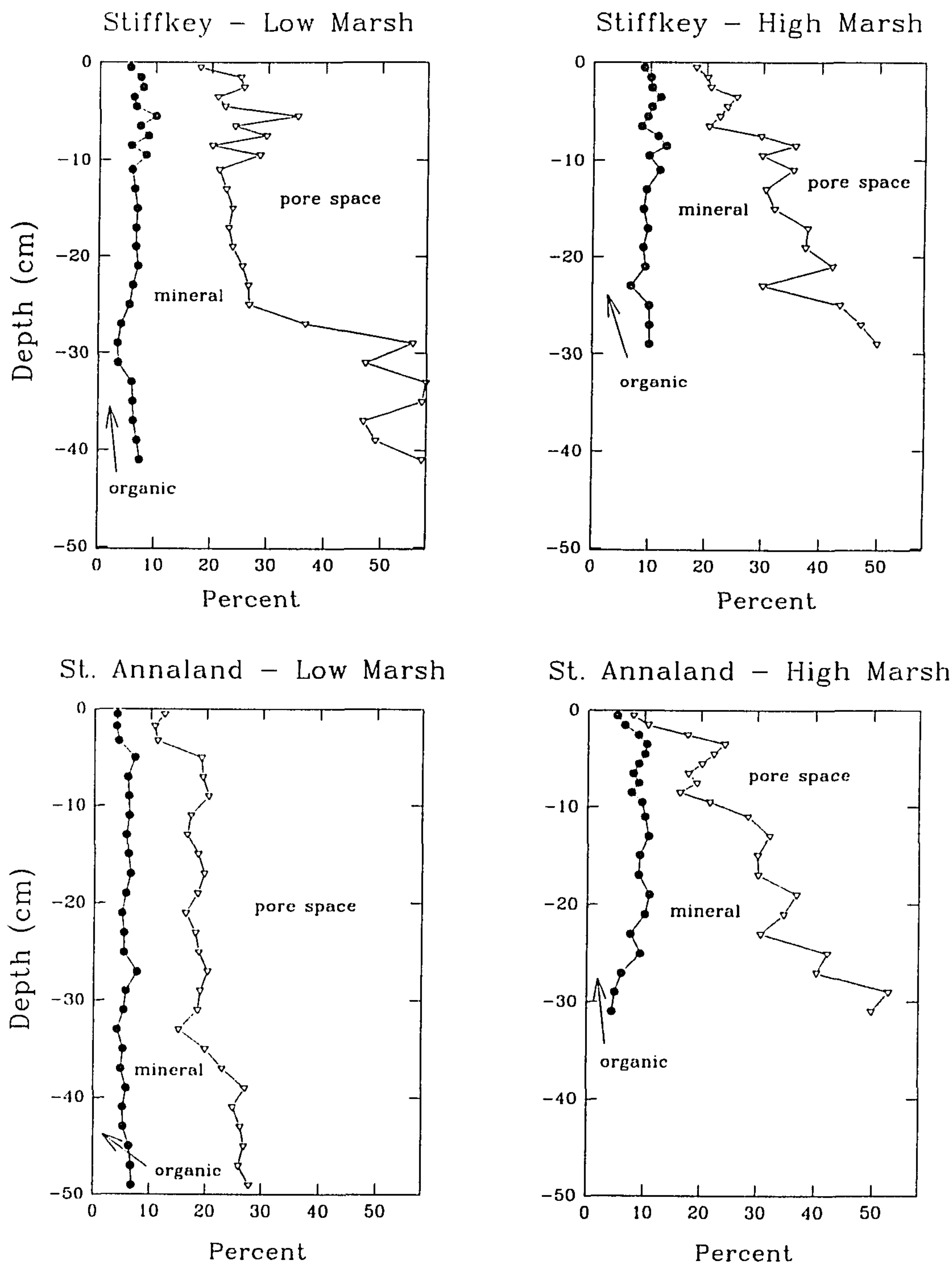

Figure 2.5a. Volume distribution profiles for high and low marsh cores from Stiffkey and St. Annaland Marshes.

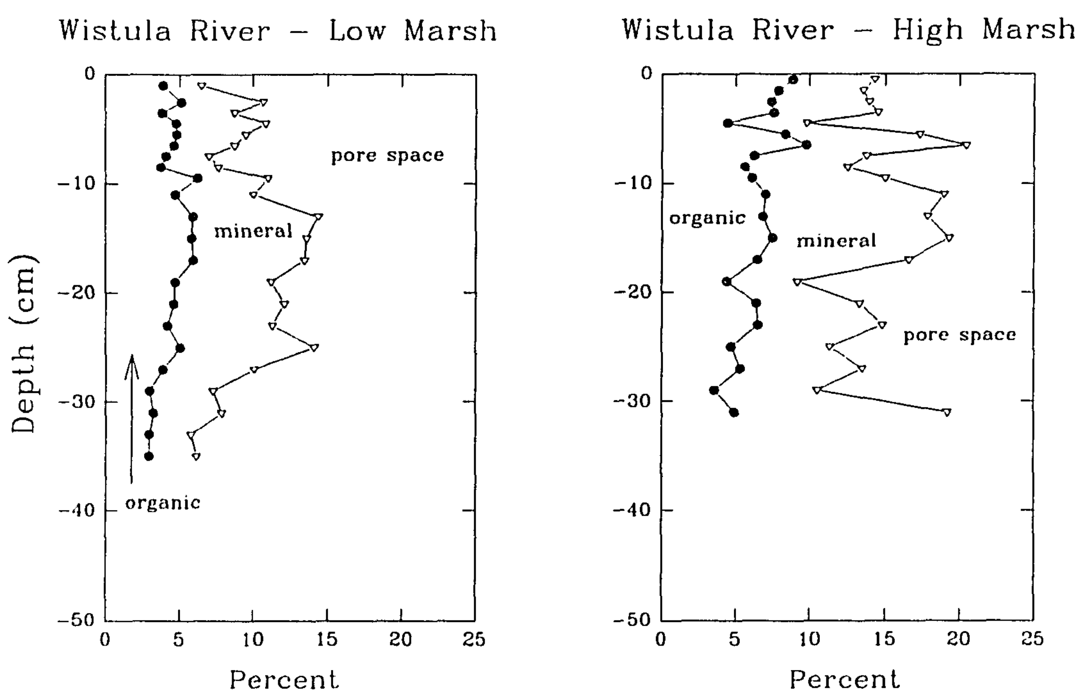

Figure 2.5b. Volume distribution profiles for high and low marsh cores from the Wistula River.

interactions, the use of a simulation model (Chapter 4) is very valuable in interpreting changes in sediment characteristics over time and depth.

Sea level rise

Time series analysis of tide gauge records have been used extensively to indicate local changes in relative sea level at many sites throughout the world (Emery and Aubrey 1991; Penland and Ramsey 1990). Data for nearby tidal stations were obtained from tidal station records by Emery and Aubrey (1991). No data were available for a station near St. Annaland marsh, so tide gauge data were obtained from the Tidal Waters Division of the Rijkswaterstaat (the Hague, the Netherlands). Linear regression analysis of mean tidal height versus time was used to determine the rate of relative sea level rise, following the technique used by Emery and Aubrey (1991). Although there are some problems with using tide gauge data for sea level rise analysis (Emery and Aubrey 1991, Turner 1991), these are the best data available for estimation of local changes in relative elevation.

Based on this type of an analysis of relative sea level rise, the vertical accretion rates from all of the cores that I collected were high enough to compensate for local rates of sea level rise (Table 2.5). The short term vertical accretion rates from ^{137}Cs dating were at least 1 mm/yr greater than relative sea level rise for all of the cores, so it does not appear that any of these sites are in short term danger of damage due to excessive flooding. Vertical accretion rates estimated from ^{210}Pb dating were closer to rates of sea level rise than the ^{137}Cs rates, but ^{210}Pb rates were still high enough to maintain marshes at the same elevation over the last 100 years.

Table 2.5. Rates of relative sea level rise, as estimated by tide gauge analysis from nearest long-term tide stations at each site.

	Nearest Station	**Time Period**	**Rate of Sea Level Rise** (cm/yr)
ENGLAND			
Stiffkey Marsh	Lowestoft	1955-1984	0.07
Dengie Marsh	Southend	1929-1982	0.15
NETHERLANDS			
St. Annaland	Bruinisse	1871-1990	0.23*
POLAND			
Oder River	Swinoujscie	1951-1982	0.17
Wistula River	Gdansk/ Nowy Port	1951-1982	0.26

*Rates are from Emery and Aubrey (1991), except for Bruinnisse. These data were obtainted from the Tidal Waters Division of the Rijkswaterstaat (the Hague, the Netherlands).

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CHAPTER 3

MULTIPLE REGRESSION ANALYSIS OF SEDIMENT ACCRETION PROCESSES IN COASTAL WETLANDS

INTRODUCTION

In recent years there have been many studies of accretionary processes in coastal wetlands. Much of this work has been done in the Mississippi River deltaic plain, where vertical accretion rates are large, but there are also very high rates of subsidence and coastal land loss (DeLaune et al. 1983, Hatton et al. 1983, Salinas et al. 1986). In other areas of the United States and Europe, where data have been collected on accretion rates, a wide range of results has been found, with accretion rates varying from 0 to 1.5 cm per year. Many of these marshes are accreting at a rate fast enough to compensate for present rates of sea level rise (Harrison and Bloom 1977, Griffin and Rabenhorst 1989, Patrick and DeLaune 1990), but some are being inundated and converted to open water habitats (Stevenson et al. 1985).

Despite the large number of recent studies evaluating sediment accretion rates, very little analysis has been done to synthesize the results from these different studies in order to determine if there are common factors from the various studies that are important in affecting accretion rates. Many factors have been proposed to be important in affecting accretion rates at a given site, but most of these proposed factors are intuitive, and have not been tested. Some of these factors include: plant community and density of vegetation (Richard 1978, Gleason et al. 1979, Stumpf 1983), tidal elevation (Harrison and Bloom 1977, Bricker-Urso et al. 1989), sediment

input from riverine, estuarine and marine sources (Salinas et al. 1986), proximity to the sediment source (French and Spencer 1993), total organic input from local marsh production (McCaffrey and Thomson 1980, Hatton et al. 1983, Bricker-Urso et al. 1989), and relative sea level rise (Gehrels and Leatherman 1989). In a recent review of salt marsh accretion rates, Stevenson et al. (1986) have shown that there is a strong correlation between tidal range and accretionary balance (the difference between accretion and subsidence). However, this was the only variable addressed in the study, and it is clear from other work that tidal range is not the only factor that influences accretionary balance. Also, results from Chapter 2 indicate that their conclusions may have been a result of the sites that they included in their analysis.

In order to interpret these factors, a large scale statistical analysis of the available data is needed. I have attempted to do this, using a multiple regression approach. It is clear that this is not the only approach to analyzing sedimentation processes. Much can be learned from more mechanistic approaches (van Erdt 1985), from modelling (French 1993, Chapter 5), and from additional field studies, but a general synthesis of factors could provide important insight into processes affecting accretion rates. Understanding the relative importance of the various primary factors discussed above will be very important for long-term management of coastal wetland systems. In light of the predicted increases in sea level rise (Gornitz et al. 1982), many changes must be made in coastal planning and policy in order to ensure the long-term survival of coastal wetlands (Titus 1986, 1991).

METHODS

Data were initially compiled from the literature and from my own data for 21 sites, including 82 cores from North America and Western Europe. Only studies which used either ^{137}Cs or ^{210}Pb dating were included in the analysis. I excluded marker horizon studies because of the potential for large variability in these short-term measurements. In addition, ^{14}C dating and other more long term techniques, such as pollen analysis, were excluded, because very few studies have been done with these methods and the inclusion of only a few data points could have skewed the analysis.

The data entries that I used consisted of a set of data for an individual sediment core: accretion rates, bulk density and organic content data, etc... I also identified site-specific data that would be useful for analysis, and these data were either taken from original references, accompanying citations from the same study site, or from other sources (Hicks and Hickman 1988, Penland and Ramsey 1990, Emery and Aubrey 1991, NOAA 1993) for tidal range data and rates of relative sea level rise . Data from many of the cores that I initially compiled had to be discarded from the analyses because they were not suitable for this study. For example, many studies gave information on accretion rates, but no data were available for sediment characteristics. In addition, some outliers were discarded because of unusually high or low accretion rates or independent variables. These points skewed the regression analysis. For the final analysis, 53 cores were used from 16 sites (Table 3.1 and see Appendix A for complete listing of data).

Table 3.1. Locations of sites, number of cores and references for samples used in regression analysis.

Location	Number of Cores	Method	Reference
Rhode Island	7	Pb	Bricker-Urso et al. 1989
Farm River, CT	1	Pb	McCaffrey & Thomson 1980
Flax Pond, NY	2	Pb	Armentano & Woodwell 1975
Great Marsh, DE	1	Pb	Church et al. 1981
Blackwater NWR, DE	2	Pb	Stevenson et al. 1985
North Carolina	4	Cs	Craft et al. 1993
San Francisco Bay, CA	2	Cs	Patrick & DeLaune 1991
Pacific Northwest	2	Cs	Thom 1992
Louisiana	6	Cs	Hatton et al. 1983
Biloxi Bay, MS	5	Cs	Chapter 2
Aransas NWR, TX	4	Cs	Chapter 2
San Bernard NWR, TX	6	Cs	Chapter 2
Florida Keys, FL	5	Cs	Chapter 2
Stiffkey Marsh, UK	2	Cs	Chapter 3
Dengie Marsh, UK	2	Cs	Chapter 3
St. Annaland, Neth.	2	Cs	Chapter 3

Data variables

I also had to pare down the approach in terms of the number of variables that I evaluated. I initially came up with a list of potentially important variables to evaluate, based on ideas presented in the literature and my own understanding of marsh processes. These variables include eight variables related to the physical characteristics of the sediments (organic content; mineral content; sediment bulk density; percent clay, silt and sand; annual suspended sediment load; redox conditions; pH; and soil salinity), four related to water level factors (local relative sea level rise, tidal range, height of sample relative to mean tidal level and tidal range, and wave energy of coast), nine related to biological factors (above-ground production and/or biomass; below-ground production and/or biomass; above-ground decomposition rates; below-ground decomposition rates; dominant plant species or communities; density of vegetation; root and rhizome types; contiguous marsh extent/area; and age of the wetland), and one methodological factor (the time period covered by the dating procedure). However, after compiling and reviewing the available data, I limited the analysis to the variables listed in Table 3.2. I limited myself to these not because I felt that the other variables were unimportant, but simply because data did not exist for most cores for these other variables. Some of the variables that I excluded may be extremely important, and they will be examined in further detail in the discussion.

I chose to look at two different measures of accretion rate to see if the various independent variables may be related to either of these accretion measurements.

Table 3.2. List of independent variables used in regression analyses.

Variable	Abbreviation	Units
(CORE SPECIFIC)		
sediment bulk density	bden	g/cm^3
surface organic content (0-10 cm)	org010	percent
ratio of surface and bottom organic content	orgratio	
position within the marsh*	posit	
time period of measurement	period	yr
(SITE SPECIFIC)		
relative sea level rise	rslr	cm/yr
tidal range	tdrnge	m
latitude	lat	deg

*class variable: low, middle and, high marsh

Previous researchers (Stevenson et al. 1986) found significant relationships with accretion balance rather than vertical accretion rates. Vertical accretion rates are simply the amount of material (cm/yr) that accumulates on the marsh surface over time. Accretion balance (cm/yr) is the vertical accretion rate minus the increase in relative sea level and reflects changes in the elevation of the marsh surface relative to sea level. Relative sea level rise was not used in the regression analysis for accretion balance since it was used in the calculation of accretion balance.

Data analysis

Correlation and multiple regression analyses were used to determine which factors were related to accretion rates. SAS was used for all data analysis. PROC REG, with the RSQUARE and CP options in SAS, was used in order to test the regression for all possible combinations of independent variables versus the two dependent variables. The independent variable "position within the marsh" was not used in this initial analysis, but was tested later, as explained below. The best regression model for each of the two dependent variables was chosen based on the highest R^2 and lowest C_P statistics (Montgomery and Peck 1982, Myers 1990). In each case, this model was then tested with the additional factor, position within the marsh. Because data were not available for actual elevations, this factor was treated as a categorical variable, with three levels: high marsh, middle marsh, and low marsh, and two indicator variables were used for position in the regression model (Myers 1990). In addition, interactions between position and the other variables were also tested.

I tested for multicollinearity (linear dependencies) between independent variables, which is often a problem in multiple regression analysis and can lead to misinterpretation or to erroneous results (Montgomery and Peck 1982). Variance Inflation Factors (VIF's) were used to measure multicollinearity (Belsley et al. 1980, Myers 1990). In no case did VIF's exceed 4, indicating that multicollinearity was not a problem with these variables. The influence of individual data points within the regression was analyzed by evaluating COVRATIO and DFFIT statistics from the SAS REG output, following the guidelines given in Belsley et al. (1980) and Myers (1990).

RESULTS

Using a single variable regression model, relative sea level rise was significantly related with vertical accretion rate ($R^2 = 0.47$). Other variables that had significant regression relationships with the two different measures of sediment accretion had low R^2 values (less than 0.1). Because only one of the independent variables had a high R^2 value in the single variable regression with vertical accretion rates, the use of multiple regression analyses was warranted.

Vertical accretion rates

Based on R^2 and C_p statistics, the best fitting model for vertical accretion rates included the variables: relative sea level rise, surface organic content, sediment bulk density, position within the marsh, and the interaction between relative sea level rise and position. The R^2 for this model was 0.83, and without position and the

interaction term in the model the R^2 value dropped to 0.64. Other models that fit the data well included these variables, as well as a single other variable, such as latitude, time period of measurement, and the organic ratio; however, the addition of these single variables added little to the fit of the regression, and based on C_p statistics the simpler model above was accepted.

Because position was a categorical variable, it was analyzed in the regression using indicator variables, and the interpretation of its regression coefficients is in relationship to the rest of the model coefficients (Myers 1990). The effect of position on the regression was to move the intercept of the regression, and the effect of the interaction between position and relative sea level rise was to change the slope of the coefficient for relative sea level rise, for each of the three different positions. Given this affect of position, the regression results can be best expressed as an equation for each of the three positions within the marsh:

low marsh: accrate = 0.798 + 1.06*rslr - 0.00664*org010 - 0.398*bden

middle marsh: accrate = 0.750 + 0.604*rslr - 0.00664*org010 - 0.398*bden

high marsh: accrate = 0.705 + 0.321*rslr - 0.00664*org010 - 0.398*bden

where accrate is the vertical accretion rate, rslr is the relative sea level rise, org010 is the surface organic content, and bden is the sediment bulk density.

The importance of the interaction term between relative sea level rise and position indicates that relative sea level rise has a different effect on accretion rate at the three different positions within the marsh, with the effect being greatest in the low marsh. This difference can be seen in the equations above (with the coefficient for

relative sea level rise being the largest for the low marsh) and in the relationship between relative sea level rise and accretion rate for the three positions in Figure 3.1. The largest difference in the effect of this interaction was between the low marsh and the other two positions. There was very little difference in the interaction effect between middle marsh and high marsh sites, and very similar results were produced if middle marsh and high marsh sites were combined into one category ($R^2 = 0.82$, with no difference between the two models, $p > 0.05$).

The largest single contribution of any of the variables in the regression was from relative sea level rise. Part of this is due to the large accretion rates and large sea level rise from the sites in Louisiana, but these are not the only cores that are contributing to this relationship. Influence analysis indicated that the samples from Louisiana marshes were influential in determining the regression (i.e., they had high COVRATIO and DFFIT values), and it is clear from a visual inspection of Figure 3.1 that they are important in the relationship between relative sea level rise and accretion rates. However, the influence diagnostics do not indicate that they are "outliers" or do not conform with the rest of the data. Instead, they reinforce and enhance the regression and should be included in the regression (Myers 1990). Further proof of the validity of the regression is the fact that similar results were obtained when I ran the model without the Louisiana data, although the R^2 value decreased to 0.50. Additional data from sites with large rates of relative sea level rise would confirm this relationship.

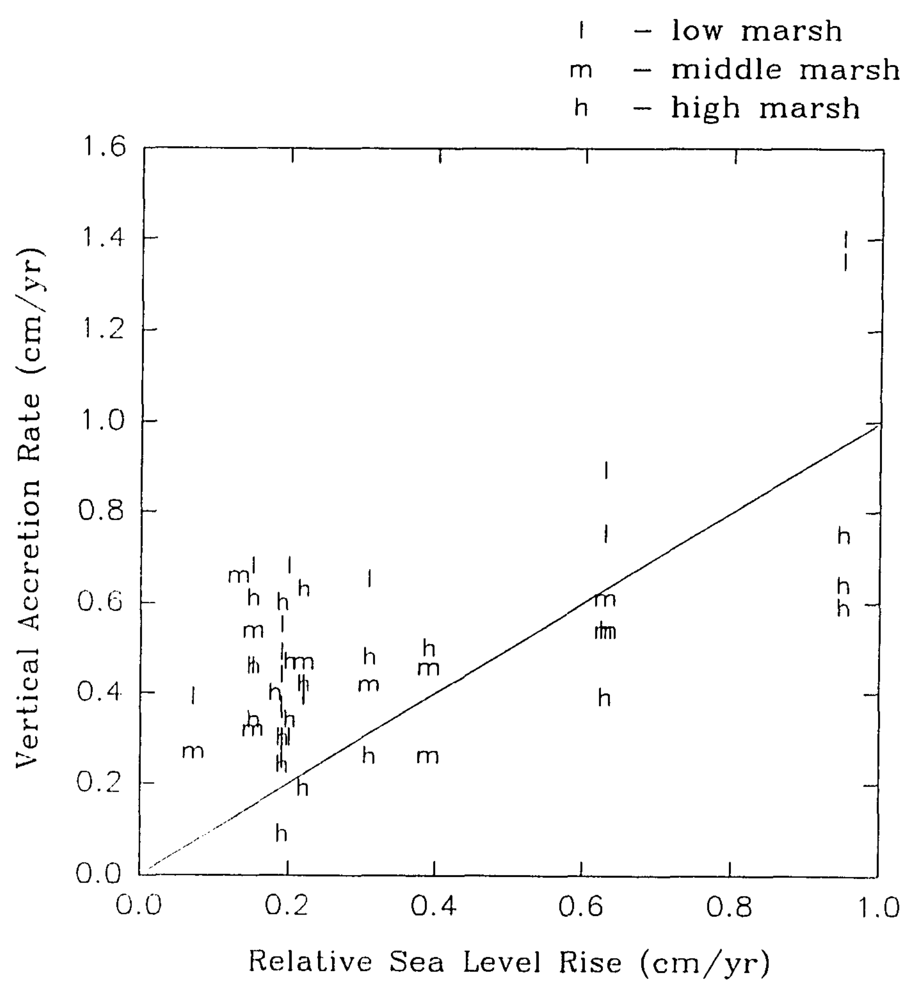

Figure 3.1. Relative sea level rise versus vertical accretion rate for the 53 cores used in the multiple regression analysis. Solid line indicates balance between sea level rise and accretion, and any sites below the line are experiencing a negative accretion balance.

In addition to the effect of relative sea level rise, the other two variables that were important were the surface organic content (org010) and the sediment bulk density (bden). Both of these had negative coefficients. The negative coefficient for surface organic content was due to the fact that samples with high surface organic content had low accretion rates. Increases in surface organic content of the sediment may cause increases in the short-term (1-10 years) vertical accretion rate, but these data all come from cores which were analyzed using ^{210}Pb and ^{137}Cs, so the rates are average rates based on 30-100 year measurements. Much of the surface organic material may be lost over longer time periods to decomposition and will not result in greater accretion rates.

Accretion balance

I obtained lower R^2 values for the regression for accretion balance. The best fitting model for this relationship, based on R^2 and C_p statistics included surface organic content, sediment bulk density and latitude. The R^2 for this model was 0.33 without position in the model and 0.43 with position. The regression equation for the three positions within the marsh was:

low marsh: accbalnc = 0.437 - 0.00705*org010 - 0.447*bden + 0.0062*lat

middle marsh: accbalnc = 0.437 - 0.00705*org010 - 0.447*bden + 0.0062*lat

high marsh: accbalnc = 0.437 - 0.00705*org010 - 0.447*bden + 0.0062*lat

where accbalnc is the accretion balance, org010 is the surface organic content, bden is the sediment bulk density, and lat is the latitude. This model is similar to the model for accretion rate, with the exception of the addition of latitude in the model.

However, the R^2 was much lower than that obtained for the vertical accretion rate model (which included relative sea level rise). There is a simple explanation for this low R^2 value. This regression model is very similar to the regression obtained for vertical accretion rates. Since accretion balance is defined as the vertical accretion rate (accrate) minus relative sea level rise (rslr), we can substitute this into the equation for accretion balance, obtaining:

accrate - rslr = (regression modelled intercept & coefficients)

OR: accrate = 1*rslr + (regression modelled intercept & coefficients)

From the equation above, it is clear we are forcing the relationship between vertical accretion rate and relative sea level rise to be 1. However, the results from the regression model for vertical accretion rate indicated that the relationship between vertical accretion rate and relative sea level rise was not 1. It changed between positions within the marsh, and varied from 0.32 to 1.06. Because we are forcing this coefficient to be 1 for all positions, when we perform the regression using accretion balance we get a very low R^2. If there is any significant relationship between vertical accretion rate and relative sea level rise that is not 1:1, this will always be the case.

