OCS Study
MMS 87-0120

Causes of Wetland Loss in the Coastal Central Gulf of Mexico

Volume II: Technical Narrative

U.S. Department of the Interior
Minerals Management Service
Gulf of Mexico OCS Regional Office

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Causes of Wetland Loss in the Coastal Central Gulf of Mexico

Volume II: Technical Narrative

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PREFACE

The northern Gulf of Mexico is the site of intensive and extensive oil and gas recovery efforts in state and federal lands. These activities take place amidst an enormously rich concentration of natural wealth, including nationally significant amounts of wetlands, fish, waterfowl, fur, and wildlife. Although wetlands sustain most of this natural wealth, they are converting to open water at historically high and alarming rates. The spatial and temporal juxtaposition of man's activities and these wetland changes led to several independent and smaller efforts revealing major management issues not addressed by agencies on a large scale. The university research community and supporting infrastructure that developed most of the baseline information was poised to attack this complex issue when this study was suggested and as the expectation arose that better resource management would result through improved understanding.

The Science Review Board members remarked that this would be a benchmark study, and we would like to believe that it may prove useful for many. For some researchers, it was a first involvement in a project of this scope, funding, or level of interdisciplinary interactions. This project was an opportunity to seriously test hypotheses, examine neglected data sets or develop new data, models, and techniques. It was certainly the best funded project yet addressing the subject matter. We hope to have laid to rest many myths, opened new avenues of research, and contributed to the public welfare through additions to our knowledge.

Reviews of scientific and business productivity reveal that ordinarily the most productive situations have a staff with high morale. Morale may seem difficult to quantify, but usually we think of an appreciated sense of self-worth, dedication, and understanding of clearly stated goals as contributing to good morale. Curiously, the amount of high-tech equipment may be of lesser importance, and a dearth of such may be more than compensated for by a surfeit of good morale. We should recognize that just as scientific advances these days are unlikely without good financial continuity and quantity, advances are also unlikely without competence and effort. Both need to be nurtured, and we sought to do that in this project.

Group morale reflects the interest of individuals who are rarely of uniform interest, personalities or style. We attempted to acknowledge the intelligence of the various individual situations by keeping in contact with members through individual, working group, whole group meetings, and with various mechanisms. Meeting frequency and duration varied, too. Some researchers just don't always respond to memos. Others are better at science than managing their budgets, while still others have a great overall perspective, but under-developed attention for details. We had to believe that each individual's situation reflected what his general environment allows and encourages. It would be a mistake to ignore the messages of the PI's own experience. To force a narrow administrative and management style would be a foolish attempt to restrict the group's potential intelligence in favor of order, but against creativity, energy, and morale. That narrow approach would also have been administrative suicide. Thus, we attempted to be flexible while retaining some minimum structure to realize the necessary deadlines, scientific rigor, needs for cross-communication, budgetary responsibilities, and synthesis. We also had to be careful about not over-extending ourselves due to insufficient vision, experience, leadership examples or outright laziness. A successful effort for us was thought to be a tenuous balance between relaxing and striving and between perspective and detail.

The project management and the Minerals Management Service (MMS) wanted what the PIs wanted, that elusive, yet final, answer. So, rather than deciding what the PIs should do, the project management sought experienced people to advise what should be done and what their interests were. In many cases we offered our suggestions first. We then sought a mutually-agreeable relationship to work on the substantiative issues thus identified and worked with them as the project developed. It was more important, I think, to work successfully on part of a large issue than to identify a large problem and apply an unwieldly approach.

MMS provided leadership by funding this study and acting on various suggestions of the Scientific Review Board, the study scientists, and other outside reviewers. We enjoyed two MMS technical coordinators over 27 months. Dr. Norman Froomer helped set the tone of the project in a professionally delightful way for the first 15 months and assisted in supplemental funding efforts. Dr. Robert Rogers made a smooth transition into the last 12 months and carried the project to conclusion with our confidence. Mr. Carroll Day, the MMS contracting officer, moved in the bureaucratic world of inter- and intra-agency interactions, negotiated for supplemental funds for two additional studies, and kept the paperwork in order without too much fuss on his part.