DISCUSSION

Some studies have shown that there was no relationship between relative sea level rise and accretion rate (Wood et al. 1989, Stevenson et al. 1986, Chapter 2); however, those studies did not include as large a data set as I have analyzed. This

relationship has often been postulated based on analyses of marsh development (Redfield 1972, Gehrels and Leatherman 1989). Studies of other types of coastal environments have also stressed the importance of sea level rise in controlling sedimentation rates (Nichols 1989). Given the decrease in accretion rates with increases in relative elevation that have been documented within a particular wetland (Pethick 1981, Letzsch 1983, Chapter 2), it seems likely that wetlands in general could also respond to increases in sea level rise through changes in relative elevation.

However, it is clear that there are limits to their response, as can be witnessed by the loss of coastal wetlands in Louisiana (DeLaune et al. 1983, Salinas et al. 1986, Baumann et al. 1984) and Chesapeake Bay (Stevenson et al. 1985). Most of the coastal wetlands that have been studied for accretion rates come from geologically stable areas with rates of relative sea level increase close to 0.1 or 0.2 cm/yr. In order to further test the relationship between relative sea level rise and vertical accretion rates, more measurements from areas with medium to high rates of relative sea level rise (i.e., relative sea level rise greater than 0.3 cm/yr) are needed.

The interaction between relative sea level rise and position within the marsh indicates a complex impact of relative sea level across the marsh. Low marsh sites respond differently to sea level rise than do middle and high marsh sites, with low marsh sites being able to respond to increases in sea level with the largest increase in vertical accretion rates. As mentioned above, many field studies of accretion have obtained results indicating that the proximity of sediment sources, relative elevation, and drainage are all important in affecting vertical accretion rates, with the highest

rates of accretion in the low marsh (Richard 1978, Pethick 1981, Oenema and DeLaune 1988, Chapter 2). But the importance of the interaction effect between relative sea level rise and position within the marsh is that given the same increase in relative sea level, low marsh sites would respond differently than high marsh sites.

I have pointed out the importance of organic matter accumulation in low marsh sites (Chapter 2), and it is likely that in the upper marsh, the vegetation can not respond to increases in sea level. The lack of response could be due to poor drainage (Wiegert et al. 1983), or other effects on production in the high marsh, such as sulfide toxicity (Mendelssohn and McKee 1988). Additionally, other researchers have indicated that position within the marsh affects the mineral input of sediment to a site, and that changes in mineral inputs across the marsh are important to increasing overall accretion rates in the low marsh. If mineral sediments are most important at a particular site, then it is unlikely that high marsh sites could respond as greatly as low marsh sites to increases in sea level. An increase in sea level is not likely to increase sediment input to high marsh sites to a great extent because of the restriction of sediment delivery to upper marsh areas, unless there are large changes in hydrology within the marsh.

Other studies have also indicated that high marsh sites are very susceptible to conversion to lower marsh in areas where relative sea level rise is not great enough to cause wetland loss (Warren and Neiring 1993). The interaction that I found indicates that high marsh sites will be most vulnerable to increases in sea level rise because the increase in vertical accretion rates for a given increase in relative sea level rise is

lowest in this section of the marsh. This finding is confirmed by the fact that sites that are presently experiencing negative accretion balances are all high marsh sites (Figure 3.1). As these sites become flooded, they will either convert to low marsh sites, or if the drainage and/or sediment delivery are not sufficient they will likely convert to unvegetated tidal flats.

There was no significant relationship in this data set between tidal range and accretion balance, although this relationship has been proposed (Stevenson et al. 1986, Harrison and Bloom 1977). However, the only cores which had a negative accretion rate came from sites with tidal ranges less than 1 m (Figure 3.2). Given the current data set, it is not clear whether this is due to differences in tidal range, or simply due to the fact that all of the sites with high rates of subsidence and relative sea level rise come from areas with low tidal range. Data from Gulf coast sites with low tidal ranges (Chapter 2) demonstrated that marshes with low tidal ranges do not necessarily have low or negative accretion balances. However, it is possible that some low tidal range sites could be more vulnerable to submergence. In order to further evaluate this relationship, data are needed from areas with high rates of subsidence and large tidal ranges. With this type of data, we could identify if low tidal ranges or high rates of subsidence are most important in affecting the accretion balance.

Relationship between vertical accretion rate and organic and mineral accumulation rates

Vertical accretion rates can also be converted to mass-based rates and divided into organic and mineral components, using sediment organic content and bulk density profiles. The relationship between vertical accretion rates and these mass-based

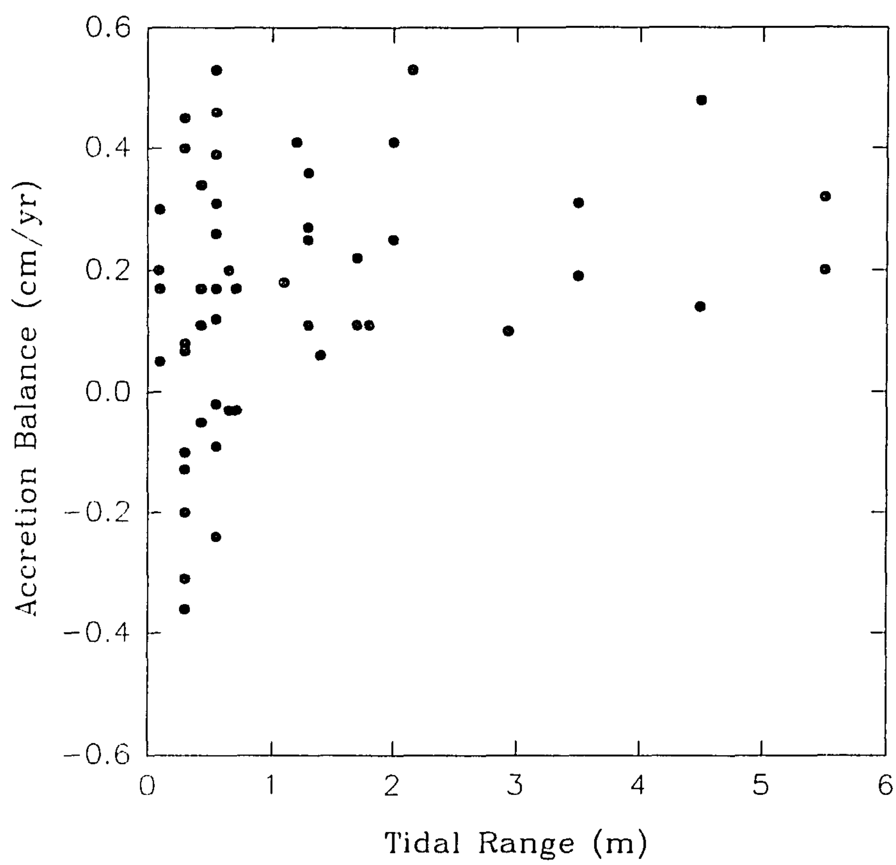

Figure 3.2. Tidal range versus accretion balance for the 53 cores used in the multiple regression analysis.

accumulation rates have been used in previous studies from individual sites (Nyman et al. 1993, Bricker et al. 1989) and from the Gulf coast (Chapter 2) to provide insight into the processes controlling sedimentation dynamics. The results from this large data set confirm the results from these previous studies. Vertical accretion rates were significantly correlated with organic accumulation rates ($r^2 = 0.69$, $p = 0.0001$) Figure 3.3, top). The relationship between vertical accretion rates and mineral accumulation rates was not nearly as strong ($r^2 = 0.14$, $p = 0.0068$) (Figure 3.3, bottom). Mineral and organic accumulation rates were significantly correlated ($r^2 = 0.15$, $p=0.0057$); however, this appears to be a very flat relationship (Figure 3.4). As hypothesized by Bricker et al. (1989), there may be an asymptotic relationship between these two variables.

Despite the large variability in marsh types that were included in the analysis, there was a very strong relationship between vertical accretion and organic accumulation rates across all of these different marsh types. I feel that the best explanation for this is that organic matter provides structure in coastal wetland sediments, and that this sediment structure is most important in determining vertical rates of accretion. Results from the simulation model (Chapter 6) also indicate that sediment structure is extremely important in affecting accretion rates. The most important question for marsh survival is then - what controls underground organic accumulation rates and sediment structure? Although organic matter accumulation is most important, mineral matter inputs are extremely important in affecting plant growth and hence organic matter accumulation (Nyman et al. 1993).

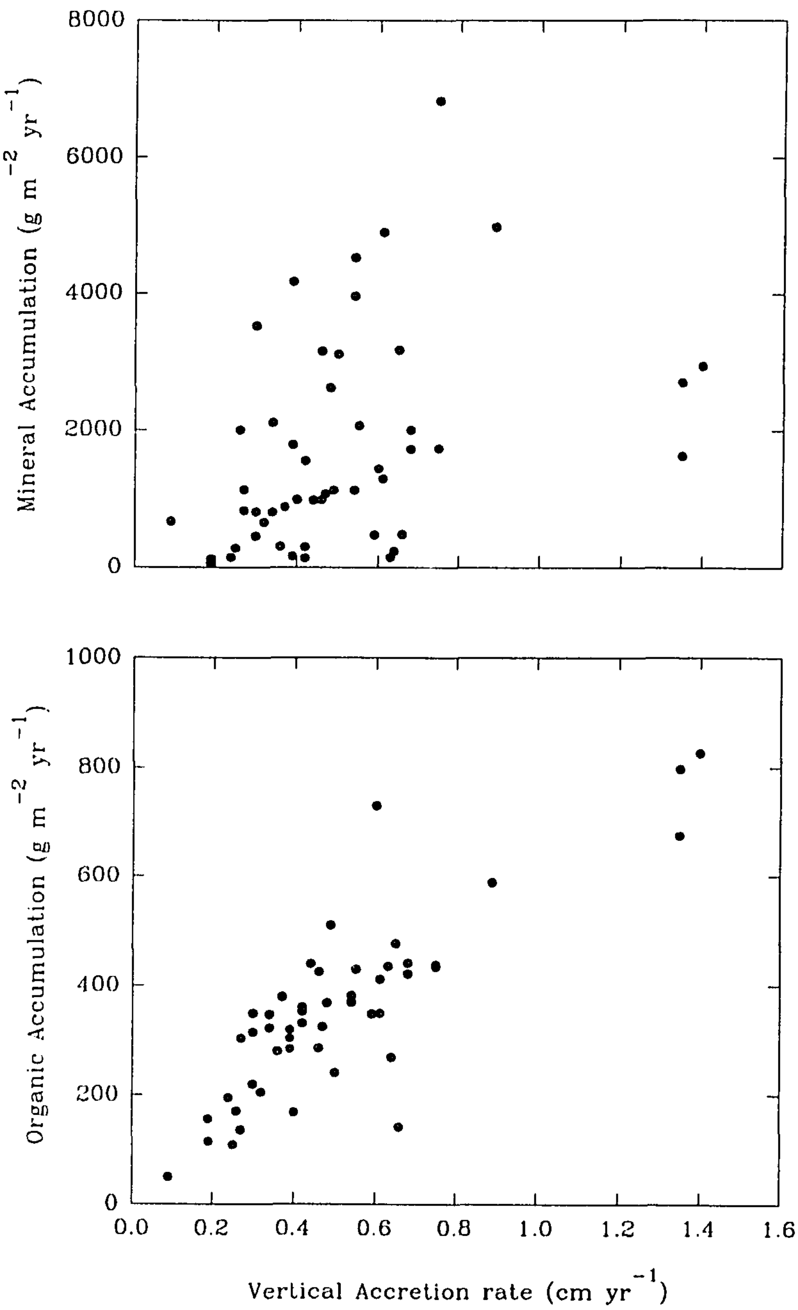

Figure 3.3. Correlation of vertical accretion rates with mineral accumulation rates (top) and organic accumulation rates (bottom) for all cores.

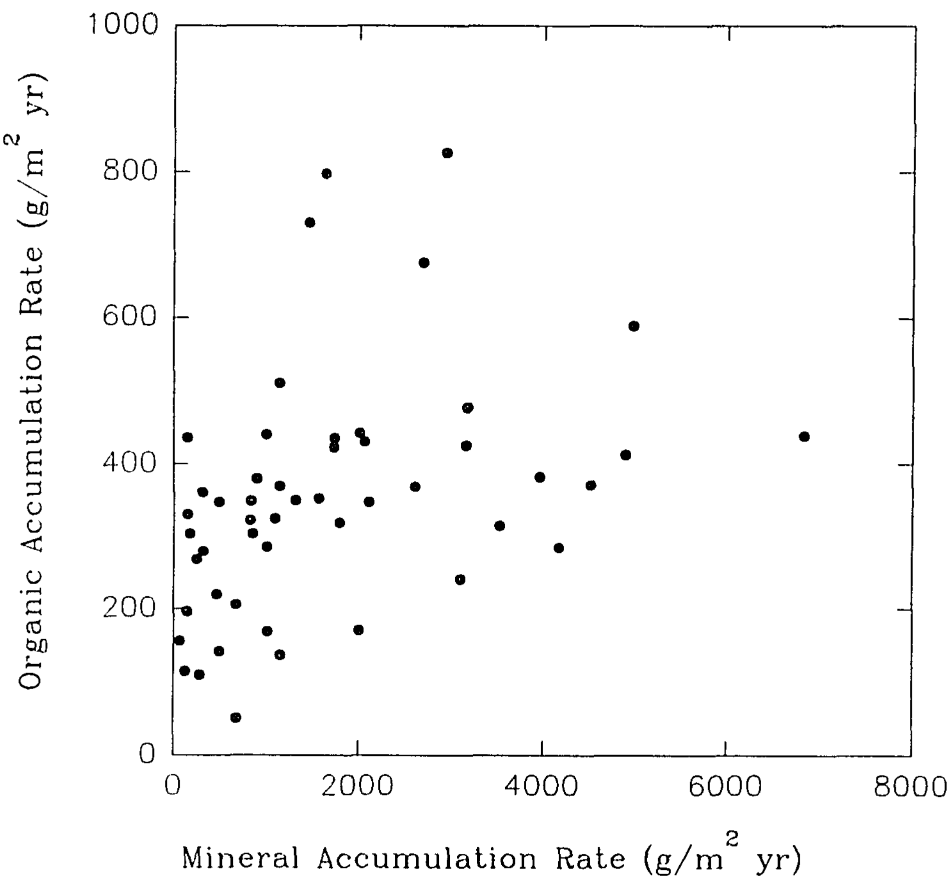

Figure 3.4. Correlation of mineral and organic accumulation rates for all cores.

Excluded variables and future studies

Some of the variables that I originally wanted to include in the analysis, but which were dropped due to the lack of available data, may be very important for analysis of sedimentation processes. Given the important relationship that I have found between organic accumulation and vertical accretion rates, it would be very desirable to have additional data on rates of below-ground production and decomposition. Most of the organic matter that accumulates within the sediment comes from below-ground production and not from surface deposition (Oenema and Delaune 1988). If estimates of below-ground production were available for areas where rates of organic accumulation have been measured, I would expect a high degree of correlation between these variables. Also, it would be interesting to see if the trends in organic accumulation that have been found across the marsh (Chapter 2) are paralleled in below-ground organic matter production. Similarly, rates of below-ground decomposition would give more insight into the processes that are controlling organic matter accumulation.

Secondly, much better data are needed in order to fully evaluate the importance of relative elevation and distance from a tidal channel in affecting accretion rates. If data were available on the actual elevation of the sampling site (relative to tidal range and mean tidal level), this variable could be treated as continuous rather than as a categorical variable as I have had to do. Because the classification of a site as high, medium, or low is somewhat arbitrary, this classification may have added error to the regression, and the actual elevation is likely

to give much better results in future regression analyses. Also, better data are need for the distance of the site to the sediment source, usually the adjacent tidal channel. This factor is related to elevation and has been shown to be important in minerogenic marshes in the UK (Stoddart et al. 1989, French and Spencer 1993) and may be important in other coastal wetlands. However, this type of data has rarely been collected, despite the fact that it is easily measured and could be added to the other routine data collected in accretion field studies. Without data on this factor from more sites, its effect versus the effect of relative elevation can not be adequately evaluated.

In the future, studies of marsh accretion should report mineral and organic matter accumulation rates. Many past studies only reported vertical accretion rates and so they could not be used in this analysis. The relationship between the two sediment components (as well as pore space) and vertical accretion rates is fundamental to a complete understanding of sedimentation processes in coastal wetlands.

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CHAPTER 4

SEDIMENT ACCRETION IN COASTAL WETLANDS: A SIMULATION MODEL OF PROCESSES

INTRODUCTION

In the last decade, wetlands have come under increasing protection as the general public becomes more aware of their ecological importance (National Wetlands Policy Forum 1988). Despite legal protection, the long term survival of tidal wetlands is not a simple issue due to the complex ecological processes which naturally maintain tidal marshes. Because of their location at the land/ocean interface, coastal wetlands must maintain an elevation within the tidal range or they will cease to function as wetlands (Baumann et al. 1984, Salinas et al. 1986). If the surface of the marsh becomes too low, plants can become stressed, and the marsh may disappear (Mendelssohn et al. 1981) . Conversely, if the marsh collects excessive sediment, its elevation may become so high that upland species will colonize and may outcompete wetland plants.

The relative elevation of coastal wetlands is a function of a variety of factors, including eustatic sea level rise, subsidence (which in itself includes many factors), and the vertical accretion of mineral and organic sediments. Present estimates of global sea level rise range from 0.12 cm/yr to 0.24 cm/yr (Gornitz et al. 1982, Peltier and Tushingham 1989, Woodroffe 1993). Recent estimates of future eustatic sea level rise vary from 20 to 115 cm by the year 2100 (Woodroffe 1993), and these increases are likely to have enormous impacts on coastal wetlands world-wide (Titus

1986, 1991). Tidal marshes have developed during periods of relatively low sea level rise and rapid accumulation of mineral and organic sediments. They can withstand some increases in sea level rise; however, at present there is no clear understanding of the ability of marshes to compensate for increased rates of sea level rise (Orson et al. 1985).

Many methods have been used to measure accretion rates, and each method gives a result relative to a different time period (DeLaune et al. 1978, McCaffrey and Thomson 1980, Cahoon and Turner 1989, Boumans and Day 1993). The time period covered by a particular dating procedure is very important in interpreting results of accretion studies because as material accumulates on the surface of the marsh, other below-ground processes (including organic production, physical compaction, and decomposition of organic material) occur and affect the absolute level of the marsh. All of these processes are occurring simultaneously, and it is difficult to determine from a single measurement the importance of each of the various processes for a given site. Marker horizon studies give information on short-term accretion rates, which probably only reflect accumulation events on the surface of the marsh. Whereas, long-term measurements (radioisotope dating, such as ^{137}Cs and ^{210}Pb) integrate rates over time, reflecting accumulation at the surface of the marsh, as well as processes that occur within the sediment column (Stevenson et al. 1986). The results from these measurements are average rates relative to a given time period. Processes at depth are continually occurring and are integrated throughout the depth of the sediment over time. A technique that covers a much longer time period such

as ^{14}C will usually result in lower accretion rates because of sediment compaction and decomposition of sediment organic material (Figure 4.1). Other methods, such as surveying (Anderson et al. 1981) and sedimentation-erosion tables (Boumans and Day 1993) indicated changes in sediment surface elevation that are relative to a stable benchmark. Because of these complex interactions of time and depth it is not possible to directly compare accretion measurements that have been made with different techniques, and the relative magnitude of each process can not be identified without modelling the interaction of time, depth and below-ground processes.

Previous researchers have also used models to examine sediment processes in coastal wetlands. Morris and Bowden (1986) used a cohort model to evaluate sediment nutrient dynamics in a tidal fresh water marsh; however, the focus of their model was not sedimentation processes. Cohort models follow changes in individual groups or cohorts of sediment over time. Chmura et al. (1992) used a simulation model to evaluate changes in elevation and accretion; however, their model did not use yearly cohorts. Instead they divided the sediment column into three layers and assumed that rates of processes were similar within each layer, which is probably an oversimplification of below-ground processes. Also, their model did not allow for comparisons of modelled and actual sediment characteristics, such as bulk density and organic content (Chmura et al. 1992). French (1993) has modelled sediment dynamics from a English salt marsh, but he focused on mineral inputs to marsh. His calculated rates of mineral sediment inputs are very precise, but he has not considered changes in the sediment due to compaction or decomposition.

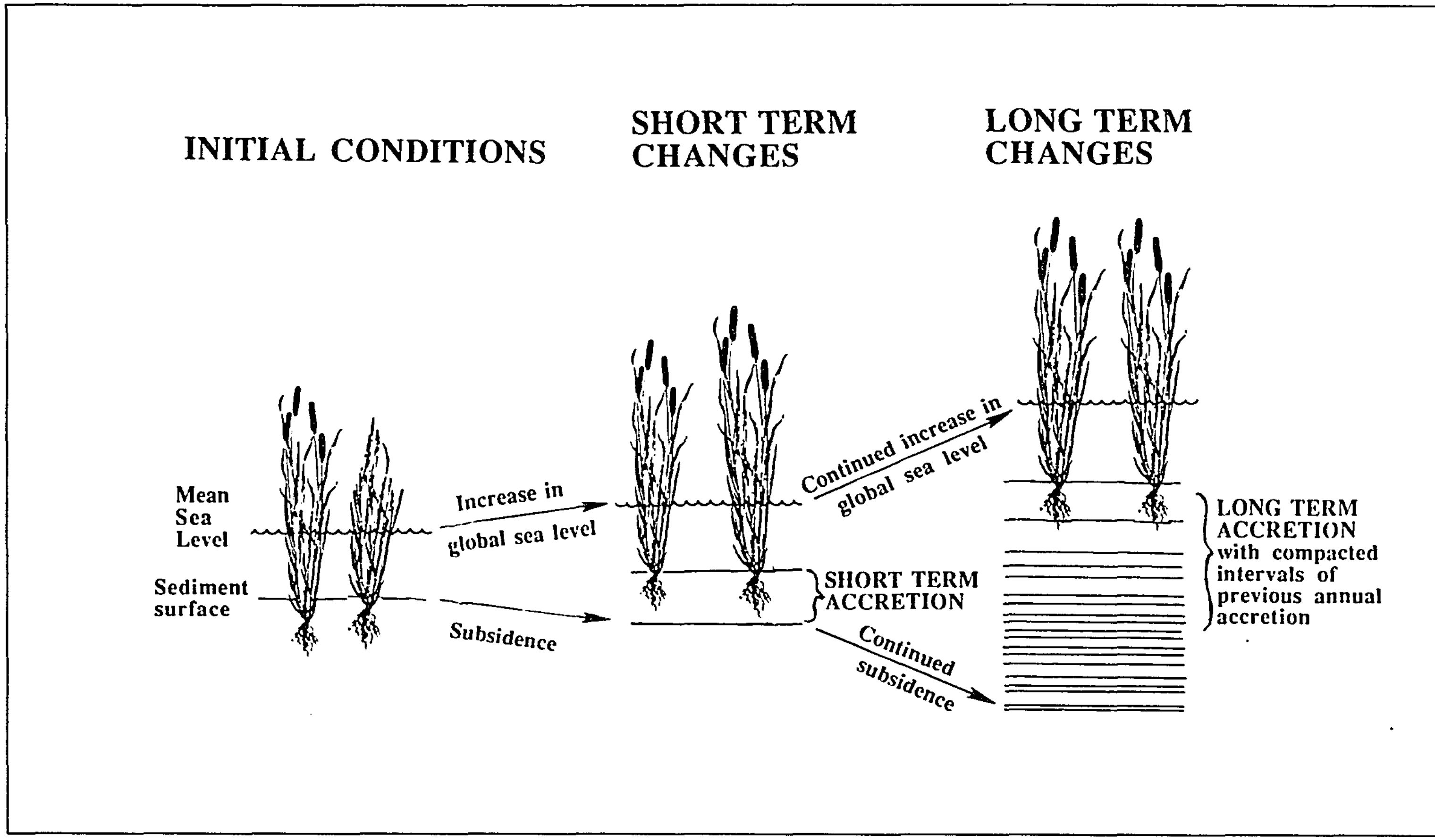


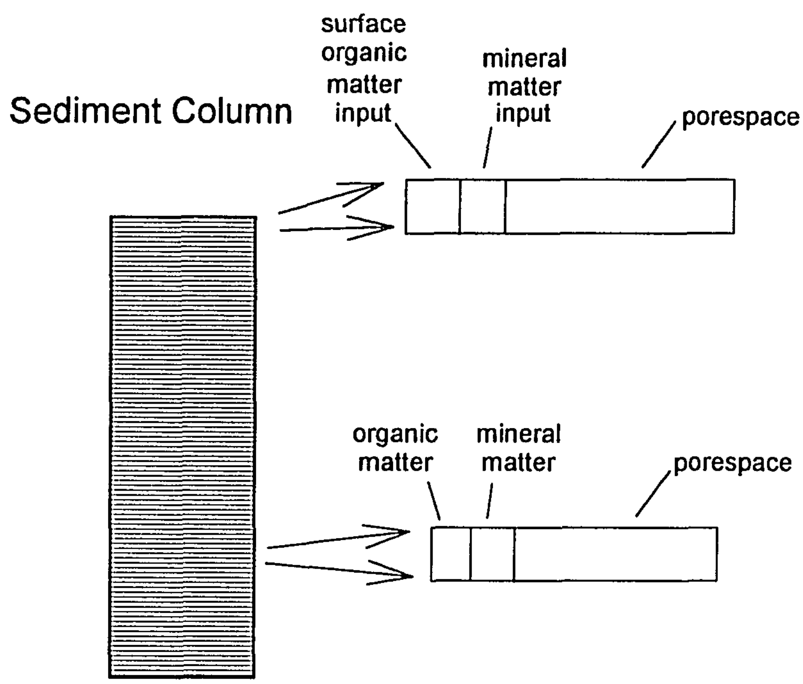
Figure 4.1. Conceptual model of changes to coastal wetland sediments over time.

The basic approach of the model described here is to simulate changes in sediment characteristics which occur over time as sediment accumulates on the surface of the marsh. In addition to the inputs at the sediment surface, changes are simulated throughout the sediment column, including below-ground organic matter production, decomposition, and compaction. I used a cohort approach, similar to the approach used by Morris and Bowden (1986), following yearly changes in each cohort of mineral matter, organic matter and pore space. This model is designed to be used for a particular site or sediment core. Site specific parameters, such as underground production rates, mineral deposition rates, or decomposition rates can easily be changed in order to calibrate the model for a particular site.

METHODS

Model parameters

In the model, mineral and organic matter accumulated on the surface of the marsh every year, and all cohorts of sediment (organic matter, mineral matter, and sediment pore space), were moved from one age class to the next. After a cohort had been moved, changes due to decomposition, below-ground organic production and shallow compaction were calculated. The model followed both the mass and volume of each cohort (Figure 4.2). This was necessary in order to keep track of the depth to which each cohort was buried, and this depth determined below-ground organic production and decomposition rates. The time step for the model was 0.1 years, and



Surface organic matter input = f(above ground production, tidal range)
mineral matter input = f(elevation)

org(t + 1) = org(t) + root production - decomposition
porespace(t + 1) = porespace(t) - compaction

Figure 4.2. Diagram of the processes which are included in the simulation model.

the model was run for 300 years. A complete listing of the Fortran program is given in Appendix B. The following processes were included in the model:

1. SURFACE INPUTS

a. INORGANIC SEDIMENT INPUTS

In modelling the relationship between mineral sediment inputs and sediment surface elevation, I have simplified ideas from an earlier model of mineral sediment dynamics (French 1993), because I am not using a daily time step. Mineral inputs are a function of elevation, relative to mean sea level, with maximum values at elevations below mean sea level and a linear decrease to zero input at mean high water.

b. ORGANIC SEDIMENT INPUTS

a. Surface inputs - This was assumed to be constant and a function of above-ground organic production at the site. Future versions of the model will make above-ground production a function of elevation. A percentage of the organic matter input was designated as refractory and was not susceptible to decomposition.

2. BELOW-GROUND ORGANIC PRODUCTION

The relative distribution of below-ground organic production was modelled using an exponential decrease in production with depth (Figure 4.3). Total annual production, as integrated over the entire depth of the modelled sediment column, was constant over the entire run of the model. The coefficient for the exponential decay was calibrated based on organic content profiles for the particle core. As with surface

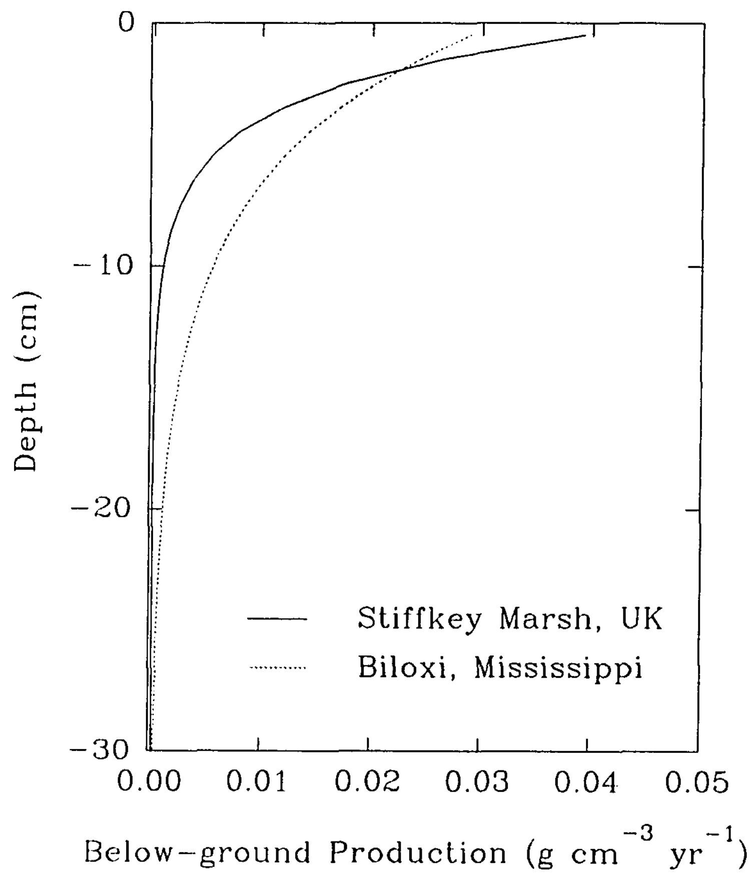


Figure 4.3. Depth profiles of model-calibrated below-ground organic matter inputs for Stiffkey and Biloxi simulation model.

inputs, a percentage of the below-ground organic matter input was designated as refractory.

3. DECOMPOSITION

This was modelled using a three stage decomposition process. Newly produced organic matter (surface inputs as well as below-ground inputs) had the highest decomposition rate. One year old organic matter had a slightly lower rate of decomposition, and older organic matter had a much lower rate. This approach assumed that most labile material was decomposed very readily, and what remained after two years was relatively refractory. In addition to differences between stages, decomposition rates for all three stages were a function of depth, with an exponential decrease in all decay rates with depth (Figure 4.4).

4. COMPACTION / CHANGES IN PORE SPACE

In most wetland sediments, the bulk of the sediment is occupied by air or water - pore space. I started the model with a pore space based on the average pore space of the top 5 cm of the actual core. This value was expressed as a percentage of the total sediment volume for a particular cohort because this is the most common way that these values are given in the literature. As more sediment accumulated on the surface of the marsh, the pore space decreased asymptotically to a pre-set minimum value. The minimum value was chosen based on pore space values from the bottom of the core, assuming that most of the loss of pore space occurs in the upper part of the sediment column. The decrease was a function of the density of material above a particular cohort, such that for each cohort the annual decrease was

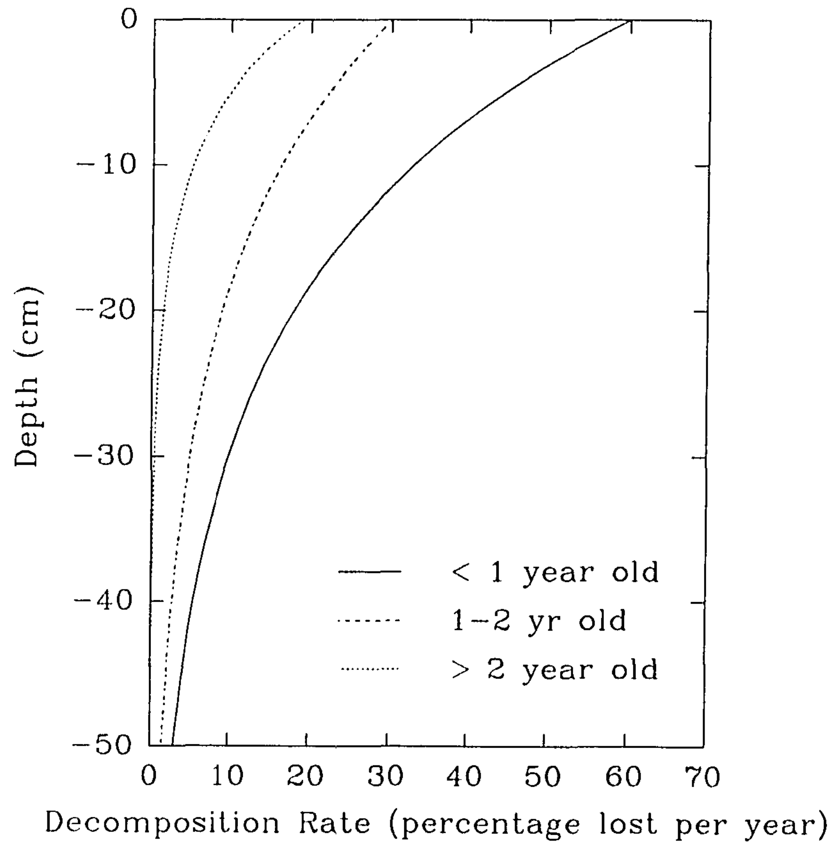


Figure 4.4. Depth profiles of model-calibrated decomposition rates for the three age classes of organic matter for the Stiffkey simulation.

equal to: $1 - (c/(k_1+c))$. C is the density of material above the cohort, and the constant k_1 controls the rate of decrease in the curve and was chosen to obtain the best fit to actual data from the cores.

Additional variables of interest

In addition to changes in the mass of organic and inorganic sediments over time, the following variables were calculated for each sediment cohort:

1. Volume (This was calculated based on the specific density of organic and mineral matter (assumed to be 1.14 and 2.61 g/cm^3, respectively), and the percentage of pore space in the sediment. See above for changes in these.);
2. Depth (This was equal to the volumes of all of the cohorts above a particular cohort);
3. Mass above the cohort (summed for each cohort);
4. Sediment organic content (percent); and
5. Sediment bulk density (g/cm^3).

Other variables included in the model were:

1. Relative elevation of sediment surface;
2. Eustatic sea level rise (0.15 cm/yr); and
3. Subsidence due to all factors except for compaction of surface sediments.

Calibration of the model

I calibrated the model to fit data from two cores taken from salt marshes which were studied for Chapters 2 and 3. The cores came from the high marsh

station at Stiffkey Marsh in Norfolk, England (Chapter 3) and from the mid marsh station at Biloxi Bay, Mississippi (Chapter 2). The first site represents a high tidal range coastal wetland with relatively low accretion rates and very high sediment mineral content; whereas, the second site has a low tidal range, higher accretion rates and higher organic content. The data that were used from the core for calibration included: accretion rates from ^{137}Cs dating, bulk density, organic content, and sediment volume distribution profiles. The parameters used for each calibration are given in Table 4.1.

Output from the model was a data file with complete sediment characteristics (organic mass, mineral mass, and volumes of organic, mineral matter and pore space) for each yearly cohort. These data were used to prepare sediment profiles of bulk densities and organic content. Because I knew the age of each cohort, I could calculate accretion rates for the model to compare to actual values. I also made a visual comparison of actual organic content and bulk density profiles and model generated profiles in order to calibrate the model. The model was run with a series of values for each individual parameter, and the output was compared to actual data in order to determine the best values for each parameter.

Sensitivity analysis

The relative importance of the various parameters of interest was tested using sensitivity analyses. A series of runs for each parameter was performed with step-wise increases (1, 3, 5, 10, 25, 33, 50, 100 and 200 percent increases) and decreases (1, 3, 5, 10, 25, 33, 50, 75, and 90 percent decreases) for each parameter, while all

Table 4.1. Parameters needed to run simulation model and values for calibrations for both Stiffkey Marsh, UK and Biloxi, Mississippi.

PARAMETER	Variable Name	Stiffkey Calibration	Biloxi Calibration
Surface Mineral Matter Depostion			
peak mineral matter deposition	minin	0.48	0.6 g/cm^2
Surface Organic Matter Deposition			
total surface organic matter deposition	orgin	0.03	0.5 g/cm^2
percent refractory organic matter	refrac	6.0	1.0 percent
Below Ground Organic Matter Production			
total below ground production	undpro	0.12	0.18 g/cm^2
decay constant for production curve	kdist	0.4	0.175
Decomposition			
Year 1			
Maximum decomposition rate	mx1	60.0	70.0 percent
decay constant for decomp. rate curve	kdec1	0.06	0.06
Year 2			
Maximum decomposition rate	mx2	30.0	35.0 percent
decay constant for decomp. rate curve	kdec2	0.06	0.06
Year 3 and later			
Maximum decomposition rate	mx3	19.0	30.0 percent
decay constant for decomp. rate curve	kdec3	0.13	0.125
Compaction			
curve constant	k1	170	250
initial pore space	h2oin	82.0	90.0 percent
final pore space	porelim	40.0	68.0 percent
Other variables			
Eustatic Sea Level Rise	slr	0.15	0.15 cm/yr
Subsidence below zone of current sedimentation	subsid	0.0	0.075 cm/yr
Tidal range	tdrnge	600.0	55.0 cm
Initial elevation (relative to MSL)	intelv	250.0	20.0 cm
Organic particle density		1.14 g/cm^3	
Mineral particle density		2.61 g/cm^3	

other factors were held constant. The relative changes in 5, 30, and 100 year accretion rates were compared for each of these runs. In some cases the entire range of increases or decreases was not run because extreme changes resulted in meaningless values for the particular parameter, i.e., a yearly decomposition rate of greater than 100%.

RESULTS

Calibration of the model

I was able to closely simulate measured accretion rates for the core from Stiffkey marsh using the Fortran model. For this core, the ^{137}Cs peaks from 1986 (Chernobyl accident) and 1963 (nuclear weapons testing) were found at 1.5 and 7.5 cm ($\pm$ 0.5 cm) respectively. For comparison, the model predicted that these peaks would be found at 1.6 and 8.8 cm, respectively. Similarly, for the Biloxi core, the 1963 ^{137}Cs peak was found at 15 cm ($\pm$ 1 cm) in the core, and the model predicted it to be at 15.8 cm.

The fit of the model to the actual data also can be seen in the profiles of bulk density, organic content, and the volume distribution of organic matter, mineral matter, and pore space, for the model and for the actual data (Figures 4.5 and 4.6). Because the goal of the model was to match the data from the core, and actual data was used to calibrate the model, it is naturally to be expected that the model closely resembles actual data. However, the model is still valuable because it can be used to identify important processes that affect overall accretion rates through sensitivity

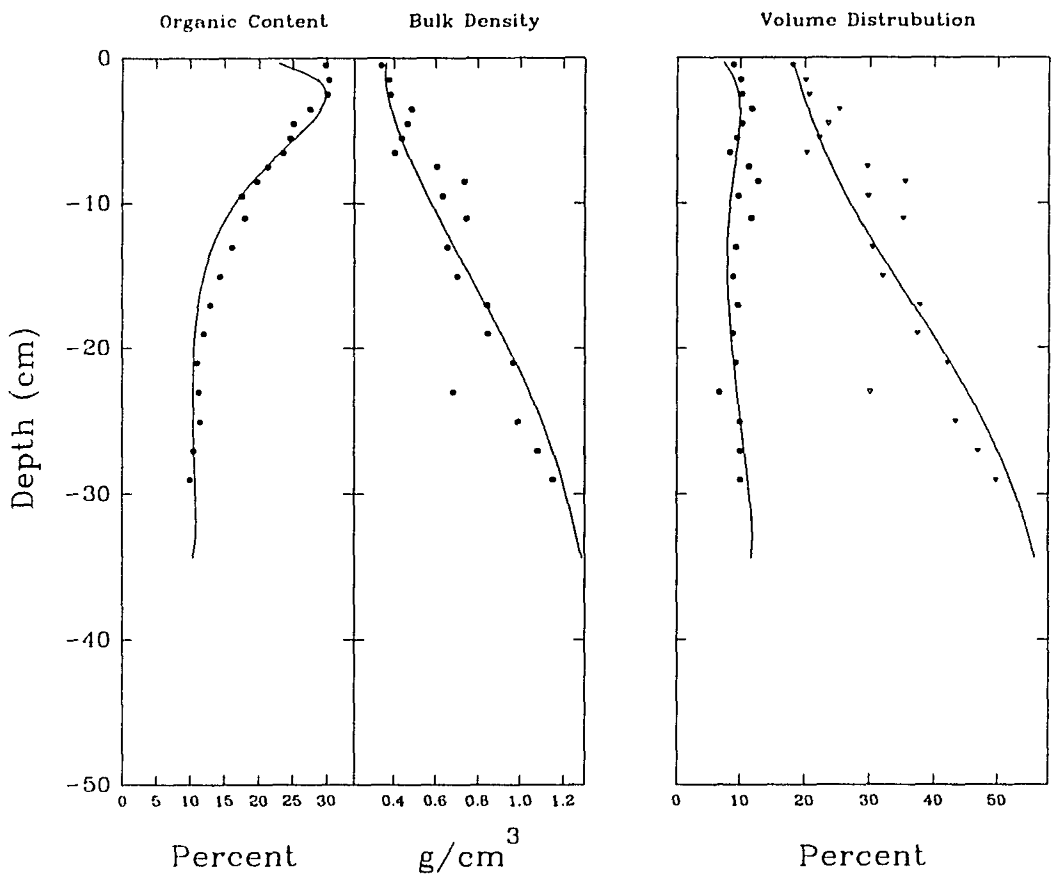


Figure 4.5. Depth profiles of organic content (left), bulk density (center), and volume distributions from the actual core (SYMBOLS) and as simulated using the model (LINES) for core from the high marsh at Stiffkey Marsh, UK.

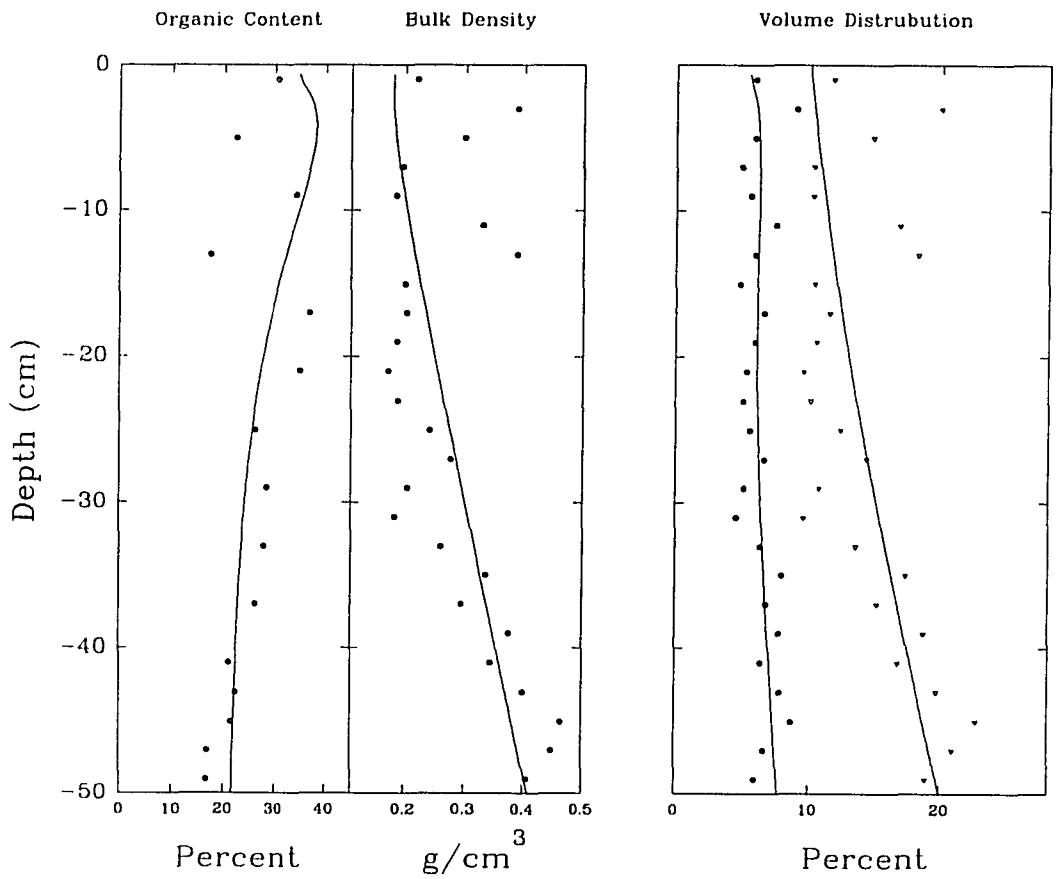


Figure 4.6. Depth profiles of organic content (left), bulk density (center), and volume distributions from the actual core (SYMBOLS) and as simulated using the model (LINES) for core from the mid-marsh at Biloxi, Mississippi.

analysis, to make predictions for the future, and to identify subjects/areas which need future study and additional data. The differences between the two calibrations and some issues that came up with the calibration of the Biloxi core are considered in the discussion below.

Sensitivity analysis

The impact of changes to the various parameters in the model on overall accretion rates, varied from almost negligible (decay constants for decomposition rates) to order of magnitude changes in accretion rates (initial pore space values). Table 4.2 indicates the range of changes for each of the parameters, and the implications of some of these changes are covered in more detail in the discussion.

First, if we consider short-term accretion rates, the following parameters had little impact (i.e., less than 20 percent change in the five year accretion rate due to the changes in the parameter of interest, ranging from a 90 percent decrease to a 200 percent increase):

decay constants for decomposition rate curves (kdec1, kdec2, kdec3), maximum decomposition rates for year 1 and 2 (mx1, mx2), compaction curve constant (k1), and the percent of refractory organic matter (refrac). Most of these variables affected decomposition rates. Changes in the decay constant for below-ground organic matter production (kdist), the final pore space (porelim), the total surface organic matter deposition (orgin), tidal range (tdrnge), and the maximum decomposition rates for year 3 (mx3) resulted in changes from 20 to 50 percent in short-term accretion rates. The most important variables affecting short-term accretion rates (causing impacts

Table 4.2. Results from sensitivity analysis for parameters in the simulation model. See Table 4.1 for an explanation of the variable names. Results are for percent changes in the approriate accretion rate (5 year or 100 year) relative to the "standard" run of the model as calibrated for Stiffkey Marsh, UK.

Parameter Name		5 yr. Max Increase	5 yr. Max Decrease	100 yr. Max Increase	100 yr. Max Decrease
kdist		22.1	-25.7	0.3	-17.0
kdec1		0.8	-0.3	1.1	-0.7
kdec2		0.6	-0.2	1.3	-0.8
mx2		7.3	-11.2	5.4	-7.6
mx1	**(+50/-90)	18.5	-7.3	6.9	-2.8
porelim	**(+100/-90)	6.0	-24.1	12.7	-14.4
orgin		40.0	-18.8	12.9	-5.4
k1		1.7	-11.3	13.9	-40.3
refrac		1.6	-3.8	15.5	-7.0
kdec3		0.9	-19.6	25.7	-7.3
tdrnge		30.5	-7.3	30.8	-6.3
mx3		0.0	-26.2	45.0	-11.4
minin		47.2	-62.5	55.8	-65.9
undpro		50.4	-21.2	65.4	-21.2
slr		73.8	-36.3	77.0	-34.8
h2oin	**(+20/-50)	638.1	-71.3	172.8	-51.6
intelv		145.6	-73.3	186.2	-79.0

** Values in parentheses indicate maximum increases and decreases for a given parameter in the sensitivity analysis. See text for the complete range of increases and decreases that was tested.

greater than 50 percent) were: rates of peak mineral matter deposition (minin), total below-ground production (undpro), and eustatic sea level rise (slr); initial pore space (h2oin), and the initial elevation (intelv) (Table 4.2). The largest impacts on short-term accretion rates were due to changes in parameters which affected the upper part of the sediment column or the relative elevation of the wetland.

The parameters that had very little impact (again, less than 20 percent change in accretion rates) on longer term accretion rates included: the decay constant for below-ground organic matter production (kdist), decay constants and maximum decomposition rates for year 1 and 2 (kdec1, kdec2, mx1, mx2), the final pore space (porelim), the total surface organic matter deposition (orgin), and the percent of refractory organic matter (refrac). Changes from 20 to 50 percent in long-term accretion rates were caused by changes in: compaction curve constant (k1), the decay constant and maximum decomposition rates for year 3 (kdec3 and mx3), and the tidal range (tdrnge). The largest changes in long-term accretion rates were caused by the same five variables that affected short-term accretion rates: peak mineral matter deposition (minin), total below-ground production (undpro), eustatic sea level rise (slr), inital pore space (h2oin), and the inital elevation (intelv) (Table 4.2). The variables that had little impact on short-term accretion rates but a moderate impact on long-term accretion rates were those that affected decomposition and compaction, indicating the potential importance of these factors in affecting the long-term stability of a wetland.