It is one of the benefits and purposes of our collegial work situation that so many are open-minded, willing to help, and constructive. Universities house a myriad of diverse talents and energies. Many people simply volunteered or were cajoled or bribed with a field trip out of the office. The persons listed in the Acknowledgments assisted in this project. We are grateful to all of them and hope they learned as much about the landscape, their colleagues, and south Louisiana as we did.

A Science Review Board (SRB) was appointed as an advisory panel for the development of methodology, mobilization of study, and final critique. By offering periodic review on the progress of the project, the SRB contributed enthusiasm, guidance, and thoughtful criticism. We deeply appreciate their participation.

Although the successes belong to everyone, I would like to thank the following three people for completing their work well, with a constructive attitude: Dr. Donald Cahoon, Science Manager, Mr. Rodney Adams, Business Manager and Ms. Jami Donley, Report Coordinator. In addition, we all have a wider circle of friends, colleagues, and students who helped directly and indirectly as humorists, cooks, teachers, listeners and guides throughout this project. I hope we have been able to give back to them something useful to the long-term management of these resources.

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Program Manager
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GLOSSARY

accretion, vertical - vertical increase in marsh surface, expressed as cm yr^{-1}

accumulation, mineral - the deposition of inorganic sediment matter on the marsh surface, expressed as g cm^{-2} yr^{-1}

accumulation, organic - the deposition of organic matter (mainly plant material) on the marsh surface, expressed as g cm^{-2} yr^{-1}

aggradation, marsh - building of a surface of the marsh as a result of continuous or intermittent mineral sediment deposition and organic matter accumulation, expressed as g cm^{-2} yr^{-1}

bulk density - g of dry soil or sediment (organic and mineral) per cc of substrate

disparity, surface - a difference in marsh surface-water surface elevations resulting when the rate of relative water level rise exceeds the rate of vertical marsh accretion

impacts, direct - those man-induced activities directly linked to the physical conversion of one habitat type to another (e.g., dredge and fill activities)

impacts, indirect - those man-induced activities indirectly linked to the physical conversion of one habitat type to another (e.g., dredging may lead to intrusion of salt water that indirectly leads to wetland loss through its effect on plant productivity and health)

land loss - transformation of any land habitat into open water, caused when relative water level rise exceeds vertical marsh accretion; specifically, land loss is defined in Chapter 20 as any cell that changed from land to water between 1956 and 1978 (this is a gross loss because it does not account for land gain)

model calibration - a process to adjust the physical parameters with different forcing functions to compare the model results with field data

plant stress - reduction in plant growth and/or reproduction

relative water level rise - the change in the depth of water over the land at a point in the marsh resulting from the combined effects of eustatic sea level changes, land subsidence and changes in fresh water supply to the marsh

saltwater intrusion - the movement of higher salinity water into a less saline environment on time scales longer than the diurnal tidal scale; as defined, this type of saltwater intrusion can be persistent or temporary

submergence - vertical downward movement of the surface of a thickness of soils resulting from the consolidation or density increase of some portion of the sediments over time and basement sinking (i.e., crustal downwarping of the earth)

Chapter 1

INTRODUCTION TO THE STUDY

by

R. Eugene Turner and Donald R. Cahoon
Coastal Ecology Institute

This study concerns coastal ecosystems and how wetland habitats change through the actions of men, industries, and governments and the way these actions interrelate with natural processes. The geographic focus is the northern Gulf of Mexico from East Bay, Texas, to Waveland, Mississippi (Figure 1-1). The purpose of the study was to determine the relationships between wetland loss and Outer Continental Shelf (OCS) development of oil and gas resources. The rationale for the study is that wetlands have societal value, that wetland management is possible, and that improved knowledge is useful to understand, predict, avoid, and mitigate undesirable impacts.