Because changes in accretion rates were not always linearly related to changes in a particular parameter, it is also valuable to evaluate the response graphs for each of the particular parameters versus 5, 30, and 100 year average accretion rates (Figures 4.7 and 4.8). Compaction and decomposition caused average accretion rates to decrease with an increasing period of measurement (5, 30, or 100 years) in almost all cases. The changes due to surface organic inputs (orgin) were greatest on five year accretion rates, whereas underground production (undpro) had a large effect on both 5 and 30 year accretion rates (Figure 4.7). The relationships for the maximum decomposition rates and decay constants were usually very flat (mx1, mx3, and kdec3 in Figure 4.7); however, there was a decrease in the 5 year accretion rate at very small levels of mx3. This was due to the fact that so little organic material decomposed that the marsh built up to a very high level, and mineral inputs decreased due to the change in elevation.

Figure 4.8 illustrates some of the parameters that had a large impact on vertical accretion rates. Note that there is a change in scale on the y-axis on this figure. Changes to the final pore space (porelim) had little impact on 30 and 100 years rates, and a slightly larger impact on 5 year accretion rates. Increases in mineral matter deposition (minin) had a large impact over the entire range of change, with an asymptotic relationship with large increases in mineral inputs. This is due to the fact that with large mineral inputs, the marsh built up to a high enough elevation that the response dropped off. Eustatic sea level rise (slr) had a large impact throughout. Increases in sea level decreased the relative elevation of the wetland, and

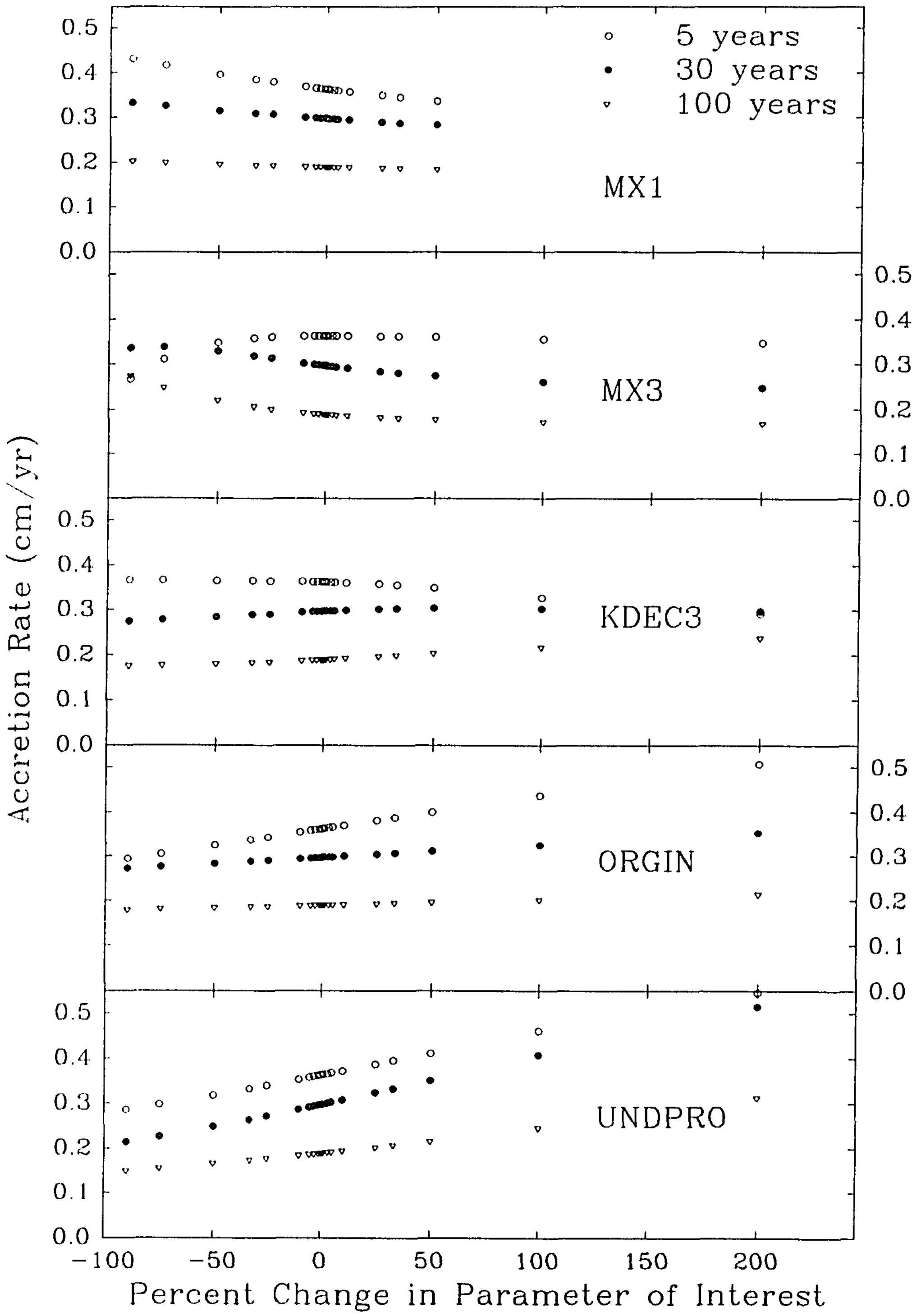


Figure 4.7. Results from sensitivity analysis for below-ground organic production (UNDPRO), surface organic input (ORGIN), decay constant for decomposition of the oldest organic matter (KDEC3), and maximum decomposition rates for the oldest and newest organic matter (MX3 and MX1). Accretion rates are average values relative to the time period indicated.

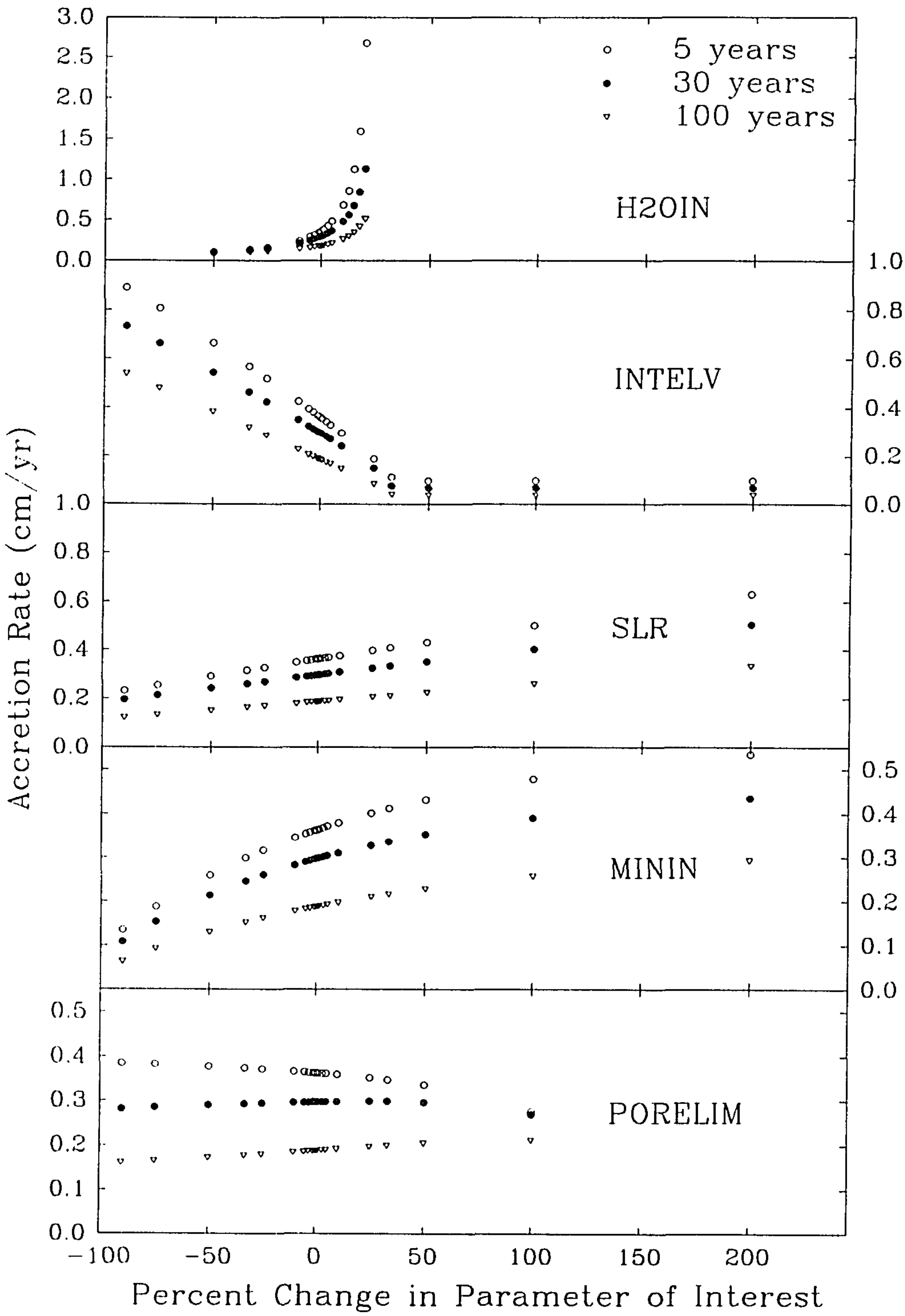


Figure 4.8. Results from sensitivity analysis for final pore space (PORELIM), mineral matter deposition (MININ), eustatic sea level rise (SLR), initial elevation (INTELV), and initial pore space (H2OIN). Accretion rates are average values relative to the time period indicated.

allowed for more material to accumulate on the marsh surface. Subsidence would have an identical effect in this model. The initial elevation of the sediment surface (intelv) had a non-linear relationship, with a threshold value above which there is very little input of mineral matter and accretion rates are very low. Above this threshold, the relative elevation of marsh is so high that mineral inputs are extremely low or zero. Below this threshold the relationship is linear. The initial pore space value (h2oin) is extremely important because so much of the sediment is pore space. Very slight changes in the initial pore space caused order of magnitude changes in accretion rates. Short term rates were particularly affected by these changes.

DISCUSSION

Calibration

Differences in the calibration of the model to the two cores indicate some of the differences in processes from these two areas. First, surface organic matter deposition and below-ground organic matter production rates were assumed to be higher in Biloxi Bay because of lower latitude and higher temperatures, as has been shown from published data (Turner 1976). Similarly, decomposition rates for all three classes of organic matter were assumed to be higher for the Biloxi core. Lower percentages of refractory organic matter in Biloxi were used in order to obtain organic content values at depth. Mineral input was greater in Biloxi since measured rates of mineral accretion were higher for this core. The rate of compaction was less in the calibration for the Biloxi core (both the total amount of compaction, i.e.

porelim, and the curve was shallower - i.e., k1) because there was less of an increase in bulk density in this core. Compaction may be affected by increases in tidal range because of the better drainage and increased consolidation in wetlands with large tidal ranges.

Additionally, the marsh at Biloxi Bay, like other Gulf coast marshes, is often subject to large storms, and storm inputs of sediments are probably extremely important to mineral accumulation rates along the Gulf coast (Reed 1989, Rejmanek et al. 1988). These storm inputs can be seen as sharp increases in sediment bulk density at 3-5 and 11-13 cms in the actual data for the Biloxi core (Figure 4.6). The increases in bulk density are probably due to high mineral content and coarser sediments from the storm deposition. The model does not attempt to simulate infrequent storm inputs, and as a result, the model's mineral sediment accumulation rates are lower than were actually measured for the core (744 vs. 1136 $g/m^2/yr$ for the model and the actual data, respectively). However, if we assume that the increase in bulk density is entirely attributable to storm deposits and subtract the mineral inputs from storm deposits by smoothing the bulk density curve for the Biloxi core, the measured rates would be 702 $g/m^2/yr$.

Differences in the distribution of organic production with depth were also apparent between the two sites. The peak in organic content was at a shallower depth in the Stiffkey core (Figure 4.5), indicating that below-ground production is mostly occurring close to the surface in this core. I had difficulty simulating the gradual increase in organic content to approximately 20 cm for the Biloxi core with an

exponential decrease in below-ground organic matter production (Figure 4.6). Very little data exist on the relative production of below-ground organic matter with depth (Gallagher and Plumley 1979), and it is possible that a linear decrease in production or some other distribution may better describe actual production trends. However, without additional field data on actual production values it is impossible to fully address this question.

Finally, the only way to fit the model for the Biloxi core was to increase subsidence to 0.075 cm/yr greater than is reported for Biloxi (reported relative sea level rise is 0.15 cm/yr from Penland and Ramsey 1990). I found that given the small tidal range (0.55 m), no matter how other parameters were set, the wetland would accumulate material so rapidly that it would quickly build into the extreme upper part of the tidal range if I used the accretion rate from the 1963 ^{137}Cs peak. It is very possible, that subsidence in the marsh has been higher than reported from tide gage studies because of additional compaction and decomposition in deeper marsh sediments. The marsh at Biloxi has probably developed for much more than 300 years, and the deep sediments that have accumulated will continue to decompose and compact, contributing to overall subsidence rates within the wetland. Turner (1991) has identified this problem in using tide gauge data to estimate sea level rise for marsh accretion balance studies.

Important model parameters

The results from the sensitivity analyses indicate which factors are important in controlling wetland accretion processes in the model. However, as always the

model is built on a series of assumptions and available data, so the output will reflect the assumptions that are an inherent part of this model. Still the model can help to focus attention on potentially important process, and it allows us to formalize many assumptions that are always made but often not addressed.

Previous work has documented the importance of elevation in determining accretion rates (Pethick 1981, Hatton et al. 1983, Letzsch 1983, Stoddart et al. 1989, Chapter 2), but it is not clear from these field studies whether mineral or organic matter accumulation are more important in causing this effect. It is likely that this could change depending on the characteristics of a particular marsh. Sediments from marshes along the Norfolk coast in Great Britain, such as Stiffkey Marsh, have very high mineral content and high bulk density, and it has been shown that differences in mineral inputs are very important in affecting vertical accretion rates (Stoddart et al. 1989, French and Spencer 1993). However along the Gulf coast, coastal wetland sediments are typically more organic, and organic accumulation is more important than mineral accumulation in controlling accretion rates (Hatton et al. 1983, Nyman et al. 1993, Chapter 2). The effect of elevation in the model was primarily through mineral matter inputs, which decreased linearly versus relative elevation. However, organic matter also decreased because total organic matter production was a function of total sediment volume, and increases in mineral deposition caused increases in sediment volume.

Decomposition rates did not have a large impact on accretion rates. Very little work has been done to try to identify the long-term rates of decomposition or to

evaluate changes in decomposition rates with depth (Hemminga et al. 1988). Because of that these variables were model calibrated. In coastal wetlands that are developing over deep sediments of older wetlands, long-term rates of decomposition are probably important in controlling compaction rates. As the depth of the sediment column increases, the importance of this decomposition increases because it will be integrated throughout the entire sediment column. Some of the other factors affecting organic accumulation had relatively large impacts on accretion rates. Surface organic inputs had a large impact on short-term rates, and below-ground organic production had impacts on 5 and 30 year accretion rates. However, there are few data available on the relative production of below-ground organic matter with depth (Gallagher and Plumley 1979). As mentioned above, it is likely that this parameter may change between different coastal areas.

Sediment structure

Pore space was the most important factor affecting the overall accretion rates in the model, because of the large amount of sediment volume that it occupied. Very slight increases in the initial pore space values caused order of magnitude differences in accretion rates (Figure 4.8). Unfortunately, the modelling of changes in pore space was done in a very simple manner. I did not include any information on the importance of sediment structure in the model because so little data were available on these processes. Most studies of compaction of organic rich sediments have been done with fresh water peats and are concerned with engineering properties of compaction due to applied loads, not natural compaction processes (Landva and

Pheeney 1980, Hobbs 1986). It is clear that natural peat compaction occurs over time and is an important contributor to overall subsidence rates, (Bloom 1964, Kaye and Barghoorn 1964), but detailed studies of sediment structure are lacking.

It is not clear which sediment component (organic or mineral) is more important in providing structure. Bricker-Urso et al. (1989) indicated that pore space appears to be a function of organic content, and correlations of vertical rates and organic accumulation rates in Gulf coast marshes support this (Hatton et al. 1983, Chapter 2). However, it is not clear how this relationship may change across sediments with different organic contents. Kaye and Barghoorn (1964) found that compaction occurred more quickly in mineral soils than in peats. In order to more fully understand the processes occurring in these coastal wetlands sediments, we need to get to the mechanisms that control sediment structure. Many important questions remain unanswered at present: What controls the initial volume of sediment pore space? Is the organic framework of the sediment collapsing, or are clay layers being compressed? Is the sediment losing strength due to the loading of more sediment on the surface or is the organic framework being weakened through slow decomposition. At what depth in the sediment column does all of this compaction occur? Hopefully, in the future many of these questions can be answered through laboratory studies of sediment strength, compressibility and structure, using techniques that have been applied in the past to technical examinations of peats (Landva and Pheeney 1980, Hobbs 1986).

Effects of sea level rise

The influence of sea level rise on accretion rates is analogous to lowering the elevation of the marsh because it is the elevation of the wetland surface relative to water level that determines mineral accumulation rates in the model. I did not address the issue of proximity to a given sediment source, which has been shown to be important for reasons similar to elevation (French and Spencer 1993). Because I am modelling the dynamics of a single core/location, this factor would not change for a given calibration, but the mineral matter deposition (minin) could be changed for a given core. In the future, a potential use of the model could be linking up a series of models across a landscape, and this would be an interesting part of such a landscape model.

In order to examine potential impacts of sea level rise on the stability of the wetland, I evaluated the relative elevation of the wetland surface and the predicted accretion rates under a series of different sea level rises. The rate of sea level rise was constant over the entire 300 year run of the model and varied from 0.05 to 0.5 cm/yr, in 0.5 cm/yr increments. Salt marsh vegetation at Stiffkey has a range of about 1 to 1.2 m, and assuming that our sample was collected in the upper section of the marsh, this site could withstand a decrease in relative elevation of about 1 m before conversion to mud flat would occur (French 1993).

In all cases of sea level rise that were tested (0.5 to 5.0 cm/yr), the marsh maintained its elevation within 1 m of the original elevation over the 300 year run (Figure 4.9). However, at a rate of 0.5 cm/yr the relative elevation was getting close

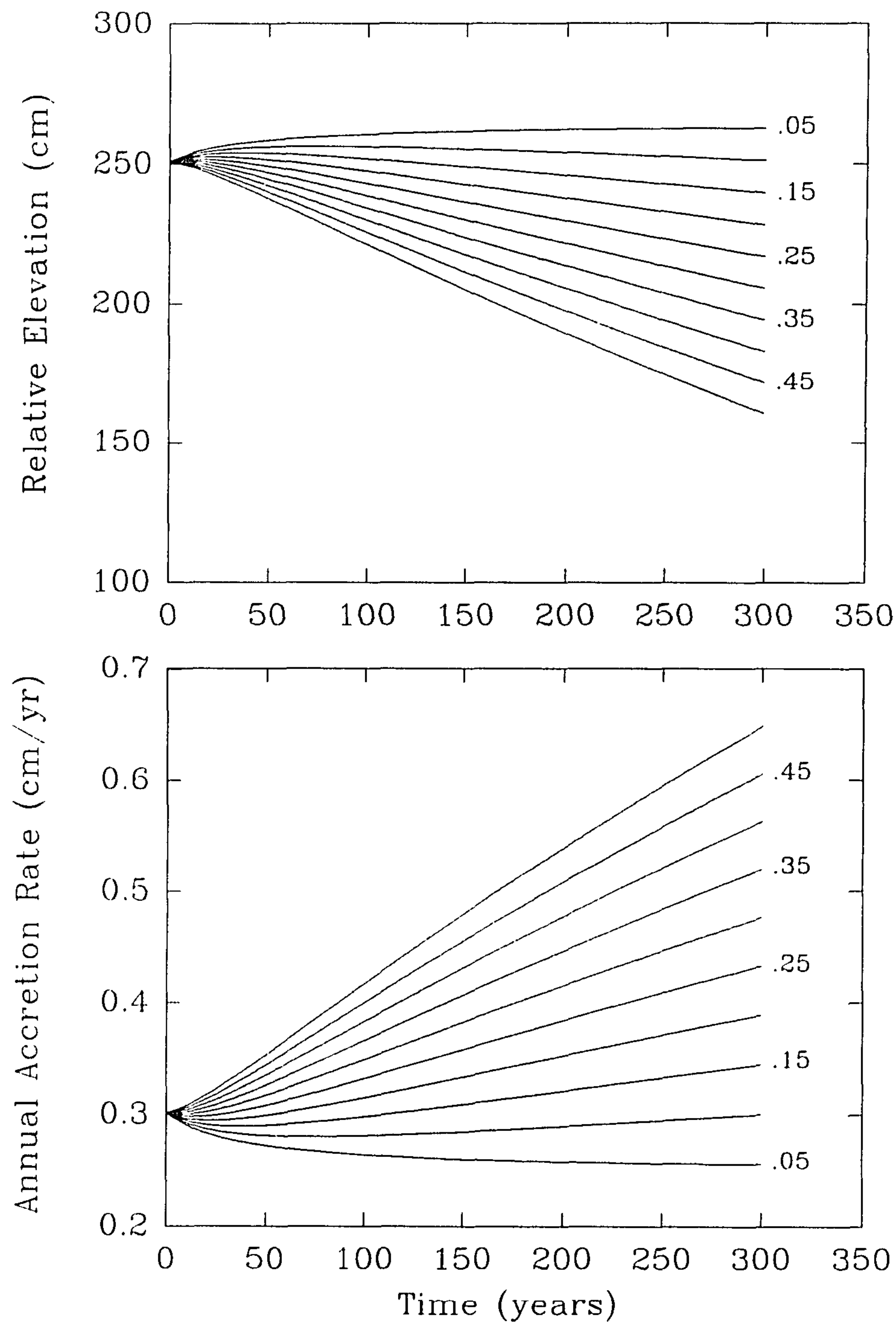


Figure 4.9. Predicted relative elevation of wetland sediment surface under a series of different sea level rise scenarios, ranging from 0.05 to 0.5 cm/yr for Stiffkey Marsh (TOP), and predicted rates of vertical sediment accretion under the same series of sea level rise scenarios (BOTTOM). Rates of sea level rise for each run are given to the right of every other line.

to this threshold level. Rates higher than 0.5 cm/yr for this time period could lead to complete loss of marsh. French (1993) found similar results using a model which focused on mineral accumulation for a nearby marsh in Norfolk. An important impact of increase rates of sea level rise would be the conversion of the location that was sampled from high marsh to low marsh, based on its projected relative elevation (French 1993, Warren and Niering 1993). Additionally, it should be noted that sites lower in the marsh would probably be converted to tidal mud flats much earlier than these high marsh sites.

The predicted rates of annual vertical accretion (Figure 4.9) are within the range of short term accretion rates that have been measured in this area (French and Spencer 1993, Stoddart et al. 1989). However, these high values are only found in the low part of the marsh. Larger increases in sea level rise could result in rates of accretion that would be too high to maintain in this area. French (1993) found similar results, with drowning and excessive accretion rates occurring at 1.0 cm/yr rate of sea level rise, but a sustainable marsh (with conversion to lower elevations) at 0.4 cm/yr. This site has a very large tidal range and wide distribution of salt marsh vegetation, in terms of relative elevation. Locations with smaller tidal ranges will have a smaller distribution of vegetation (McKee and Patrick 1988), and could be more susceptible to increases in sea level rise, because of the smaller elevational range of vegetation.

Future studies

In addition to addressing the mechanics of changes in pore space and compaction rates with depth, as discussed above, there are some important additions

that could be made to the model that would make it more realistic. However, additional data are needed before most of these changes could be made. These modifications include:

1) Additional data are needed for more long-term accretion rates, along with deeper profiles of organic content and bulk density. These data would put much better constraints on below-ground processes in the model. At present, I can only calibrate the model based on 30 year accretion rates. If a series of accretion rates over varying time scales were available for the same area, we could get a much better picture of the importance of decomposition and compaction. Also, detailed studies of variations in production and decomposition rates (especially long-term decomposition rates - i.e., kdec3 in the model) versus depth are needed. These studies would allow us to use actual rates to try to predict the observed organic content and bulk density profiles rather than relying on model-calibrated values.

2) Storm-related sediment inputs could be added to the model. Using data from the historic occurrence of storm events for a given area, these could be added to the model using a simple probability function. This input could have two components: a) the probability of a storm occurring and b) the intensity of a storm when one occurs. Both of these parameters could be generated from a particular distribution, such as the normal or poisson distribution, depending on the data for the area being modelled.

3) Both below-ground organic matter production and surface organic matter deposition could be related to sediment elevation. Wiegert et al. (1983) have shown a

significant increase in above-ground production with increased drainage and soil water movement. Other studies have also documented trends in production across the marsh (DeLaune et al. 1979). However, little data are available for the effects of elevation on below-ground production, and this is the most important component of organic production in the model. This relationship may be important in controlling vertical accretion rates across the marsh.

The simulation model presented can accurately simulate sediment processes over time, and helps to identify processes that are important in affecting accretion rates. The use of both accretion rates and sediment characteristics to calibrate the model make it more realistic than other sediment accretion models that have been developed previously. In addition, the model is useful in predicting the fate of marshes in the future, given potential changes in rates of sea level rise. Some of the processes that have been identified in the model need additional research in order to make the model more realistic, including: sediment structure and pore space, below-ground production profiles, and long term accretion rates.

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CHAPTER 5

HEAVY METAL CHRONOLOGIES IN SELECTED COASTAL WETLAND SEDIMENTS FROM NORTHERN EUROPE

INTRODUCTION

Present anthropogenic inputs of heavy metals into the environment far exceed natural inputs (Nriagu and Pacyna 1988), and these inputs may pose large health risks in areas where metals accumulate. The impact of heavy metal pollution to coastal and estuarine areas could be substantial because of a variety of inputs to these areas. Sources include riverine inputs, local runoff, atmospheric deposition, and coastal waters. Local runoff and riverine inputs can carry municipal sewage (Galloway 1979) and industrial effluent (Gross 1978). Both of these inputs have been shown to carry high levels of heavy metals in many areas (Helz 1976). Coastal waters are usually relatively clean and probably do not have a major effect on heavy metal concentrations in most estuarine situations. Recent interest in protecting coastal areas and waters has sharply decreased heavy metal concentrations in coastal waters and increased monitoring of pollutants in these environments (O'Connor and Ehler 1991, Valette-Silver et al. 1993).

In addition to monitoring present inputs of pollutants into coastal areas, there is great interest in determining historic inputs and accumulation rates in coastal environments. Sediments have been used extensively as indicators of chronological inputs to coastal areas, including both subtidal sediments (Santschi et al. 1984, Schmidt and Reimers 1991, Swartz et al. 1991, Valette-Silver 1993) and wetland

sediments (Griffin et al. 1989, Bricker 1993). Recent studies in coastal wetlands have successfully developed chronologies of metal inputs (McCaffrey and Thomson 1980, Bricker 1993, Zwolsman et al. 1993). One of the consistent findings of many of these studies is the recent decrease in heavy metal concentrations in sediments (Macdonald 1991), especially for lead (Evans 1982, Trefry et al. 1985, Dörr et al. 1991). These results have also been confirmed by observed decreases in metal concentrationss in snow deposits from Greenland (Boutron et al. 1991).

Salt marshes are sinks for sediments and other materials, and they are excellent areas to study the pollution chronology of coastal and estuarine systems because usually there is little bioturbation of salt marsh sediments (Stumpf 1983). In studies of areas where data are available on historic inputs to the coastal environment, the chronology that is found in the sediments usually closely matches historical inputs (McCaffrey and Thomson 1980, Bricker 1993, Zwolsman 1993). However, in many areas, very little information exists on histories of metal inputs, and in these cases, we must rely upon sediment chronologies to attempt to reconstruct the history of pollution for a given area (Alongi et al. 1991).

As part of a study establishing accretion rates and evaluating the use of ^{137}Cs from the Chernobyl accident for sediment dating, I collected cores from five areas in northern Europe, and I have used the chronologies established from the ^{137}Cs dating to project sediment concentrations of Cu, Cd, Cr, Pb, and Zn over the last 100 years. Most other studies that have evaluated heavy metal chronologies in wetland sediments have only used one core from a given wetland (McCaffrey and Thomson 1980,

Bricker 1993, Zwolsman et al. 1994), with the exception of one of the areas that Bricker (1993) sampled in Rhode Island. The cores that I collected were from the upper and lower marsh, areas that had different vertical accretion rates and different hydrological and biogeochemical regimes (i.e., different flooding patterns). By comparing chronologies from the two cores at a site, we can evaluate the mobility of metals within the sediments. Because the cores were from different relative elevations and flooding regimes it is highly unlikely that diagenetic processes would result in similar chronologies for the two cores. In addition, differences in relative concentrations of metals between the two cores may give some evidence of the relative importance of atmospheric versus riverine or estuarine inputs of pollutants.

METHODS

Samples were collected from the upper and lower marsh in the following areas (Figure 5.1): Wistula River, Poland; Oder River, Poland; St. Annaland Marsh (Eastern Schelde), Netherlands; Dengie Marsh, UK; Stiffkey Marsh, UK. For a more detailed description of each site, see Chapter 2.

Core collection and analyses

At each site, two cores were collected for analysis, one from the low marsh and one from a mid to high marsh area. I collected samples from low marsh sites that were at least 10 M away from the lowest edge of the marsh (avoiding a stream-side effect) and were located in vegetation typical of the "low marsh" of that

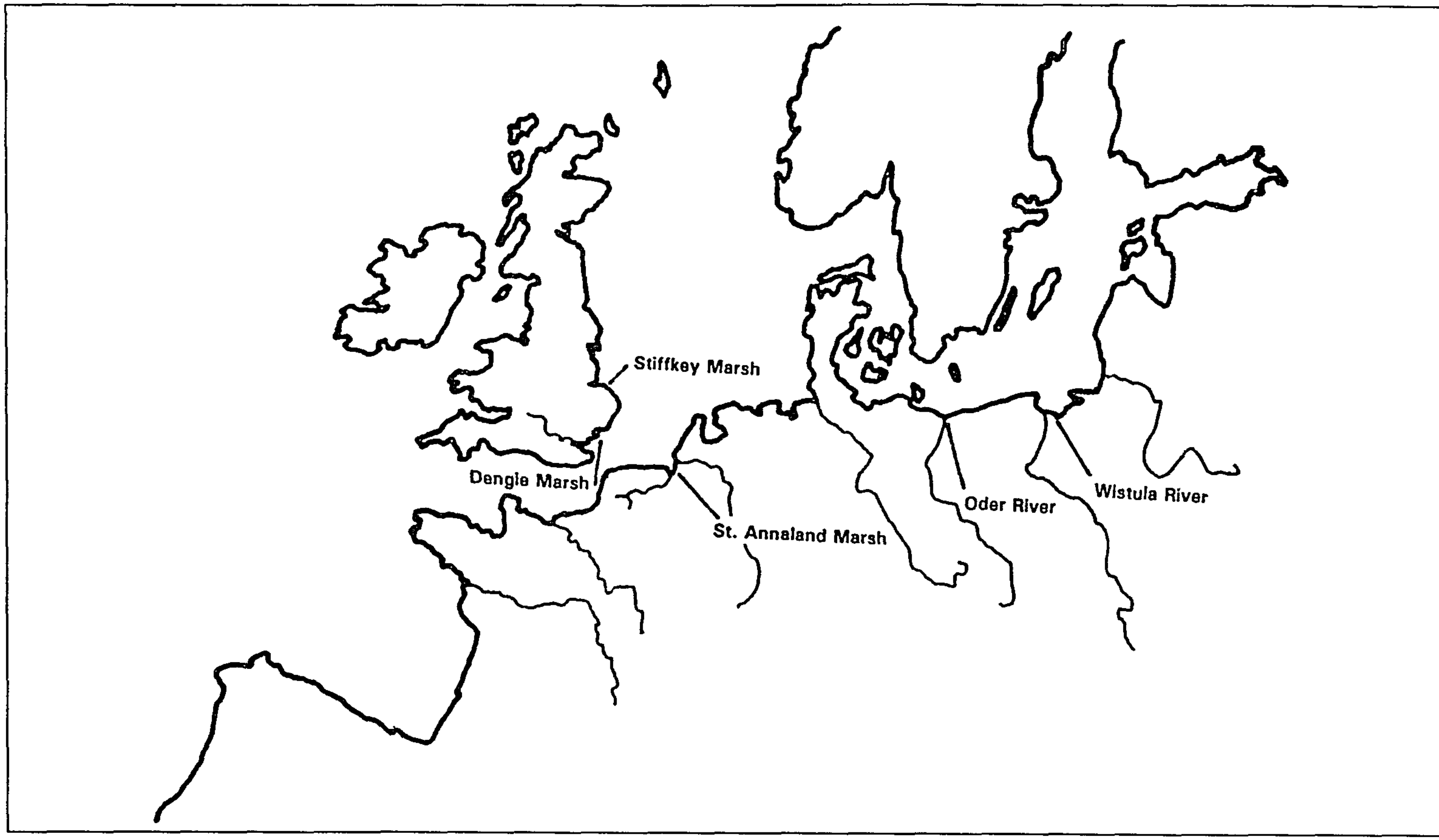

Figure 5.1. Location of sampling sites along northern European coastline.

particular area, usually either *Spartina anglica* or *Phragmites*. High marsh sites were located in the same marsh but at higher elevations.

Cores were collected in 15 cm diam. aluminum cylinders to a depth of 50 cm. In all cases compaction was less than 2 cm throughout the 50 cm core. Cores were extruded with a wooden plunger and were sectioned every 1 cm from the top 10 cm and every 2 cm from 10 to 50 cm. One cm sections were used in order to give better precision in locating the Chernobyl ^{137}Cs peak. Individual sections were placed into ziplock bags, weighed wet, dried at 80°C, weighed dry, and crushed with a mortar and pestle. ^{137}Cs activity of the bulk sediment was counted with a Lithium Drifted Geranium detector and multi-channel analyzer.

^{137}Cs is a product of nuclear weapons testing and does not occur naturally. Significant levels of this isotope first appeared in the atmosphere in the early 1950's with the peak quantities detected in 1963. Average vertical accretion rates were calculated using the 1963 peak. I also found ^{137}Cs peaks from the Chernobyl accident in 1986; however, only rates based on the 1963 peak were used for this part of the study.

Duplicate samples of approximately 1 g each from every other section of the core were analyzed for metals. Samples were digested in 5 ml of hot nitric acid (150°C) for three hours, and then evaporated to a volume of approx. 1 ml. Samples were diluted with ultra-pure water to 50 ml. The supernatant fluid was analyzed using a Jarrell-Ash ICAP for the metals of interest (Cd, Cu, Cr, Fe, Mn, Ni, Pb, and Zn). Dilutions (1:10 and 1:100) of most samples were run in order to measure Fe

and Mn in their linear range. Metal concentrations were calculated relative to the mineral matter weight of each sample, using the sample weight and the mineral and organic content values that were determined from Chapter 2.

Concentrations were then converted to time based profiles, using vertical accretion rates obtained from ^{137}Cs dating. Also, correlations between metal concentrations were tested for the entire data set, and for individual cores. Statistical analysis was completed using SAS.

RESULTS AND DISCUSSION

Fe and Mn profiles

In most of the cores, both Fe and Mn profiles were typical of total Fe and Mn concentrations in sediments under flooded conditions (Santschi et al. 1990), with the highest values in the surface oxidized zone and decreasing concentrations with depth (Figures 5.2 and 5.3). Fe profiles were more variable than Mn profiles, and Fe concentrations remained higher to a greater depth than Mn in most of the cores, as expected given the redox chemistry of Fe and Mn (Gambrell and Patrick 1978). In some of the cores (low marsh cores from the Wistula River, St. Annaland, and Stiffkey Marsh, and both cores from Dengie Marsh) Fe concentrations increased in the bottom 5-10 cm of the core. It is not clear why Fe concentrations increased in these deep samples; however, Bricker (1993) also found similar increases in Fe and Mn at depth in cores from Rhode Island salt marshes. It is possible that the increases could be due to textural differences throughout the cores.

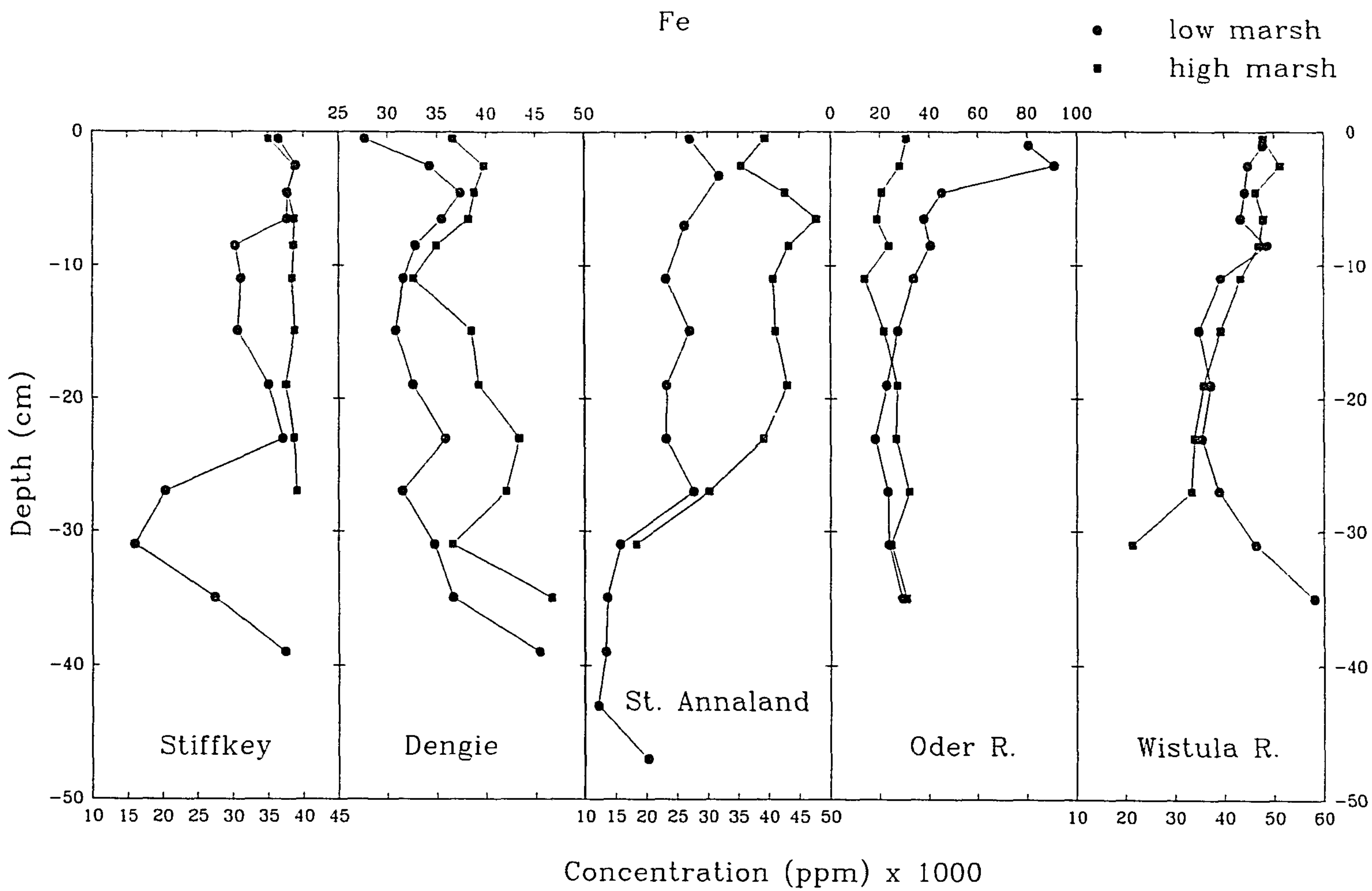

Figure 5.2. Depth profiles of Fe concentrations from the five sampling sites. Each panel includes data for low and high marsh samples.

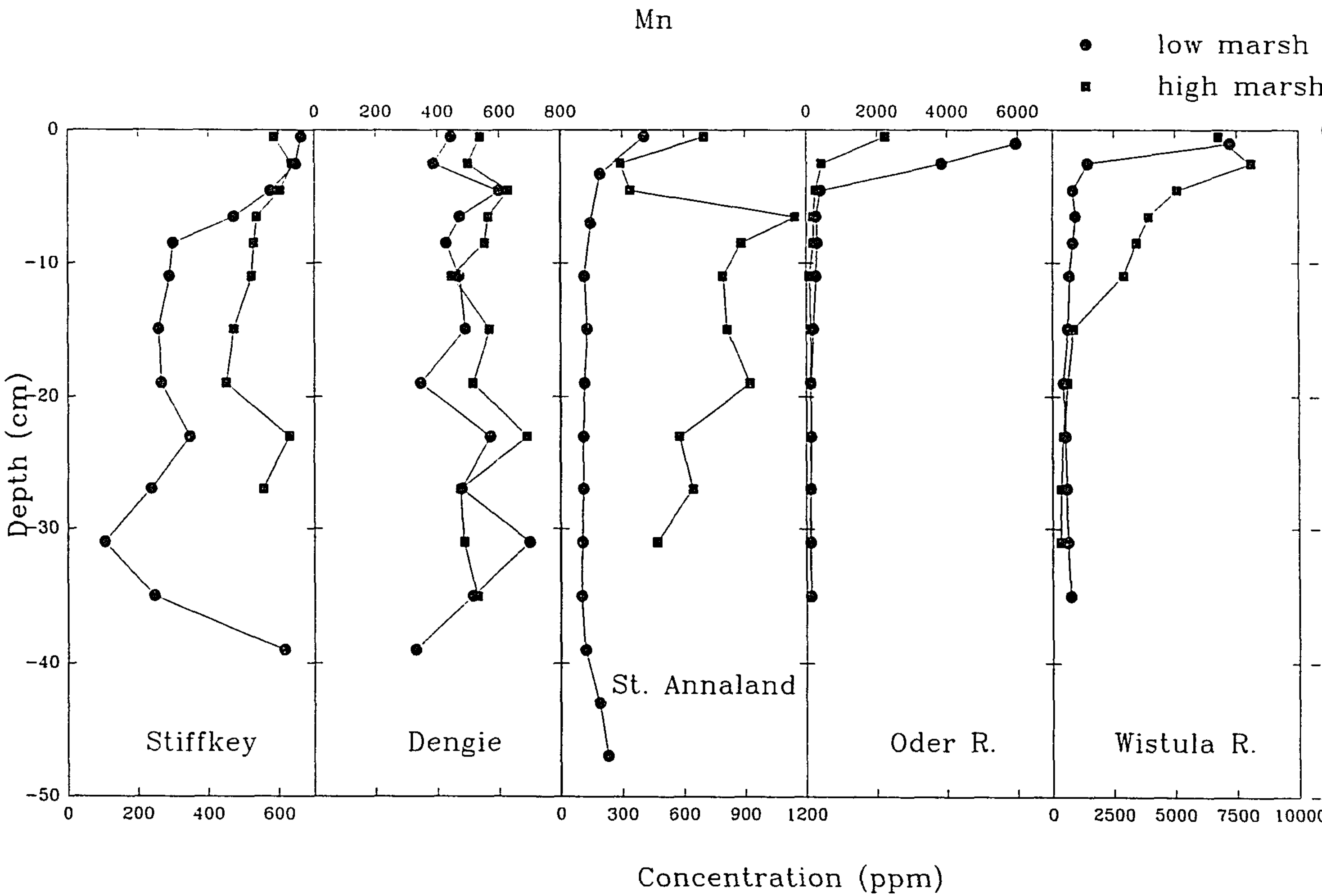

Figure 5.3. Depth profiles of Mn concentrations from the five sampling sites. Each panel includes data for low and high marsh samples.

Fe concentrations dropped from high values of 3-5 percent to low values of 1 to 2 percent in the range of 20 to 30 cm in cores from the low marsh at Stiffkey and in both cores from St. Annaland (Figure 5.2). Values remained high throughout the core from the high marsh at Stiffkey, indicating that this short core did not go deep enough to reach the area where Fe is reduced and depleted in the sediment column. There was a sharp drop in Fe concentrations near the surface (5 cm) of the core from the low marsh at the Oder River; whereas, the high marsh core from this site did not show any trend with depth. Both low and high marsh cores from the Wistula River had a gradual decline in Fe concentrations until approx. 25 cm. Below this depth, Fe concentrations dropped sharply in the high marsh core and increased sharply in the low marsh core.

The core from the low marsh at Stiffkey had a drop in Mn concentrations at approx. 8 cm, and this core showed a large increase in Mn in the bottom 10 cm of the core, similar to the trend in Fe concentrations for this sample. Mn concentrations from the high marsh at Stiffkey, gradually decreased in the top 20 cm of the core but also increased near the bottom of the core. Mn profiles from the low marsh at St. Annaland had a sharp drop in the upper 5 cm of the core; however, the sample from the high marsh showed a highly variable profile, with a sharp drop near the surface, a large increase just below this, and then a gradual decline with depth. The reason for this type of profile is unclear. The four cores from the Polish coastal wetlands all showed very distinct drops in Mn concentrations with depth. The zone of Mn enrichment varied from approx. 5 to 15 cm in these cores. Upper marsh cores had

greater depths of Mn enrichment in most cases, indicating the greater degree of aeration and drainage in these cores (Figure 5.3).

Profiles of both Fe and Mn from Dengie Marsh were very different and did not have higher concentrations near the surface. Manganese concentrations were relatively constant with depth, and Fe concentrations did not show a consistent trend. It is possible that this could be due to differences in clay content with depth in these sediments. However, the Fe and Mn profiles from the other areas, along with the ^{137}Cs and ^{210}Pb profiles, indicate that these cores were physically undisturbed prior to sampling.

Cd, Cr, Cu, Ni, Pb, and Zn concentrations

There was a very high degree of correlation between concentrations of the six metals of interest (Cd, Cr, Cu, Ni, Pb, and Zn) for all samples grouped together. This can be seen in the resemblance of the various profiles for the different metals from a particular site, as well as in the high Pearson correlation coefficients for these metals (Table 5.1). Pearson correlation coefficients for these six metals ranged from 0.69 to 0.93, using data for all cores, and were also high when the data were analyzed separately for the individual cores. Correlations were lowest between Ni and Pb (0.70), and between Cr and the other five metals (0.69 to 0.76). Values for the other metals were all greater than 0.81.

The high correlations between metals indicate that they are coming from similar sources and behave similarly chemically. Most other studies of metal distributions in intertidal and subtidal sediments have also found high inter-metal

Table 5.1. Pearson correlation coefficients and their probabilities for metal concentrations and organic content from all samples (n=143).

	ZN	CD	PB	CR	NI	FE	MN	ORG
CU	0.91465 0.0001	0.89028 0.0001	0.93293 0.0001	0.75903 0.0001	0.85205 0.0001	0.17905 0.0324	0.06541 0.4376	0.90440 0.0001
ZN		0.92918 0.0001	0.84024 0.0001	0.70593 0.0001	0.83642 0.0001	0.22320 0.0074	0.20056 0.0163	0.85892 0.0001
CD			0.81095 0.0001	0.69309 0.0001	0.82219 0.0001	0.33880 0.0001	0.26748 0.0012	0.83042 0.0001
PB				0.69926 0.0001	0.70363 0.0001	0.06569 0.4357	-0.05206 0.5369	0.88002 0.0001
CR					0.74215 0.0001	0.32825 0.0001	0.18658 0.0257	0.70278 0.0001
NI						0.51606 0.0001	0.35182 0.0001	0.85747 0.0001
FE							0.55842 0.0001	0.25455 0.0022
MN								0.21223 0.0109

correlations (Santschi 1984, Bricker 1993). Although individual metals may be associated with a specific industry or source (i.e., Pb from leaded fuel emissions, Zn from smelting, etc...), most anthropogenically influenced metals show very similar profiles in the sediments (Valette-Silver 1993).

Correlations for the six metals versus Fe and Mn were lower, with values ranging from 0.07 to 0.35, for all correlations except Ni and Fe, which was 0.52. The low correlations for the other metals indicate that the anthropogenically input metals behave differently than Fe and Mn, which are strongly influenced by redox conditions. Griffin et al. (1989) also found a higher correlation between Ni and Fe than for other metals; whereas, Zwolsman et al. (1993) indicated that the Ni was coprecipitating with Mn, not Fe, in salt marsh sediments from the Western Scheldt. The Pearson correlation coefficient (0.35) between Mn and Ni was significant but lower than between Ni and Fe; however, it does appear that Ni is associated with both of these metals and may reflect their distribution rather than anthropogenic enrichment. There was a strong relation between organic content and the metals, with Pearson correlation coefficients ranging from 0.70 to 0.90. Many previous studies have documented the correlation of metals with organic matter in the sediment (Vestergaard 1979, Fiejtel et al. 1988).

Average values of metals were consistently higher in the three surface samples versus the three bottom samples of each core, and there were peak values below the surface for most metals. Comparisons are given for Pb from all of the cores (Table 5.2), and similar values were found for the other metals. Samples from the Oder

Table 5.2. Relative Concentrations of Pb in cores. Concentrations given are for mean concentration from the 3 top and bottom sections from each core, and for the single highest concentration found throughout the core.

		Bottom Conc.	Top Conc. (ppm)	MAX Conc.
ENGLAND				
Stiffkey Marsh	low marsh	19.1	81.8	100.2
	high marsh	63.3	84.8	103.9
Dengie Marsh	low marsh	49.6	50.9	64.1
	high marsh	78.4	51.0	80.4
NETHERLANDS				
St. Annaland	low marsh	64.9	76.1	69.9
	high marsh	50.6	107.1	192.0
POLAND				
Oder River	low marsh	373.3	344.6	427.2
	high marsh	115.7	261.6	376.7
Wistula River	low marsh	51.3	63.0	69.9
	high marsh	39.7	48.2	60.7

River and St. Annaland showed the highest levels of surface accumulation of metals; whereas, both Wistula River and Dengie Marsh had very low increases.