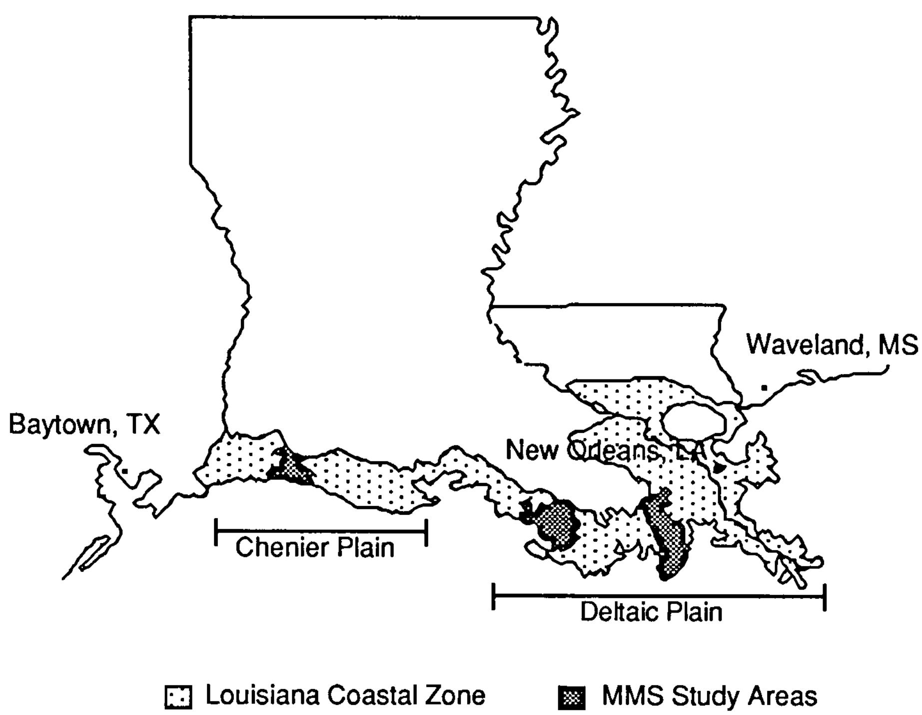

Figure 1-1. The geographical limits of the study area. The three shaded areas were primary study areas for field work.

Water, plants, sediments, soils, landscapes, history, and industry were studied by experts over a 27-month period to develop a consensus report. It was necessary to quantitatively evaluate the contributions of other factors causing habitat alteration to effectively assess the effects of OCS-related activities. Therefore, the study was extensive. Although the study is now complete, every effort reported herein uncovered more questions. The practical implications are enormous for resource management and adaptation.

Statement of the Problem

Coastal wetlands in the Louisiana-Mississippi portion of the study area were converted to open water at an average annual rate of 0.86% from 1955 to 1978, thereby continuing a geometric increase (Figure 1-2). This rate amounts to 288,686 ha for the entire 23-year period. At that rate, the state of Rhode Island would be lost within 21 years, the District of Columbia within 7 years, or within 55 years the Netherlands would lose to the sea all of the land reclaimed over the last 800 years. This is equivalent to the area of a suburban home (1500 ft^2) being lost in one minute.

There is, naturally, concern about these habitat changes because of the enormous economic, social, geopolitical, and environmental values involved in such massive and rapid landscape alterations (Table 1-1). Louisiana's coastal wetlands comprise 41% of the U.S. coastal wetlands and are a state, national, and international natural resource. These wetlands directly support 28% of the national fisheries harvest, the largest fur harvest in the U.S., the largest concentration of overwintering waterfowl in the U.S., a majority of the marine recreational fishing landings, and a variety of wildlife. More than 70% of the OCS oil and 90% of the OCS gas will continue to come from offshore the study area, move through it, and enter the industrial processing plants that support the entire country (Table 1-2). Though now large, these natural and renewable resources may not sustain us through the next century because of their rapid reduction.

Table 1-1 Values for Louisiana wetlands.

Fisheries:	• 28% of the total U.S. fisheries in volume in 1986 (National Fishery Statistics Program, 1987). • $321,514,000 in dockside value, or 12% of the total dockside value for the U.S. (National Fishery Statistics Program 1987). • 4 of the 10 largest fishing ports are in Louisiana (National Fishery Statistics Program, 1987). • 12,092 fishermen on board and dockside in Louisiana in 1977, or 4.3% of the U.S. total (U.S. Department of Commerce, 1987; National Fishery Statistics Program 1984). • 68,894 commercial fishing applications were filed in 1986 (personal communication 1987, Lucy Hidalgo, Louisiana Wildlife and Fisheries). • 1,000,000 recreational fishermen in Louisiana (personal communication 1987, Benny Jay Fontenot, Louisiana Wildlife and Fisheries).
Fur:	• Bobcats, fox, otter, mink, raccoons, muskrats, nutria, and other trapped species provided over $18,000,000 to the state's economy in 1980-81 (personal communication, Greg Linscombe, 1987, Louisiana Wildlife and Fisheries). • Trapping provided employment for approximately 10,000 people in 1986 (personal communication, Greg Linscombe, 1987, Louisiana Wildlife and Fisheries).
Waterfowl:	• 5,000,000 waterfowl migrate down the central and Mississippi Flyway to winter on Louisiana's 1.5 million ha of coastal marshlands. • 3,000,000 waterfowl were found in a January, 1986, mid-winter survey of the coastal marsh and inland areas of the Mississippi Delta. • 102,000 hunters bagged 1.2 million ducks in 1985-86 (Louisiana Department of Wildlife and Fisheries, 1986).
People:	• Wetlands provide a buffer from storm damage. • Wetlands enhance water quality. • Wetlands provide homes for 1,000,000 people, including the oldest bilingual population in the U.S. (unpublished data, Louisiana Tech University College of Administration and Business, 1987).