Depth profiles of metal concentrations from the low and high marsh from each site had different shapes and peaks because of differences in accretion rates between the two areas within the marsh (Figures 5.4a, 5.4b, and 5.4c, top graphs). Accretion rates were lower in the high marsh (Chapter 2). When depth profiles were converted to time-based profiles, using accretion rates from ^{137}Cs dating, there was very good agreement between the two chronologies from each site. This agreement can be seen in the peaks in Cu at 25-30 years ago and very close to the surface in the cores from St. Annaland Marsh (Figure 5.4a). Zn concentrations remained high in the low marsh core from the Wistula River at depth, but decreased to much lower values in the high marsh core. This difference is due to the fact that the low marsh sample from this site had a very high accretion rate, and in fact these deep sediments are very young (Figure 5.4b). A similar fit between the chronologies from the cores from a particular area can be seen in the profiles of Zn from the Oder River site (Figure 5.4c). In this case, there is very good agreement in the peak at 35 years, as well as in the general increase that was found in the surface samples of the two cores.

These three examples of differences in the depth profiles and time chronologies indicate the type of relationships that were typical for most of the metal chronologies from the five marshes. The agreement between chronologies from the high and low marsh at each site is strong evidence that these profiles represent true chronologies of heavy metal inputs. If diagenetic processes were most important in

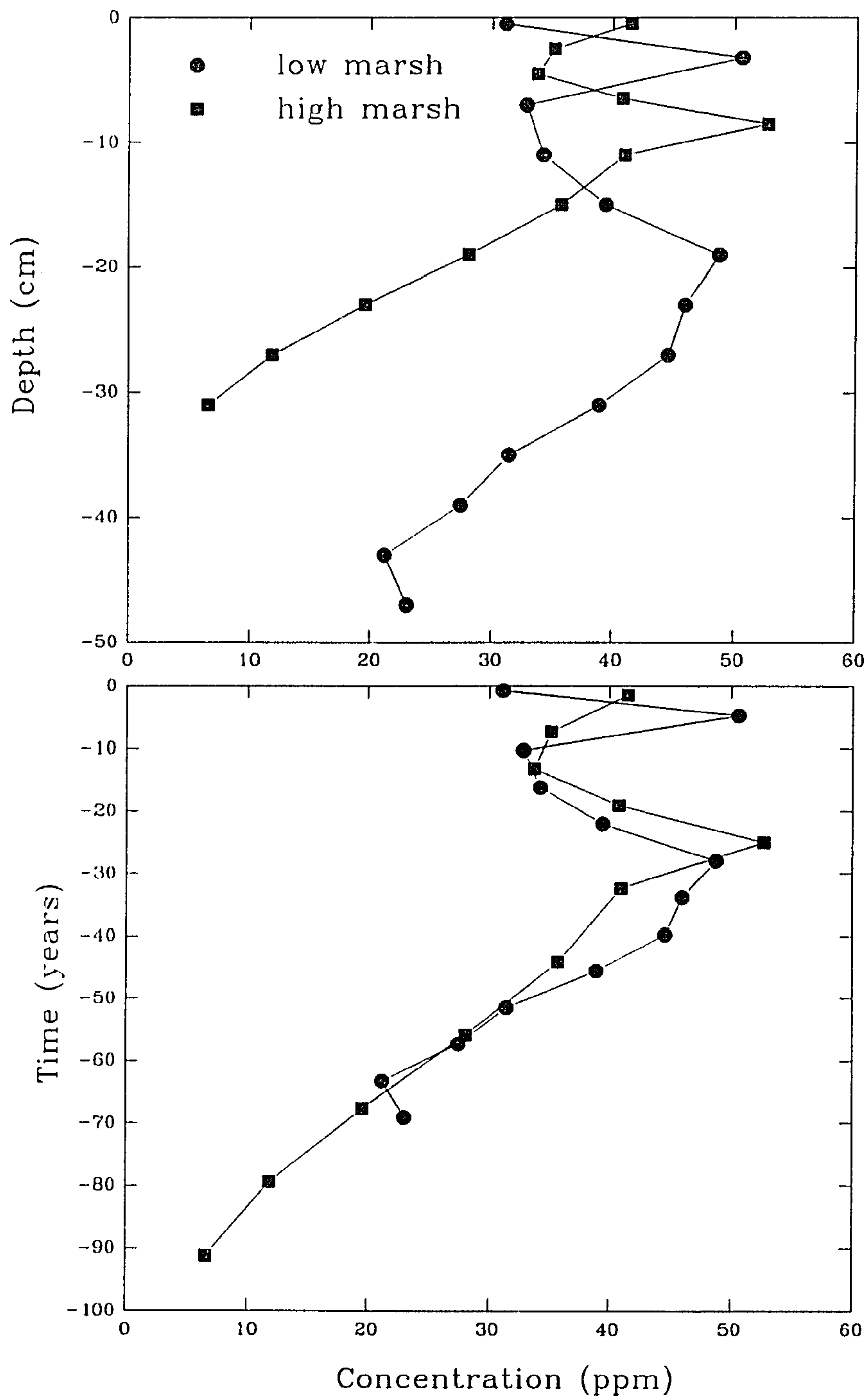

Figure 5.4a. Depth profiles of Cu from the high and low marsh at St. Annaland (top) and sediment chronologies of Cu for the same samples (bottom).

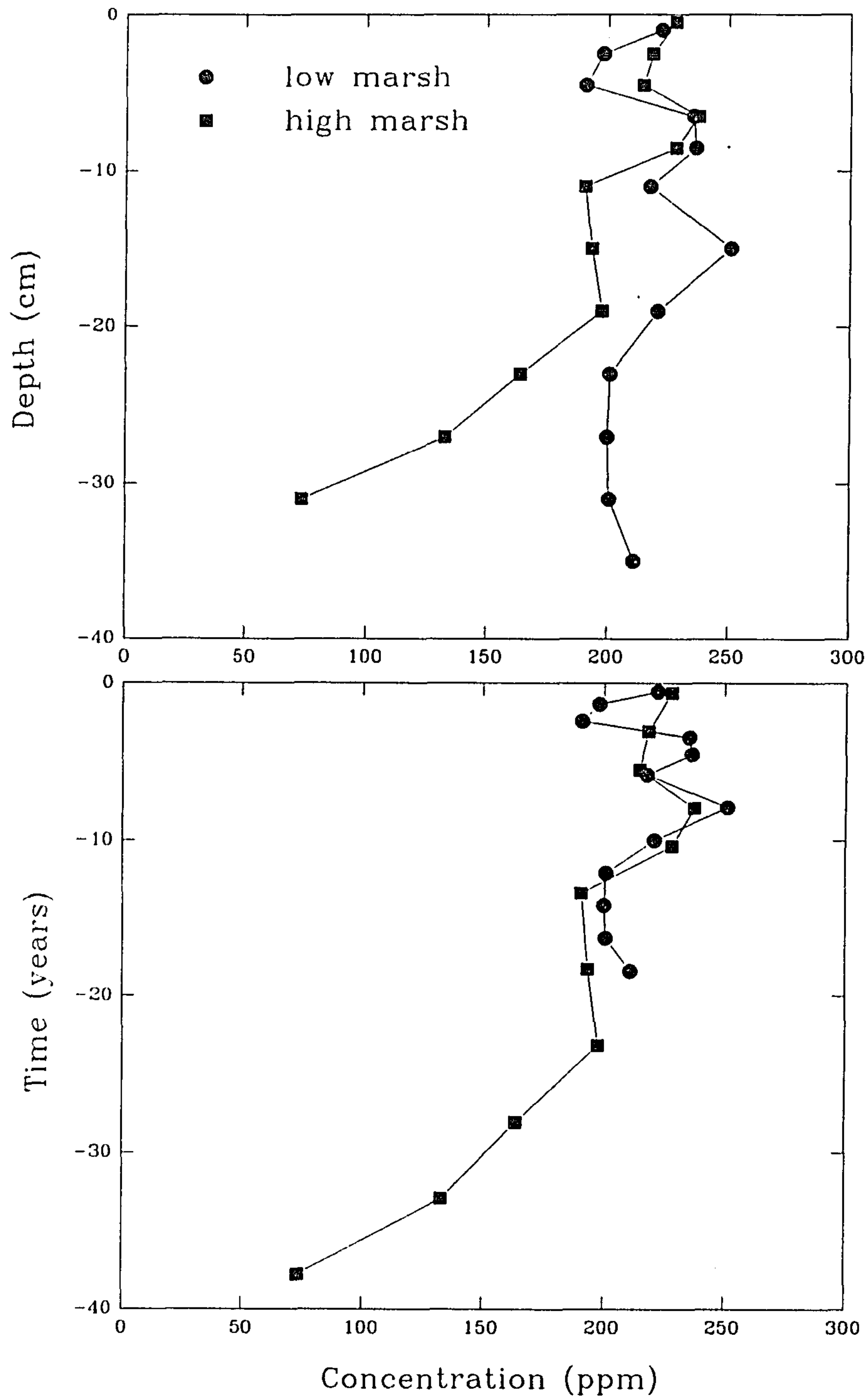

Figure 5.4b. Depth profiles of Zn from the high and low marsh at the Wistula River (top) and sediment chronologies of Zn for the same samples (bottom).

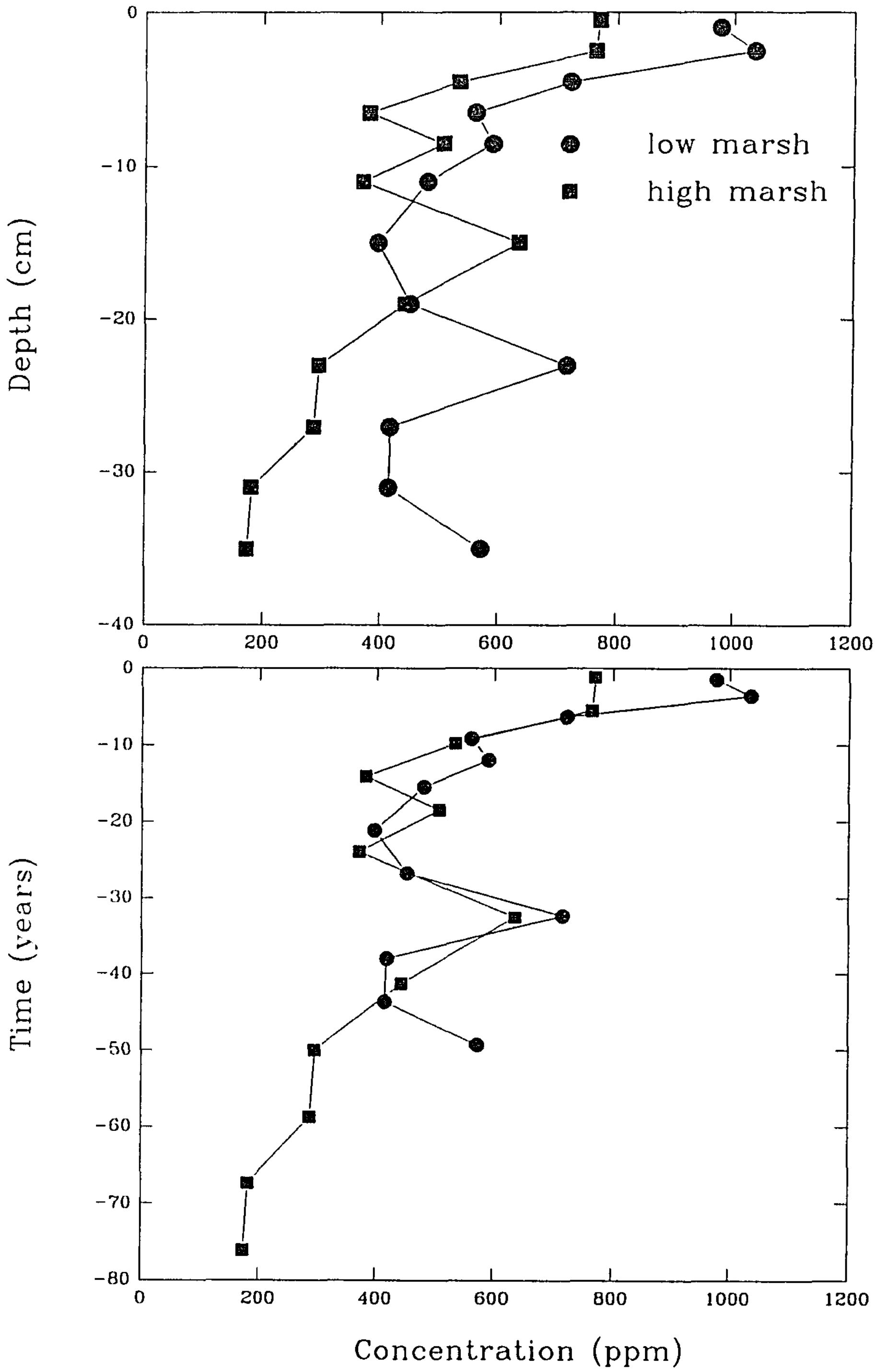

Figure 5.4c. Depth profiles of Zn from the high and low marsh at the Oder River (top) and sediment chronologies of Zn for the same samples (bottom).

determining metal concentrations versus sediment depth, it is very unlikely that we would find such good agreement in the metal chronologies from the two cores. Because of this evidence, I feel that these profiles represent real chronologies and do not reflect post-depositional mobility within the sediment column. However, there were some cases where there was not good agreement between the two cores, including the profile of Zn from St. Annaland marsh. In addition, the agreement between the two cores for most of the metal chronologies from Stiffkey marsh was not as good as at the other sites. These cases may indicate problems in post-depositional mobility.

Metal chronologies

Chronologies for each of the metals are given in Figures 5.5a-f and are discussed below by site. Sediment in cores from Stiffkey date back approximately 100 years (1890s) and indicate a strong increase in most metals approximately 60 to 70 years ago in the core from the low marsh. The high marsh core had slightly higher values in these oldest sediments. Pb concentrations in both cores reached peak values 30 to 50 years ago, and these concentrations were 2 to 4 times greater than values from the bottom of the two cores. Surface samples had a slight decrease in Pb concentrations over the last 20 to 30 years. Profiles also showed a large increase in Cu concentrations, with 2 to 3 fold difference in shallow and deep sediments. The low marsh showed a more abrupt increase in Cu concentrations 60 years ago, compared to the gradual increase in the high marsh samples. Profiles for Ni, Cd and Cr show very constant rates throughout the cores from the high marsh. There are

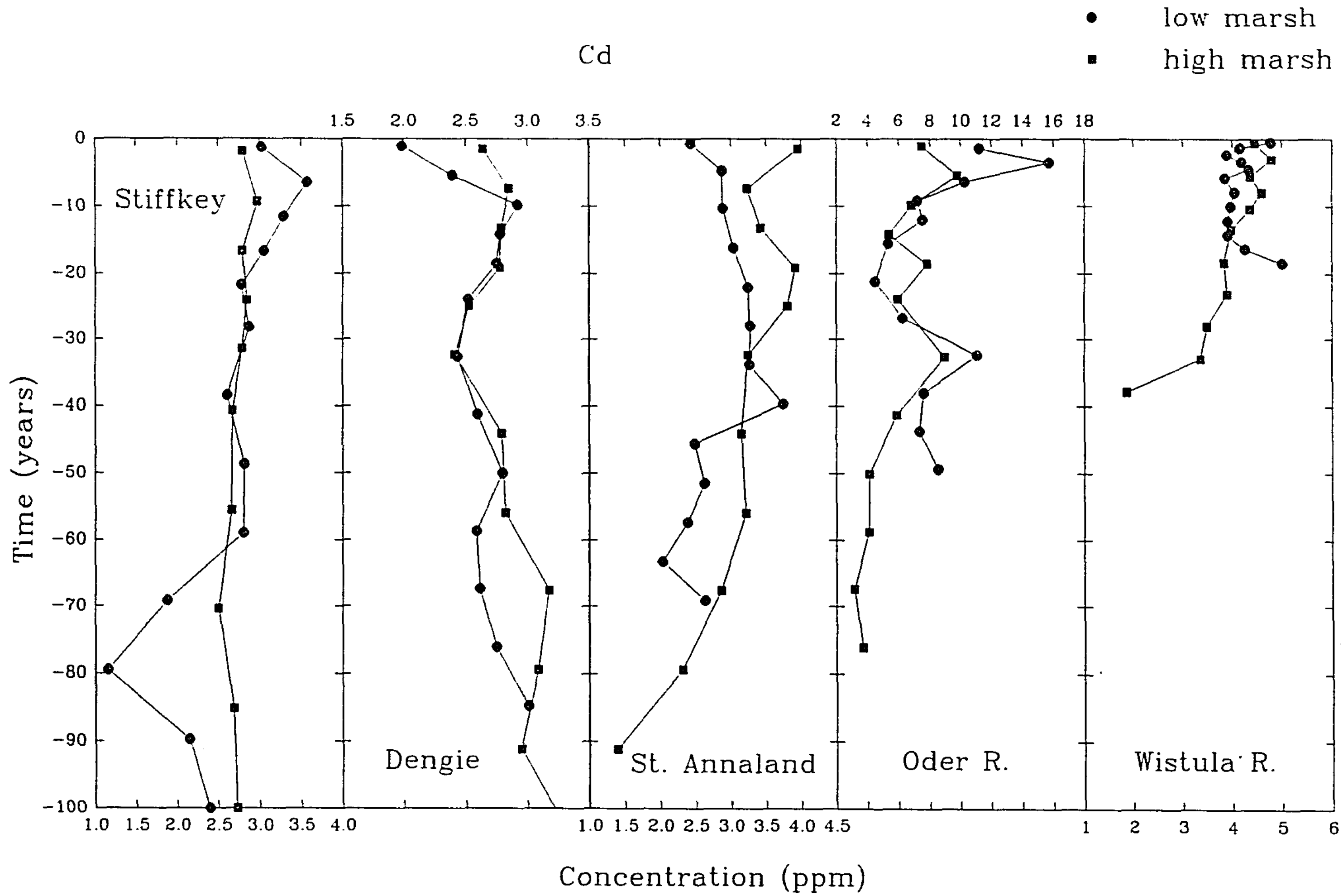

Figure 5.5a. Chronology of sediment Cd concentrations for high and low marsh samples from all five sites.

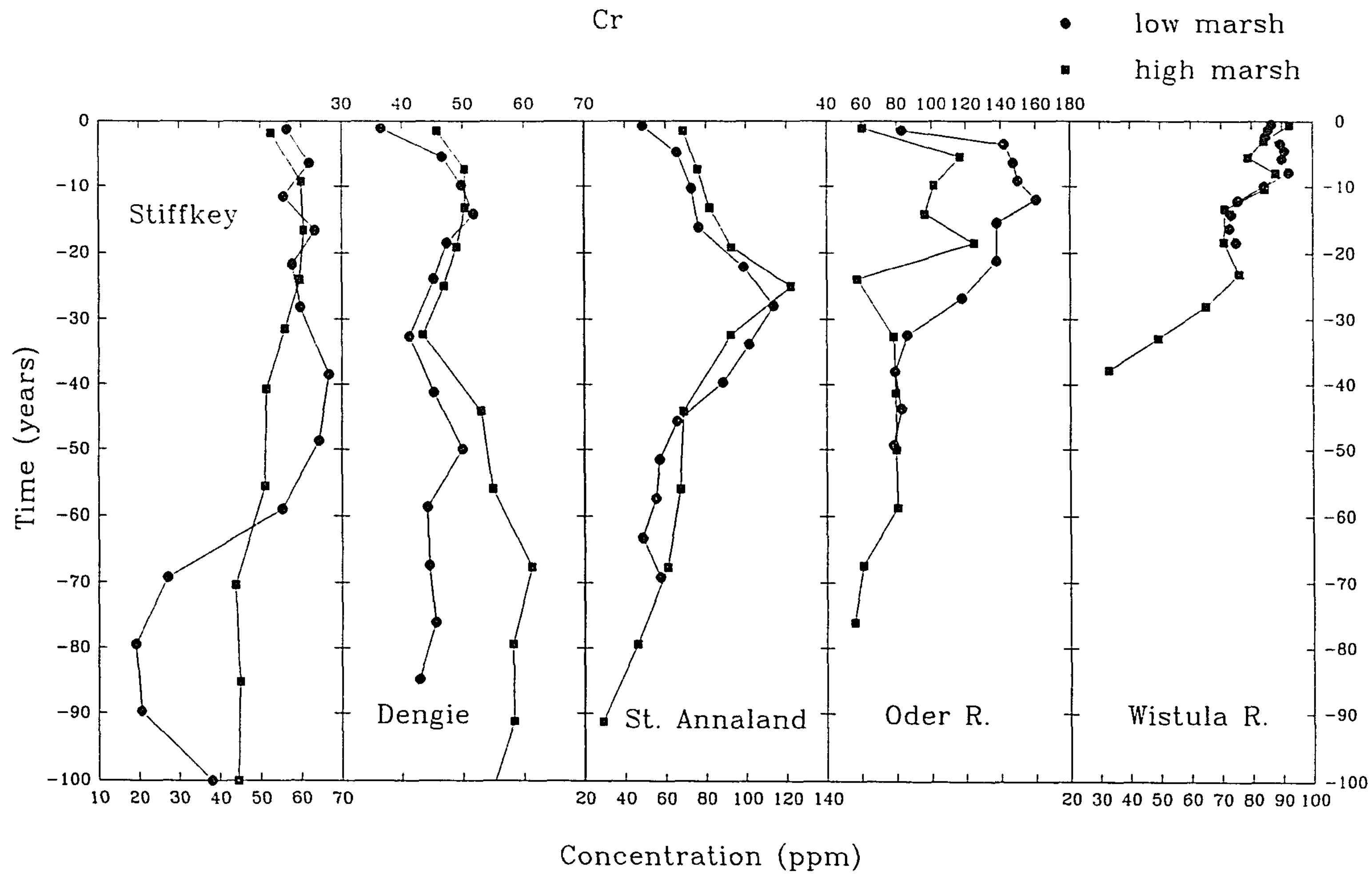

Figure 5.5b. Chronology of sediment Cr concentrations for high and low marsh samples from all five sites.

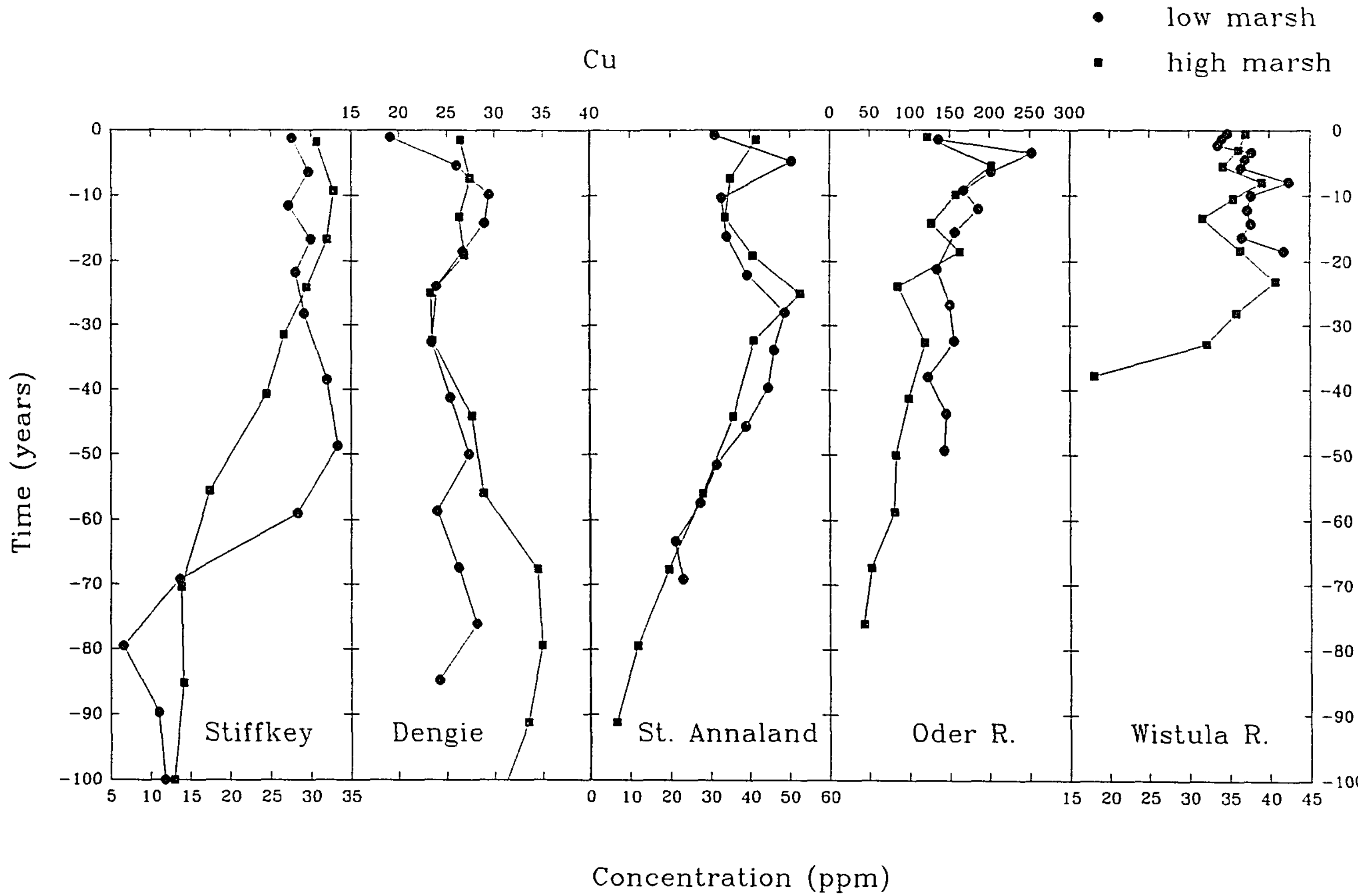

Figure 5.5c. Chronology of sediment Cu concentrations for high and low marsh samples from all five sites.

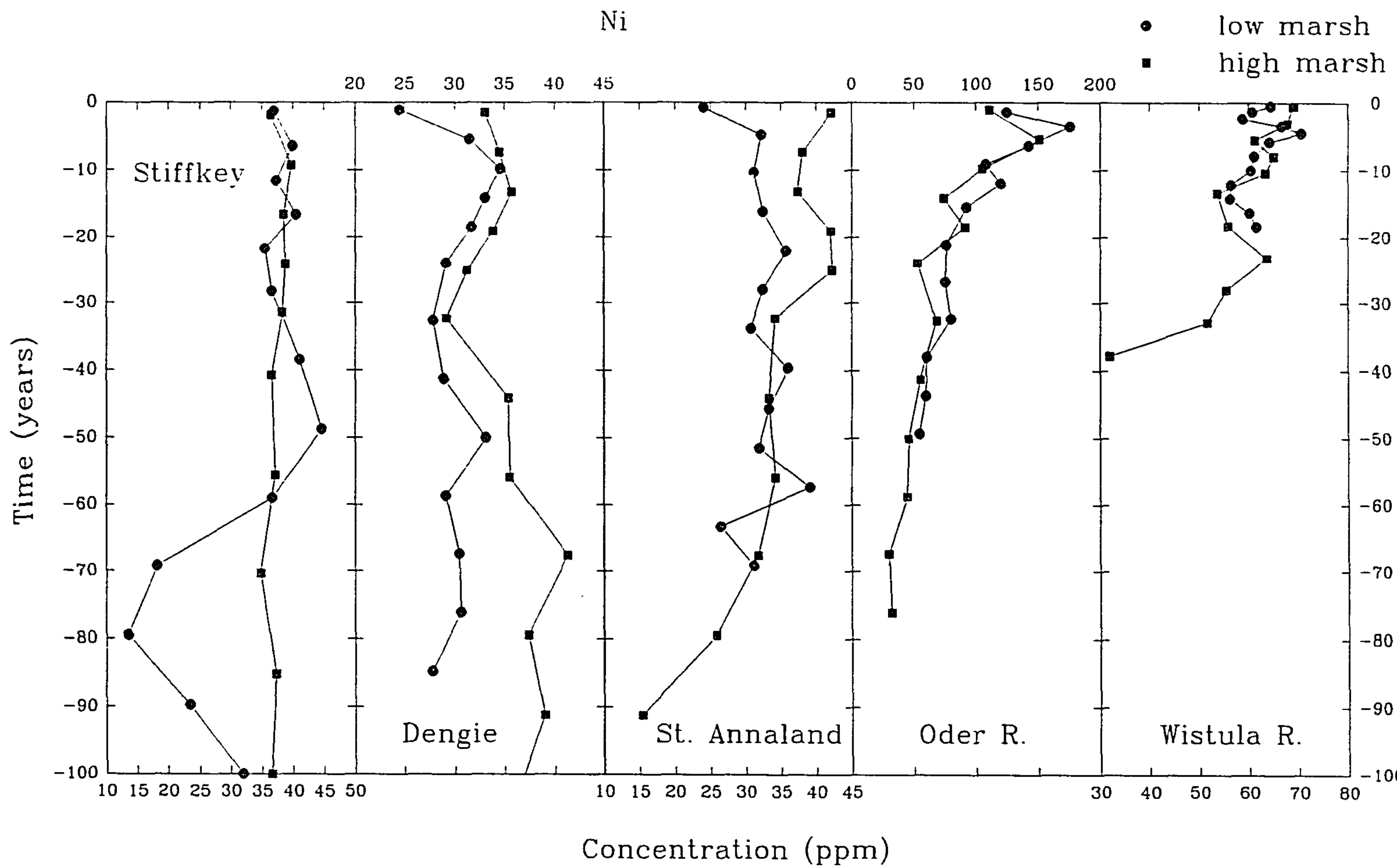

Figure 5.5d. Chronology of sediment Ni concentrations for high and low marsh samples from all five sites.

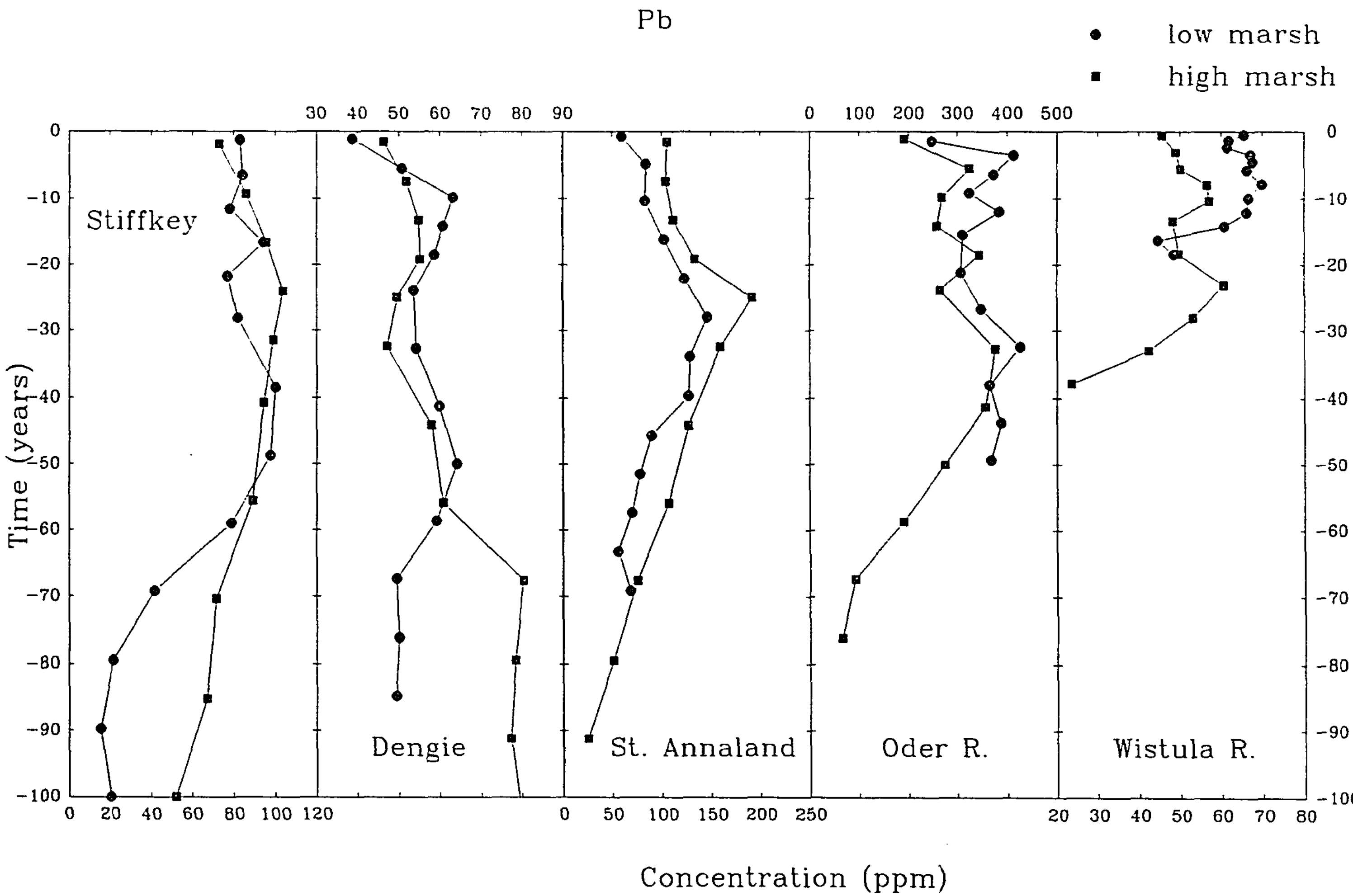

Figure 5.5e. Chronology of sediment Pb concentrations for high and low marsh samples from all five sites.

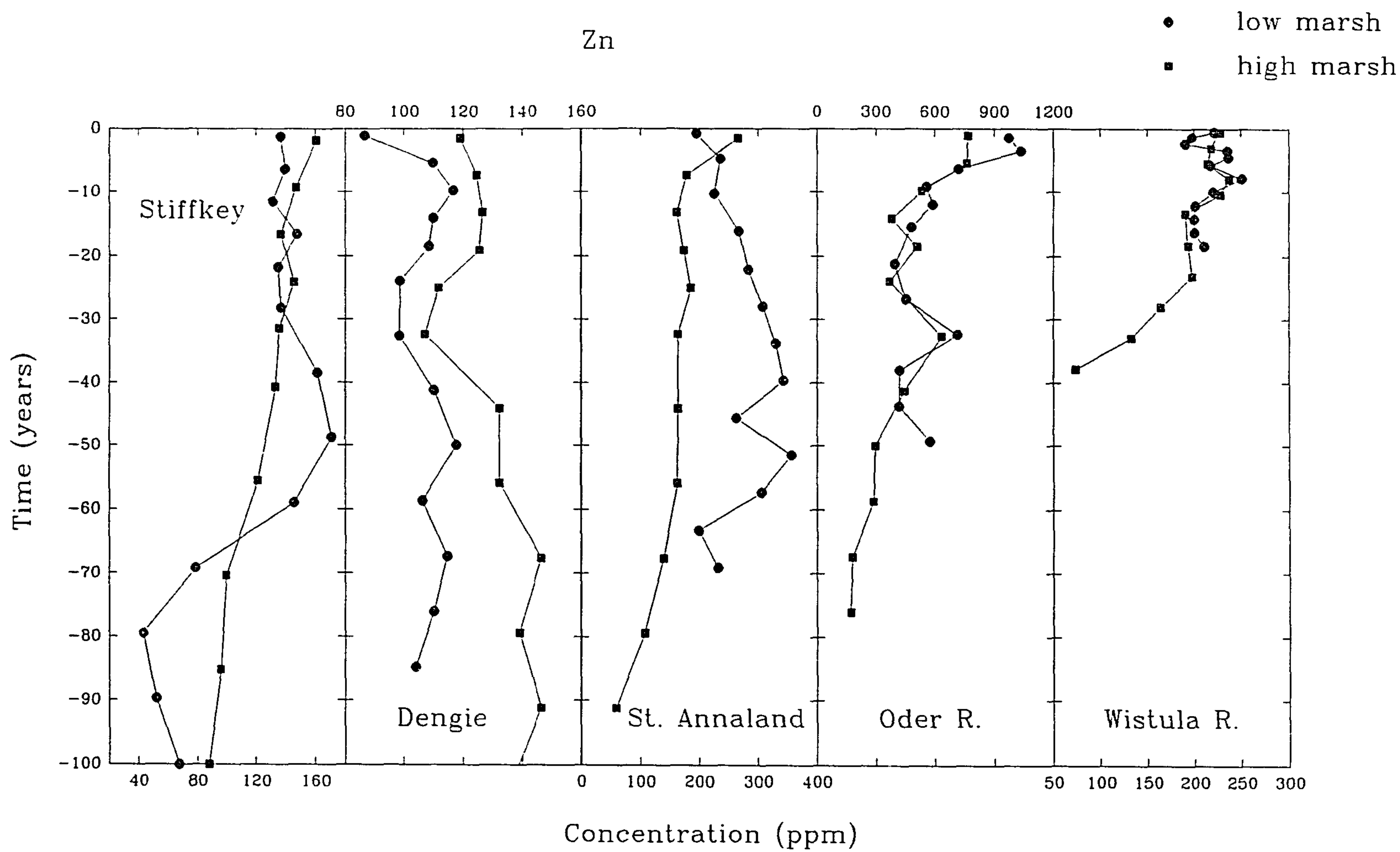

Figure 5.5f. Chronology of sediment Zn concentrations for high and low marsh samples from all five sites.

increases in these metals from the low marsh samples approximately 70 years ago. Cr and Ni show very slight decreases since then, while Cd is relatively constant except for a sharp increase in its concentration 10 years ago.

The two cores from Dengie marsh had the lowest levels of heavy metals of the 10 cores that were sampled, and the chronologies for these samples did not show increases in metals, as the other cores did. Highest concentrations were found at depth, with a gradual decrease until about 30 years ago. There was a slight increase to near peak values 10 years ago, and both cores showed very similar trends for all of the metals of interest. In addition, the range of concentrations from Dengie Marsh was much less than for the cores from other areas. These trends may be related to changes in iron hydroxides or clay content. Individual correlations of many metals with Fe were high from these cores. Given the small range of variation in metal concentrations in these samples, and the correlation with Fe profiles, there is no evidence of anthropogenic increases in metals at this site.

At St. Annaland Marsh in the Eastern Schelde, concentrations of Pb, Cu, and Cr were highest approximately 25 years ago, with Cu showing a sharp increase in the last five years. The peaks represented concentrations approximately five times greater than concentration from the bottom of the core (early 1900s) indicating a very significant increase in pollutant levels in this area. There were also increases in Cd and Ni concentrations; however, these increases were not as great, and there was no sharp decrease in these metals over the last 25 years. The Zn profile was very different than any of the other metals, and the two cores show very different patterns.

It is not clear why the Zn profiles are so different for this site. Zwolsman et al. (1993) found very similar trends in metals from two salt marshes that they sampled in the Western Schelde. They documented peak concentrations of Cd, Cr, Cu, Pb, and Zn at approximately 1961-65, although peak concentrations were lower than in my cores from St. Annaland. In addition, more recent peaks in Cu concentration were also found in their cores.

The sharp decrease in Pb, Cu, and Cr concentrations from St. Annaland coincide with hydrological changes due to flood protection projects in the southwestern area of the Netherlands (Heip 1989, Smaal and Nienhuis 1992). A large dam just northeast of St. Annaland (the Volkerak dam) was finished in 1969 (Smaal and Nienhuis 1992), and this dam lowered the input of Rhine and Meuse River water to the area substantially. Additional water control structures in the area, including Philips Dam (1986) also may have affected river input to the area (Small and Nienhuis 1992). However, Zwolsman et al. (1993) attributed the decreases in the Western Scheldt to changes in pollutant inputs, not to hydrological changes, because the Western Scheldt has not been as hydrologically altered as the Eastern Scheldt. Beurskens (1993) have also documented similar trends in heavy metal pollution from a large lake used for Rhine River sediment collection. Given the agreement in chronologies between my data and these two studies, it is not clear whether the decrease in pollutants at St. Annaland can be attributed to hydrology or to a regional drop in riverine and estuarine pollutant levels.

Samples from the Oder River in Poland, showed the highest levels of all metals from any of the cores that were sampled. This marsh is situated within Szczecin Bay and is influenced directly by industries within the bay. Most metals had relatively low levels in the oldest samples from the high marsh (70 to 80 years ago). Pb concentrations reached a peak approximately 30 years ago and have remained relatively high over the last three decades. Cr concentrations also had a sharp increase 20 to 30 years ago. There was a peak in Cd and Zn concentrations in these sediments 30 to 35 years ago, and surface samples also show large increases in these two metals. Profiles of Cu and Ni from the Oder River wetlands show very recent increases in concentrations, with relatively low levels up until five years ago. Peak concentrations of these were enriched from 3 to 5 times concentrations from 70 years ago.

Subtidal samples from dredging operations in Szczecin Bay have found very high levels of metals in organic rich sediments (Niedźwiecki et al. 1989, 1991). Peak concentrations from the surface sediments were similar to values that I found, with concentrations in the following ranges: Cd (4-17 ppm), Cu (70-300 ppm), Pb (100-350 ppm), and Zn (700-2300 ppm). Concentrations increased with increases in organic content (Niedźwiecki et al. 1991). Samples from the Eastern part of the Bay (Chudecki and Niedźwiecki 1987) also had elevated levels of Cu and Zn.

Finally, the samples from the Wistula River had the shortest chronology because of the high sedimentation rates at this marsh, with the oldest samples dating back approximately 40 years. These samples had relatively low levels of heavy

metals, especially compared to the samples from the Oder River. Concentrations of most metals at the surface were similar to background values from the Oder River. All metals showed a sharp increase in concentration 30 to 40 years ago, but have remained at relatively constant values since then.

The differences in chronologies between the Oder and Wistula River sites is probably due to the difference in hydrology between these two sites. The marshes at the Wistula River were situated at the edge of the Wistula Lagoon (Wisłany Zalew), along a small distributary of the Wistula, the Cieplicówka River. I was not able to find undisturbed marshes with sediments that could be dated, near the major mouth of the Oder River. This area is not directly influenced by industries in Gdansk, since the lagoon inlet from Gdansk Bay is far east of the city.

Although I did not find highly elevated values in the samples from this area, it is likely that other areas closer to the mouth of the river have elevated heavy metal concentrations. This area, along with the Oder River mouth, has been designated as a highly polluted area within Poland (Pawłowski 1990). Szefer and Skwarzec (1988) have documented elevated levels of Cd, Cu, Pb and Zn in sediment samples from Gdansk Bay. Peak values were found in samples from the 1950's with increases of two to three times background values. High levels of metals have also been found in fish and benthic fauna from the Gdansk Bay (Szefer 1990), and Müller et al. (1980) documented heavy metal trends from the western Baltic, and found large increases in Zn and Pb in all three cores which they collected.

Differences between low and high marsh samples

Bricker (1993) evaluated differences in atmospheric versus hydrologic inputs of metals to marshes by comparing differences in metal concentrations between high and low marsh samples. Low marsh sites reflect hydrologic inputs since they are inundated more often by tidal waters; whereas, high marsh sites are likely to reflect atmospheric inputs. However, Bricker (1993) found limited agreement between estimated atmospheric loadings and high marsh metal concentrations. Zwolsman et al. (1993) indicated that for their sites in the western Scheldt that atmospheric deposition was not an important part of pollutant inputs.

These samples do not indicate a consistent trend in atmospheric versus hydrologic inputs to these marshes. In most cases, there were shifts in the relative intensity of high and low marsh samples over time at a particular site (i.e., Cr at St. Annaland) or there were not large differences between the two cores from a site. Also, trends were not similar between metals at the same site. There were no sites where low or high marsh sites were consistently higher for the six metals that I measured. Because of the large variability between high and low marsh samples, it is not possible to identify the relative importance of the various sources at these sites with the present data.

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CONCLUSIONS

Considering the importance and public protection that are now given to coastal wetlands, we need to learn more about their natural processes and potential human impacts to these areas in order to ensure their long-term survival. The long-term survival of coastal wetlands is not a simple issue due, in part, to the complex ecological processes which naturally maintain them. In the future, wetlands will continue to be affected by increases in sea level rise and by other human impacts, including heavy metal inputs. Given these potential impacts, I have tried to evaluate some of the factors that may affect their long-term survival and to document current and historic rates of heavy metal pollution in coastal wetlands from the northern Gulf of Mexico and northern Europe. Below, I will review the results and conclusions from these studies.

The sites along the northern Gulf of Mexico had a significant difference in vertical accretion rates across the marsh, with the highest rates at the low end of the marsh and lowest rates in the upper marsh. Organic accumulation rates also declined significantly across the marsh. These results agreed with previous studies from Louisiana which have shown that there is a strong relationship between vertical accretion rates and organic matter accumulation. Further results from the statistical analysis in Chapter 3 also confirmed the relationship between vertical accretion and organic accumulation rates. These results do not imply that mineral matter is not important in maintaining the elevation of the marsh, because mineral matter input is important in affecting organic production and sediment bulk density. However, the

correlation between vertical accretion rates and organic accumulation suggest that organic matter is extremely important in forming the structure of coastal wetland sediments.

Previous studies found that accretion balance was negatively correlated with tidal range and that low tidal range sites were likely to have negative accretion balances. Contrary to those findings, all of the sites that I studied from the Gulf coast had low tidal ranges, and all of them had positive accretion balances. The regression between tidal range and accretion balance, with my new data, was significant, but the R^2 for the regression dropped to 0.43. These results were also corroborated by the statistical analyses that were performed in Chapter 3 and included data from 16 sites world-wide. There was no significant relationship between accretion balance and tidal range for these data; however, the only sites that had negative accretion balances were low tidal range sites. The sites with negative accretion balances also had high rates of subsidence, and future studies are needed to fully evaluate the relative importance of subsidence and tidal range on accretion balance.

From the study of the five sites in northern Europe, it appears that the ^{137}Cs from the 1986 accident at Chernobyl is an useful sediment marker in many areas of Northern Europe. It should be used in the future, and in conjunction with the 1963 marker and ^{210}Pb dating, it could help to identify differences in short and long-term accretion processes. Also, by using the two ^{137}Cs markers to identify a segment of the sediment column and repeatedly sampling the same area, changes in that segment

of the sediment column due to compaction and decomposition could be followed over time.

In addition, from this study, it is clear that there are large differences in sediments from western Europe coastal wetlands versus those from Poland. The accretion rates from the western European samples were typical of other studies that have been completed in western Europe, but the accretion rates from the Polish samples were much higher. Polish sediments also had higher organic content than the western European sediments. Vertical accretion rates based on ^{210}Pb were estimated for five of the northern European cores, and rates for three cores were lower than those based on ^{137}Cs dating, indicating that the more long-term measurements included effects of compaction and decomposition. Profiles of sediment bulk density and organic content confirmed that deeper sediments have undergone compaction and decomposition. Vertical accretion rates were greater than rates of relative sea level rise for all of the European cores.

In addition to the results discussed above, the statistical analyses in Chapter 3 showed that vertical accretion rates were best described by a regression using relative sea level rise, surface organic content, sediment bulk density, position within the marsh, and the interaction between relative sea level rise and position ($R^2 = 0.83$). Relative sea level rise was the most important variable affecting vertical accretion rates, and part of this impact was due to the influence of the data from the sites in Louisiana. Similar results were obtained for the regression without the Louisiana data, indicating that the trend is not only attributable to these data points. These

results indicate that marshes may respond to increases in sea level rise with increased rates of vertical accretion. However, the significant interaction between relative sea level rise and marsh position indicated that vertical accretion rates from low marsh sites respond more to increases in relative sea level rise than middle marsh or high marsh sites. High and middle marsh sites will be most susceptible to future increases in sea level rise and are likely to be converted to low marsh habitats or mudflats. Regression results for accretion balance resulted in a much lower R^2 value (0.43).

The computer model successfully simulated accretion rates and sediment profiles for sediment cores from a high marsh site at Stiffkey marsh and from a mid marsh site at Biloxi Bay, Mississippi. It also identified processes that are most important in affecting accretion rates. The use of both accretion rates and sediment characteristics to calibrate the model made it more realistic than other sediment accretion models that have been developed previously. The model was calibrated using accretion rates based on ^{137}Cs dating, and the predicted rates of vertical accretion for a 100 year time span matched exactly with the rates measured by ^{210}Pb dating for the Stiffkey core.

Sensitivity analyses indicated that the following factors were the most important in affecting model generated accretion rates: pore space, mineral matter deposition, initial elevation, sea level rise, and below-ground production. Additionally, the model was a useful tool for predicting changes to marsh relative elevation and long-term survival due to potential increases in eustatic sea level rise.

The results from the field studies, statistical analyses, and the simulation model all indicated that more data are needed concerning organic content, pore space, and sediment structure in order to obtain a better understanding of wetland sedimentation processes. The statistical analysis and the results from the Gulf coast study showed that organic matter was important in affecting vertical accretion rates, probably through sediment structure and the creation of pore space. Additionally, the model indicated that pore space is one of the most important variables affecting overall accretion rates and the relative elevation of the marsh. Given these findings, future research should be conducted to evaluate sediment structure and pore space dynamics in surface sediments from coastal wetlands.

Finally, in the last chapter, I demonstrated the utility of dated sediment cores in determining heavy metal chronologies in coastal wetlands. The chronologies of heavy metal concentrations from the two cores (high and low marsh) at each of the five sites showed very good agreement, indicating that sediment profiles represent historical inputs of heavy metals and not diagenetic processes. High correlations between Cr, Cd, Cu, Pb, and Zn concentrations indicate that these metals probably come from the similar sources and behave similarly in the sediment column. Heavy metal concentrations in the sediments from some of the areas were high, with peak sediment concentrations up to five times greater than found in the oldest sediment samples (1890's to 1900's). Chronologies indicated very different histories of pollution for the five different areas. Local hydrology appeared to be very important in affecting metal accumulation. Metal concentrations have recently decreased in the

cores from St. Annaland (the Netherlands) and Stiffkey Marsh (U.K.) but remained high throughout the upper part of the cores from the Oder River, Poland. No consistent trends were found in differences between atmospheric and hydrological inputs of metals for these cores.