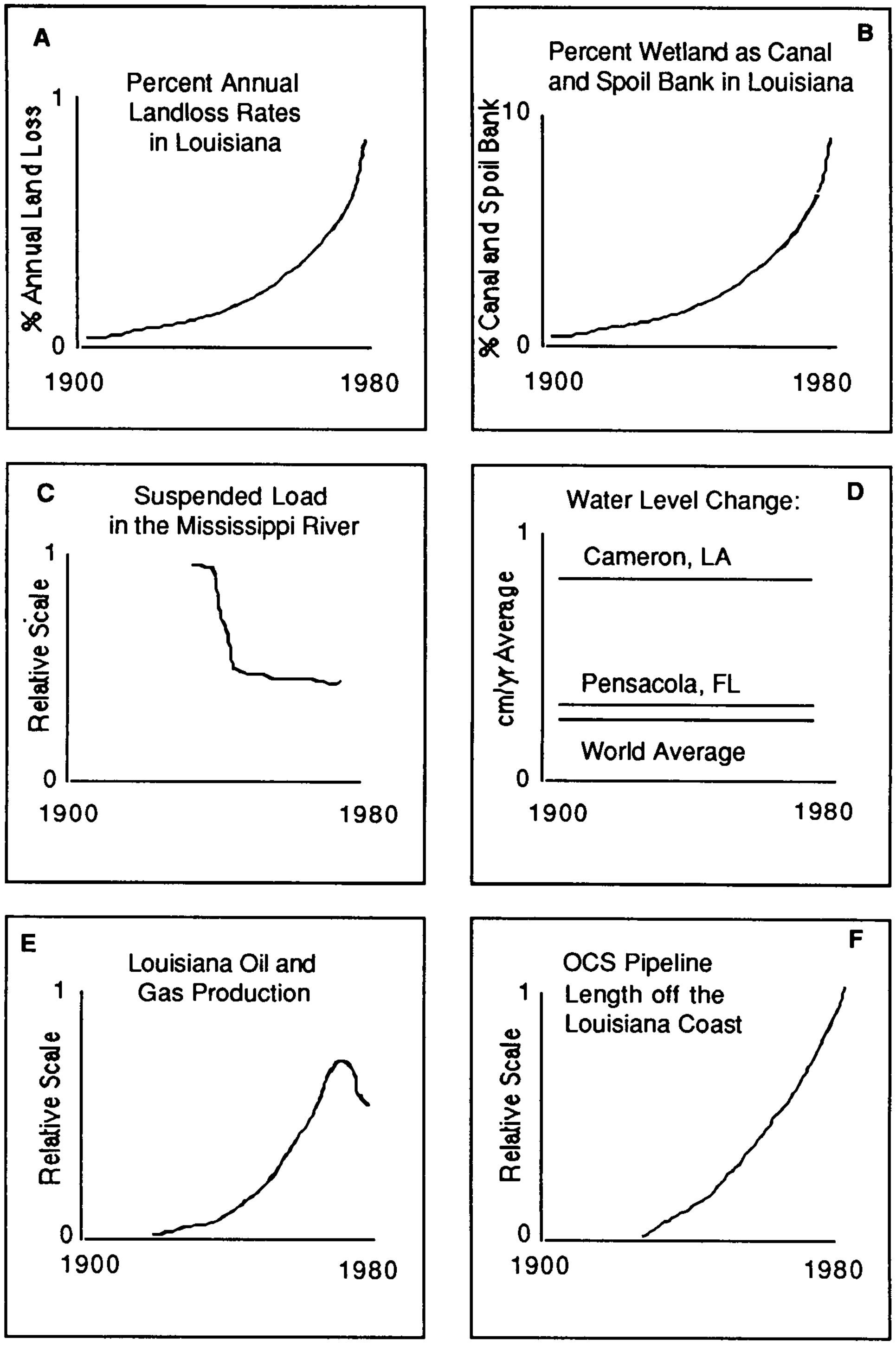

Figure 1-2. Changes in landscape patterns and use in the study area. A. Landloss rates vs time (Gagliano et al., 1981). B. Canal and spoil bank density since 1900 (Turner et al., 1982). C. Suspended sediment concentrations in the Mississippi River since 1950 (Meade and Parker, 1985). D. World, Pensacola, and Cameron water level record changes (Barnett, 1984; Chapter 11). E. LA cumulative oil and gas production (US MMS, 1983). F. Pipeline miles in the Central Gulf of Mexico OCS region since 1950 (US MMS, 1983).

Table 1-2. Summary OCS Statistics for Louisiana and the Gulf of Mexico.

	% LA Gulf of Mexico OCS Total	% LA of U.S. Total OCS	% LA of All
OCS Oil Production			
Crude oil and condensate 1985	94	87	10
Total from 1953 to 1985	98	93	7
OCS Gas Production			
Natural gas 1985	79	78	18
Total from 1953 to 1985	90	94	11
Cumulative Oil and Gas Production			
1985	86	83	14
Total from 1954 to 1985	94	94	12
Estimated OCS Reserves			
Oil 1985	93	71	13
Gas 1985	79	76	19
Oil and Gas Wells 1985	87	84	NA
% OCS Production of Total US			
1. Oil 1985		12	
2. Gas 1985		23	
	% Gulf of Mexico OCS of US Total OCS Lands	%LA of U.S. Total OCS	% LA of All Federal Lands
Revenues to US treasury			
Bonuses 1985	100	56	54
First year rental 1985	70	52	20
Oil royalties 1985	93	88	68
Gas royalties 1985	99	74	64
Bonus from lease sales 1985	100	72	70
Total receipts (rentals, bonus, royalties) 1985	76	73	60

Wetland gains and losses are the results of many interacting factors. In a natural marsh, mineral matter from rivers, reworked sediments, and plant debris is required to build wetlands. At the same time, wetlands in Louisiana's sedimentary coast are sinking and absolute sea level is rising. Any factor that significantly alters that sinking rate, water level rise or accumulation of soils could easily determine whether an area gains or loses wetland to the sea. These relationships are diagramatically shown in Figure 1-3.

Although geologic factors clearly influence the rates of wetland loss, other factors are also important. The suspended matter concentration in the Mississippi River has apparently declined in the last 30 years, probably as a result of land use changes and the trapping of sediments behind upstream dams. Soil accumulation is not simply the result of sediment supply but also of the interaction of plants and the prevailing hydrologic regime. For example, besides trapping mineral matter at the surface, plants add a substantial amount of organic material to the soil. Fresh marsh soils are mostly composed of organic debris deposited *in situ,* not brought in by currents. Even salt marsh soils may be composed of up to 50% organic matter. Furthermore, as organic material accumulates and mixes with minerals, the weight per unit volume of soil decreases. Thus, marshes need less mineral matter than unvegetated bay bottoms to maintain elevation in the face of rising sea level or sinking substrate.

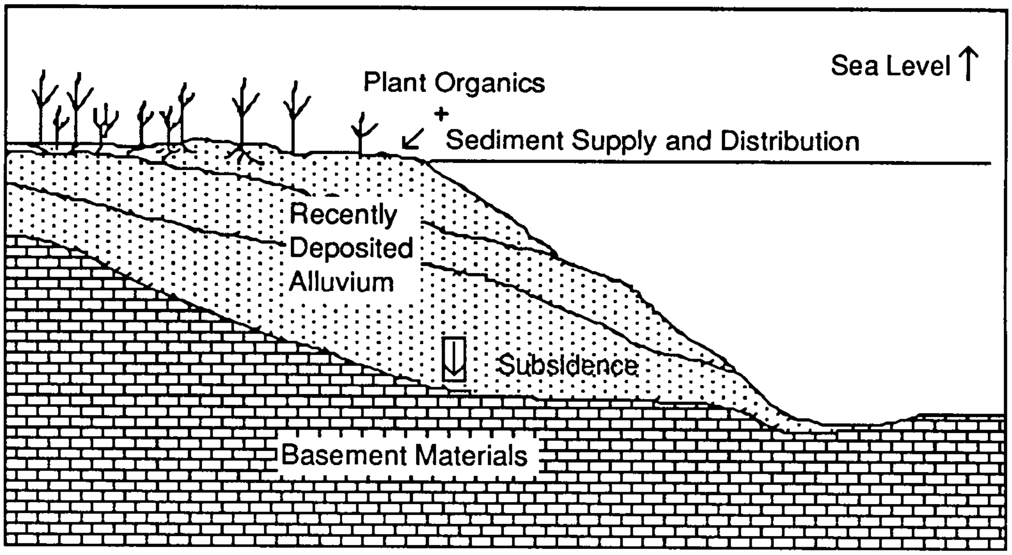

Figure 1-3. Major natural driving forces that affect whether or nor land (wetlands, in particular) maintains its vertical equilibrium or turns into open water. Geologic subsidence (sinking of the land) and rising sea level result in plant flooding. Soil build-up counteracts for this potentially detrimental effect on plant health by accumulating organic materials from both plants and sediments.

Man has changed the landscape in several notable ways that may have contributed to these large habitat changes. Examples are shown in Figure 1-2 and discussed in greater detail in Chapter 3. Direct influences of man include those arising as a result of 40% of the U.S. refining capacity and 8 major fabrication yards being located within the MMS Central Gulf of Mexico planning area (Lynch and Rudolph, 1984). Indirect impacts include canals dredged in wetlands for both OCS and non-OCS related activities. Most canals and their associated spoil banks have been constructed since 1940 to service the oil and gas industry. Each oil and gas field in the coastal wetlands has numerous canals and spoil banks. The canals are dug to bring in drilling equipment, and the spoil banks are the residual dredging materials placed on either side of the canal usually in a continuous and unbroken line.

Offshore and onshore oil and gas annual production rates peaked about 10 years ago (Figure 1-2) and have since declined in spite of the deregulation of prices in the late 1970s. Consequently, fewer canals have been built in recent years, although the cumulative total canal area continues to climb. The average canal dredged in recent years is smaller than previously, partly because of increased scrutiny by state and federal management , but also because the canal network has grown so that new canals can attach to old ones. The current surface area of canals is equivalent to 3.1% of the wetland area. Every hydrologic unit has a significant area of canals that has increased greatly in the last 25 years. Overall, the total area of spoil bank levees plus canal surface is about 6.8% of the present wetland area; increases in spoil and canal area equal 16% of the net wetland change from 1955/6* to 1978. There is hardly a place in the Louisiana coastal zone where canals and their impacts are absent (Figure 1-4).