APPENDIX A

DATA USED FOR STATISTICAL ANALYSES IN CHAPTER 3

Location	Core Number	Method	Position	Vertical Accretion Rate	Period	Relative Sea Level Rise	Accretion Balance
Rhode Island	1	P	l	0.25	40	0.19	0.06
Rhode Island	2a	P	l	0.44	68	0.19	0.25
Rhode Island	2b	P	l	0.3	100	0.19	0.11
Rhode Island	3	P	l	0.55	72	0.19	0.36
Rhode Island	4b	P	h	0.6	75	0.19	0.41
Rhode Island	5	P	l	0.37	54	0.19	0.18
Rhode Island	6	P	l	0.49	61	0.19	0.3
Farm River	x	P	h	0.3	70	0.19	0.11
Flax Pond	3	P	m	0.47	96	0.22	0.25
Flax Pond	2	P	h	0.63	87	0.22	0.41
Great Marsh	x	P	h	0.47	119	0.2	0.27
Blackwater	East	P	m	0.458	42	0.39	0.068
Blackwater	West	P	m	0.262	42	0.39	-0.128
San Fran.	Bird	C	h	0.4	25	0.18	0.22
San Fran.	Bmb	C	h	0.5	25	0.39	0.11
Biloxi Bay	1-2	C	m	0.32	27	0.15	0.17
Biloxi Bay	1-3	C	h	0.61	27	0.15	0.46
Biloxi Bay	2-1	C	l	0.68	27	0.15	0.53
Biloxi Bay	2-2	C	m	0.54	27	0.15	0.39
Biloxi Bay	2-3	C	h	0.46	27	0.15	0.31
Louisiana	x	C	l	1.35	20	0.95	0.4
Louisiana	x	C	h	0.64	20	0.95	-0.31
Louisiana	x	C	l	1.4	20	0.95	0.45
Louisiana	x	C	h	0.59	20	0.95	-0.36
Louisiana	x	C	l	1.35	20	0.95	0.4
Louisiana	x	C	h	0.75	20	0.95	-0.2
Pacific NW	ER-1	C	m	0.66	28	0.13	0.53
Pacific NW	ND-2	C	l	0.3	28	0.2	0.1
N. Car.-Reg	strm	C	l	0.27	25	0.19	0.08
N. Car.-Reg	back	C	h	0.09	25	0.19	-0.1
N. Car.-Irr	strm	C	l	0.36	25	0.19	0.17
N. Car.-Irr	back	C	h	0.24	25	0.19	0.05

Location	Core Number	Method	Position	Vertical Accretion Rate	Period	Relative Sea Level Rise	Accretion Balance
Aransas	1	C	l	0.65	29	0.31	0.34
Aransas	2	C	m	0.42	29	0.31	0.11
Aransas	3	C	h	0.26	29	0.31	-0.05
Aransas	7	C	h	0.48	29	0.31	0.17
S. Bernard	1	C	l	0.89	29	0.63	0.26
S. Bernard	2	C	m	0.61	29	0.63	-0.02
S. Bernard	3	C	h	0.54	29	0.63	-0.09
S. Bernard	4	C	l	0.75	29	0.63	0.12
S. Bernard	5	C	m	0.54	29	0.63	-0.09
S. Bernard	6	C	h	0.39	29	0.63	-0.24
Florida Keys	3	C	l	0.42	28	0.22	0.2
Florida Keys	6	C	l	0.39	28	0.22	0.17
Florida Keys	2	C	h	0.42	28	0.22	0.2
Florida Keys	4	C	h	0.19	28	0.22	-0.03
Florida Keys	5	C	h	0.19	28	0.22	-0.03
Stiffkey, UK	1	C	l	0.39	28	0.07	0.32
Stiffkey, UK	2	C	m	0.27	28	0.07	0.2
Dengie, UK	1	C	l	0.46	28	0.15	0.31
Dengie, UK	2	C	h	0.34	28	0.15	0.19
St. Annaland	1	C	l	0.68	28	0.2	0.48
St. Annaland	2	C	h	0.34	28	0.2	0.14

Location	Core Number	Tidal Range	Surface Organic Content	Lower Organic Content	Sediment Bulk Density	Lat.	Organic Accum. Rate	Mineral Accum. Rate
Rhode Island	1	1.4	27	12	0.47	41.8	110	280
Rhode Island	2a	1.3	33	8	0.46	41.8	440	990
Rhode Island	2b	1.3	33	8	0.59	41.8	350	820
Rhode Island	3	1.3	14	15	0.47	41.7	430	2070
Rhode Island	4b	1.2	40	32	0.43	41.6	730	1450
Rhode Island	5	1.1	30	18	0.44	41.5	380	890
Rhode Island	6	0.1	45	28	0.43	41.3	510	1130
Farm River	x	1.8	31	30	0.23	41.3	220	460
Flax Pond	3	2	27.6	15.6	0.28	40.9	325	1080
Flax Pond	2	2	26.6	18.7	0.26	40.9	436	149
Great Marsh	x	1.3	29.2	8.8	0.54	38.8	NA	NA
Blackwater	East	0.3	73.7	53.80	0.113	38.3	NA	NA
Blackwater	West	0.3	56.9	41.53	0.148	38.3	NA	NA
San Fran.	Bird	1.7	22.5	7.5	0.45	37.6	169	999
San Fran.	Bmb	1.7	10	5.1	0.91	37.6	241	3119
Biloxi Bay	1-2	0.55	23.81	20.03	0.3163	30.3	205.44	669.49
Biloxi Bay	1-3	0.55	20.32	23.65	0.2769	30.3	350.27	1304.34
Biloxi Bay	2-1	0.55	19.02	21.78	0.2929	30.3	422.63	1731.68
Biloxi Bay	2-2	0.55	28.58	19.75	0.2883	30.3	370.04	1136.71
Biloxi Bay	2-3	0.55	25.44	19.11	0.2998	30.3	286.24	995.97
Louisiana	x	0.3	33	69	0.18	29.7	797	1634
Louisiana	x	0.3	52	69	0.08	29.7	269	243
Louisiana	x	0.3	22	30	0.27	29.6	826	2954
Louisiana	x	0.3	42	30	0.14	29.6	348	478
Louisiana	x	0.3	20	10	0.35	29.3	675	2700
Louisiana	x	0.3	20	10	0.29	29.3	435	1740
Pacific NW	ER-1	2.16	22.2	10.6	0.134	45	142	490
Pacific NW	ND-2	2.93	9.4	6	0.665	47	315	3535
N. Car.-Reg	strm	0.3	7.5	0.79	0.773	35	137	1139
N. Car.-Reg	back	0.3	3.6	0.52	1.19	35	51	677
N. Car.-Irr	strm	0.1	43.8	59.3	0.151	35	280	311
N. Car.-Irr	back	0.1	52.7	64.1	0.139	35	196	147

Location	Core Number	Tidal Range	Surface Organic Content	Lower Organic Content	Sediment Bulk Density	Lat.	Organic Accum. Rate	Mineral Accum. Rate
Aransas	1	0.43	15.65	9.83	0.714	28.3	476.72	3185.17
Aransas	2	0.43	20.08	9.85	0.730	28.3	352.62	1557.1
Aransas	3	0.43	7.52	9.53	0.910	28.3	170.71	2002.8
Aransas	7	0.43	13.69	8.26	0.746	28.3	368.44	2626.34
S. Bernard	1	0.55	11.62	7.67	0.706	28.8	588.68	4978.64
S. Bernard	2	0.55	10.34	5.22	1.045	28.8	412.51	4897.66
S. Bernard	3	0.55	10.23	4.97	1.08	28.8	382.24	3965.53
S. Bernard	4	0.55	6.53	4.79	1.046	28.8	438.02	6831.51
S. Bernard	5	0.55	9.02	5.26	1.126	28.8	371.44	4521.84
S. Bernard	6	0.55	6.97	5.02	1.203	28.8	285.3	4175.93
Florida Keys	3	0.65	70.13	64.8	0.125	25	331.6	143.5
Florida Keys	6	0.71	52.42	67.3	0.130	25.3	304.4	171.5
Florida Keys	2	0.09	54.9	40.5	0.2125	44	361	303.7
Florida Keys	4	0.65	43.57	53.6	0.2419	25	115.4	123.3
Florida Keys	5	0.71	68.07	61.5	0.16	25.3	156.3	64.9
Stiffkey, UK	1	5.5	15.23	5.752	0.7601	53	319.2	1793
Stiffkey, UK	2	5.5	24.99	10.85	0.6771	53	304.1	839.1
Dengie, UK	1	3.5	12.4	11.83	0.8186	51.7	425.1	3172.2
Dengie, UK	2	3.5	14.23	13.82	0.7511	51.7	347.7	2113.4
St. Annaland	1	4.5	18.47	11.4	0.4302	51.6	441.5	2010.5
St. Annaland	2	4.5	30.07	7.88	0.604	51.6	322.5	814.4

APPENDIX B

COMPLETE LISTING OF FORTRAN SIMULATION PROGRAM

```
c this is sed5.for  -  sediment accretion program for fortran
c this version uses an exponentially decreasing underground production function
c this change is in the subroutine - rtprod
c
c additionally - this version uses a new decomposition model - still 3
c different rates - but the rates for each year class are from an
c exponential decay curve.
c
c declaring variables
c
      real*8 org, min, orgden, minden, h2oden, pore, minin, orgin, h2oin
      real*8 orgbd, minbd, bulkd, porg, depth, massabv, intelv, relelv
      real*8 slr, subsid, totorg, totvol, orgvol, minvol, densabv
      real*8 dt, mindev, porelim, refrac
      real*8 pctpore, tpore
      real*8 acc30, org30, min30, acc100, org100, min100
      real*8 acc300, org300, min300, finelv
      integer time, t2, endtim
      dimension org(0:1000,4), min(0:1000), pore(0:1000), totvol(1000)
      dimension orgbd(1000), minbd(1000), porg(1000), minvol(1000)
      dimension depth(0:1000),  massabv(0:1000), densabv(0:1000)
      dimension bulkd(1000), relelv(0:1000), totorg(0:1000), orgvol(1000)
      dimension pctpore(0:1000)
      dimension tpore(0:1000)
c
c initializing values
c
c     open (25, file = 'sksxx.dat', status = 'unknown')
c     open (26, file = 'orgac5dat', status = 'unknown')
      data totorg /1001*0.0/
      data min/1001*0.0/
      data pore/1001*0.0/
      data totvol/1000*0.0/
      data depth/1001*0.0/
      data massabv/1001*0.0/
      data densabv/1001*0.0/
      data relelv/1001*0/
      data pctpore/1001*0/
```

```
        orgin = 0.03
        minin = 0.48
        h2oin = 0.82
        mindev = 0.0
        strint = 5.0
        strdev = 0.3
        porelim = 0.4
        refrac = .06
c
c h20in is a percent.  It has to be converted to a volume
c to be useful for calculations.  The conversion from % to volume is:
c porespace volume = ((%)/(1-%))*(minvol + orgvol)
        orgden = 1.14
        minden = 2.61
        h2oden = 1.00
        intelv = 250.00
        slr = 0.15
        subsid = 0.0
        endtim = 301
c
c ***********************************************
c this is the beginning of the main control loop *
c ***********************************************
c
        do 100 time = 1, endtim
c
c this section moves all values down one section
c before the next round of growth, new input and decomposition
c
              do 10 t2 = time-1, 0, -1
                    org(t2+1,4) = org(t2,4)
                    org(t2+1,3) = org(t2,3) + org(t2,2)
                    org(t2+1,2) = org(t2,1)
                    org(t2+1,1) = 0.0
                    min(t2+1) = min(t2)
c                   pctpore(t2+1) = pctpore(t2)
                    tpore(t2+1) = tpore(t2)
10            end do
c
c **********************************************************
c * these are the new inputs of material onto the surface of *
c * the marsh (into the first position in the array).       *
c **********************************************************
c
```

```
            org(1,1) = orgin*(1-refrac)
            org(1,4) = orgin*refrac
            min(1) = minin
     &              *(rtemin(relelv(time-1)))
c
c
            tpore(1) = 1.00
            pctpore(1) = h2oin
            pore(1) = ((h2oin/(1-h2oin))*((org(1,1)/orgden)
     &      + (min(1)/minden)))
c
c
c
c
c ************************************************************
c * the following section is where the "yearly" calculations     *
c * take place.  It combines all of the other calculation sections *
c * from earlier versions of the model (7/10/93).                 *
c ************************************************************
c
c this section calculates the volume of each section
c based on the mass of organic matter, mineral matter, and
c water.  It will also use a compaction subfunction in
c the future. Compaction will be a function of the
c mass that is on top of the current section.
c
c
c this is where the new roots and rhizomes are put into
c the sediment.  rtprod is a subroutine/function
c that will determine root production based on depth/time
c
c this is also the decomposition section.  Again decomp is
c a subroutine based on depth/time.
c
            do 20 t2 = 1, time
                  istep = 10
                  dt = 1.0/floatj(istep)
                  do 19 ie = 1, istep
                  totorg(t2) = org(t2,1) + org(t2,2) + org(t2,3)
     &             + org(t2,4)
                  massabv(t2) = massabv((t2)-1) + totorg(t2-1) + min(t2-1)
     &            + pore(t2-1)
                  if (depth(t2-1).eq.0) then
                        densabv(t2) = 0
```

```
                    else
                          densabv(t2) = massabv(t2)/(depth(t2-1))
                    end if
                    orgvol(t2) = (totorg(t2)/orgden)
                    minvol(t2) = (min(t2)/minden)
                    if (t2.le.1) then
                          pctpore(t2) = pctpore(t2)
                    else
                    tpore(t2) = tpore(t2) - (tpore(t2) -
     &              tpore(t2)*(poresp(densabv(t2))))*dt
                    pctpore(t2)=porelim+(h2oin-porelim)*tpore(t2)
c       write (6,999) tpore(t2), pctpore(t2)
                    end if
              pore(t2) = ((pctpore(t2)/(1-pctpore(t2)))*(orgvol(t2) +
     &        minvol(t2)))
c             pore(t2) = h2oin
c the line above is for running the model without compaction
c pore space is constant for all sections.
c
                    totvol(t2) = orgvol(t2) + minvol(t2) + pore(t2)
                    depth(t2) = depth ((t2)-1) + totvol(t2)
                    porg(t2) = totorg(t2)/(totorg(t2) + min(t2))
                    bulkd(t2) = (totorg(t2)+min(t2))/totvol(t2)
                    orgbd(t2) = totorg(t2)/totvol(t2)
                    minbd(t2) = min(t2)/totvol(t2)
              org(t2,1) = org(t2,1)+((dt*(rtprod(depth(t2))*totvol(t2)))
     &                *(1-refrac))
     &          - ((dt*(((decmp1(depth(t2)))*org(t2,1)))))
              org(t2,2) = org(t2,2)
     &          - (dt*(((decmp2(depth(t2)))*org(t2,2))))
              org(t2,3) = org(t2,3)
     &          - (dt*(((decmp3(depth(t2)))*org(t2,3))))
              org(t2,4) = org(t2,4)+((dt*(rtprod(depth(t2))*totvol(t2)))
     &                *refrac)
c
c commenting out the 3 "&" lines above, cuts out decompostion
c
c
19                  end do
20            end do
c
c ********************************************************************
c * this section calculates the relative elevation of the marsh at the *
c * end of the year                                                    *
```

```
c ************************************************************************
c
        relelv(time)  = intelv + depth(time) - (slr*time) - (subsid*time)
c
c
c this loop prints at the end of all calculations - for checking data/program
c
c       do 90 t3 = 1, time
c       write(6, 1005) totorg(t3), min(t3), h2o(t3), totvol(t3), t3
c90     end do
100     end do
c
c       do 105 time = 1, endtim
c       orgdiff(time) = totorg(time) - totorg(time-1)
c       pctdiff(time) = orgdiff(time)/totorg(time)
c105    end do
c
c to get the output to print to the screen delete the cccc's
c from the next loop  -  110
c
c       do 110 time = 1, endtim
c       write(6,1004) totorg(time),min(time),pore(time),totvol(time),
c    &     depth(time), relelv(time), time
c110    end do
c
c
c
c       this is the loop that I need to use to print the data
c       to a file for sigmaplot graphs
c
c       do 200 time = 1, endtim
c       write (25,1001) totorg(time),min(time),
c    &     pore(time),totvol(time),orgvol(time),minvol(time),porg(time),
c    &     bulkd(time),depth(time),massabv(time),densabv(time),
c    &     relelv(endtim-time),time
c200    end do
c
c
c       do 201 time = 1, endtim
c       write (26,1001) totorg(time),min(time),
c    &     pore(time),pctpore(time),totvol(time),minvol(time),porg(time),
c    &     bulkd(time),depth(time),massabv(time),densabv(time),
c    &     relelv(endtim-time),time
c201    end do
```

```
c
c       do 210 time =1, endtim
c       write (26,1002) org(time,1),org(time,2),org(time,3),oldorg(time,1),
c     &  oldorg(time,2), pctpore(time), time
c210    end do
        finelv = relelv(1)
        acc30 = depth(30)
        acc100 = depth(100)
        acc300 = depth(300)
        do 250 time = 1, 300
c
        if (time .le. 30) then
        org30 = org30 + totorg(time)
        min30 = min30 + min(time)
        end if
c
        if (time .le. 100) then
        org100 = org100 + totorg(time)
        min100 = min100 + min(time)
        end if
c
        if (time .le. 300) then
        org300 = org300 + totorg(time)
        min300 = min300 + min(time)
        end if
c
250     end do
        write (6,1050) acc30, org30, min30
        write (6,1051) acc100, org100, min100
        write (6,1052) acc300, org300, min300
        write (6,1053) finelv
c
c
999     format (2(f9.5, 1x))
1001    format (8(f9.5, 1x), 4(f6.2, 1x) i5)
1002    format (6(f12.8, 1x), i5)
1004    format (' org, min, pore, totvol, depth, relelv, time', f7.5,1x,
     &     2(f7.3,1x),3(f9.4,1x),i5)
1005    format ('in the loop' 4(f10.3, 2x), i5)
1010    format ('beginning of loop', f6.2, i5, f15.5, f10.3)
1015    format(' organic subsections/pre loop' 3(f6.4, 2x))
1016    format(' organic subsections/post loop' 3(f6.4, 2x))
1030    format (' rlelv, int, depth, slr, sub,time', 3f6.3, i5)
1050    format (' acc, min & org rates for 30 years ', 3(f10.3, 2x))
```

```
1051  format (' acc, min & org rates for 100 years', 3(f10.3, 2x))
1052  format (' acc, min & org rates for 300 years', 3(f10.3, 2x))
1053  format (' final surface elevation          ', f10.3)
1055  format ('value of t2 at end of loop  ', i5)
2006  format ('if loop for porespace', i5)
2010  format ('porespace calcs', 4(f8.4, 2x))
      end
c
c
c
c
**********************************************************************
***
c *                                                               *
c *                    SUBROUTINES                                      *
c
**********************************************************************
***
c
c
c
c ********************************************
c *  RTPROD                              *
c *  Root production subroutine          *
c ********************************************
c
c
c      What follows is a subroutine for determining organic
c      production at various time / depths
c      1/11/94 - I am changing this so root production decreases \
c      exponentially with depth.
c      sed2.for has the old version of root production.
c
      real*8 function rtprod(d)
      real*8 kdist, d
      real*8 undpro
c
c parameters:
c depth(t2) is the only parameter - it is passed
c as the single variable "d"
c
c variables:
c      undpro - total underground production (g/cm^3)
c      kdist - controls the decay of the root production function
```

```
c
      undpro = 0.12
      kdist = 0.40
c
c
c this section calculates root production at a particular depth
c based on the parameters/curve that are designated above.
c
c
      rtprod = (exp(-kdist*d)*(kdist)*undpro)
      return
      end
c
c ***********************************************
c *          DECMP1                             *
c * Decomposition of "youngest" organic material *
c ***********************************************
c
c
c the next section is the decomposition subroutine FOR 1st year org matter
c it is a function that gives a decomposition rate (from 0 to 1)
c based on the depth of each section.
c
c decomp is a RATE (units g lost/g present) so it has
c to be multiplied by the organic mass (org) of each section
      real*8 function decmp1(d)
      real*8 mx1, kdec1, d
c
c as with the production function the only thing that determines
c this is the depth.
c
c Variables:
c     mx1 - maximum rate of decay for this age class
c     kdec1 - k for exponential decay curve for this
c            age class decompostion curve
      mx1 = 0.6
      kdec1 = 0.06
c
      decmp1 = (exp(-kdec1*d))*mx1
      return
      end
c
c
c
```

```
c ***********************************************
c *          DECMP2                             *
c * Decomposition of "medium" organic material  *
c ***********************************************
c
c
c
c the next section is the decomposition subroutine FOR 2nd year org matter
c it is a function that gives a decomposition rate (from 0 to 1)
c based on the depth of each section.
c
c decomp is a RATE (units g lost/g present) so it has
c to be multiplied by the organic mass (org) of each section
c
      real*8 function decmp2(d)
      real*8 mx2, kdec2, d
c
c as with the production function the only thing that determines
c this is the depth.
c
c Variables:
c     mx2 - maximum rate of decay for this age class
c     kdec2 - k for exponential decay curve for this
c             age class decompostion curve
      mx2 = 0.3
      kdec2 = 0.06
      decmp2 = (exp(-kdec2*d))*mx2
      return
      end
c
c
c
c ***********************************************
c *          DECMP3                             *
c * Decomposition of "oldest" organic material  *
c ***********************************************
c
c
c
c the next section is the decomposition subroutine FOR old org matter
c it is a function that gives a decomposition rate (from 0 to 1)
c based on the depth of each section.
c
c The rates are LOWEST for this group of organic material
```

```
c
c decomp is a RATE (units g lost/g present) so it has
c to be multiplied by the organic mass (org) of each section
c
      real*8 function decmp3(d)
      real*8 mx3, kdec3, d
c
c as with the production function the only thing that determines
c this is the depth.
c
c Variables:
c     mx3 - maximum rate of decay for this age class
c     kdec3 - k for exponential decay curve for this
c            age class decompostion curve
      mx3 = 0.19
      kdec3 = 0.195
c
      decmp3 = (exp(-kdec3*d))*mx3
      return
      end
c
c
c ****************************************************
c *         PORESP                                   *
c * Subroutine for determining changes in pore space *
c ****************************************************
c
c the next section calculates the pore space for each section.
c This is where changes due to compaction occur. I am assuming that
c all of the pore spaces are filled with water, and that any compaction
c is due to the loss of water and decrease in pore space volume.
c Pore space is assumed to be a function of the amount of material
c (both organic and mineral) that is in a given section, as well
c as the mass above that particular section.
c
c a = totorg(t2)
c b = min(t2)
c c = densabv(t2)
c d = oldorg(t2)
c e = min(t2)
c
c k1 is a constant that affects the curve for compaction
c k2 affects the relative importance of organic versus mineral
c matter in determining pore space.
```

```
c K2 > 1 - organic matter more important
c k2 < 1 - organic matter less important
c k2 = 1 - organic and mineral matter the same
c
c
c p1 & p2 are just temporary variable to make calculations easier.
c
      real*8 function poresp(c)
      real*8 c,k1
      k1 = 170
      poresp = 1-(c/(k1+c))
c     k2 = 1.0
c     poresp = (1/(1+(k1*c)))
c
c everything below here has been commented out in order to
c "simplify" the calculation of pore space.
c
c     write(6,2005) a,b,c,d,e
c     if (((k2*d)+e).le.0) then
c            p2 = 1
c     else
c            p2 = sqrt(((k2*a)+b)/((k2*d)+e))
c     end if
c     poresp = p1*p2
      return
2005  format ('porespace loop', 5(f12.4, 1x))
      end
c
c
c **********************************************************
c *                    RTEMIN                          *
c * Subroutine for determining the rate of mineral sed input  *
c **********************************************************
c
c
c this section calculates the amount of mineral sediment
c input each year - based on the relative elevation of the
c marsh surface.  Y is the relative elevation of
c the core at at given time.
c tdrnge is the tidal range in meters
c mhw is the relative elevation of mhw in meters (0 in this case)
c
      real*8 function rtemin(y)
      real*8 tdrnge, mw, y
```

```
      real tdhght
      tdrnge = 600.0
      mw = 0.0
      tdhght = (y-mw)/(tdrnge/2.0)
      if (tdhght .le. 0.0) then
             rtemin = 1.0
c     write (6, 2001) y, rtemin
      else
             rtemin = 1 - (min(tdhght, 1.0))
c     write (6,2002) y, rtemin
      end if
2001  format (' if part of loop, relelv, rtemin', f6.2, f6.2)
2002  format ('  else part of loop, relelv, rtemin', f6.2, f6.2)
      return
      end
```

VITA

John Charles Callaway was born September 16, 1962 in Santa Monica, California. He attended the University of California, Berkeley for his undergraduate education and completed a double major in Biology and Slavic Languages and Literature. He attended school in Poznań, Poland from 1985 to 1986 as an exchange student, studying Polish language and literature. John received an M.A. degree from San Francisco State University in the Department of Biology in 1990, where he evaluated the impact of an introduced species of cordgrass in San Francisco Bay. He started his studies towards his Ph.D. at LSU in 1990 and will continue to work in wetland ecology at San Diego State University, as a postdoctoral research associate.

DOCTORAL EXAMINATION AND DISSERTATION REPORT

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Title of Dissertation: Sedimentation Processes in Selected Coastal Wetlands From the Gulf of Mexico and Northern Europe.

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June 20, 1994