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* Aerial photography of the Louisiana coast was obtained during 1955 and 1956 (Wicker, 1980, 1981), and both dates have been used in the literature in referring to this data set. For the majority of this report, the maps used were from 1956 photography; however, both dates are given throughout the report to avoid inaccuracies.

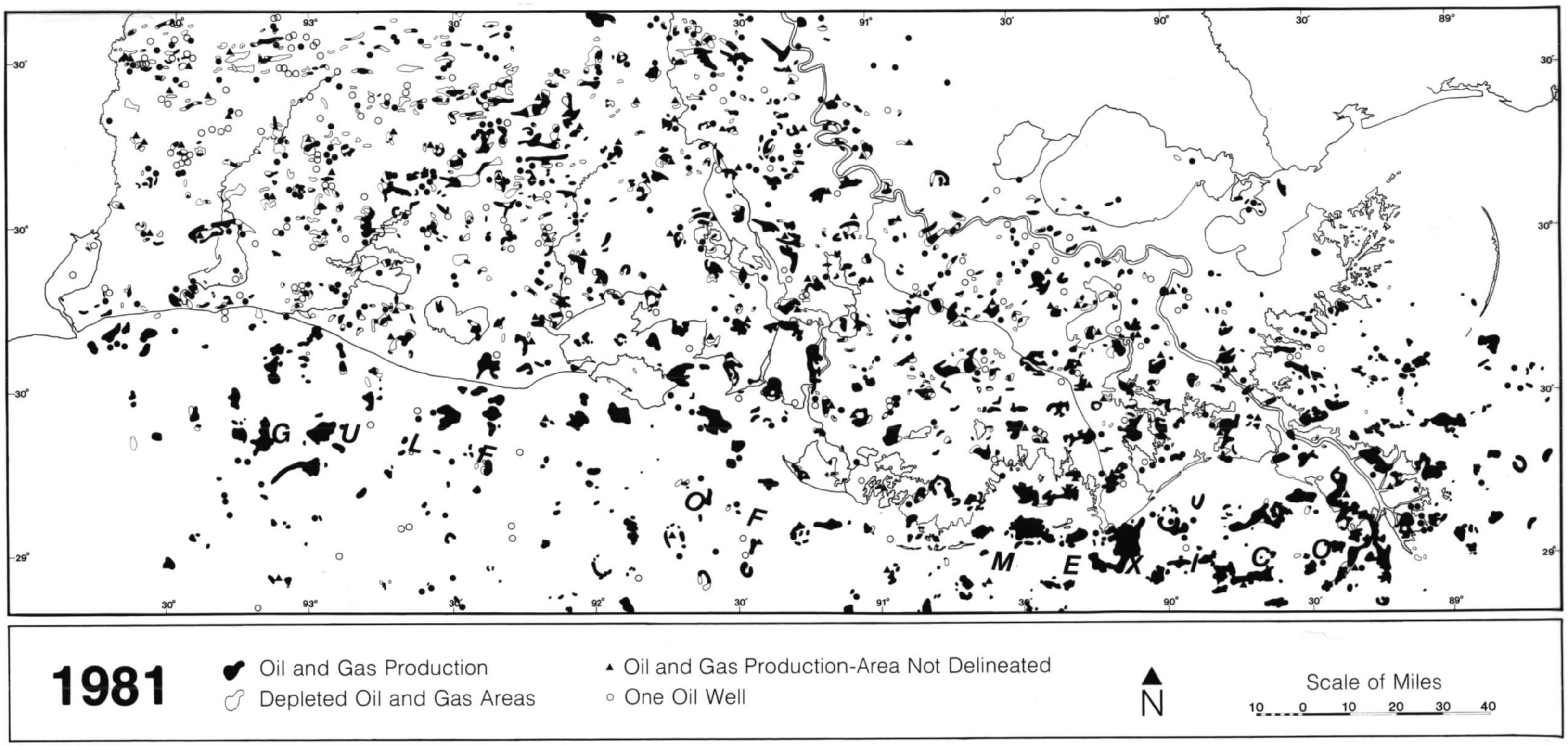

Figure 1-4. Distribution of oil and gas fields in southern Louisiana.

The temporal and spatial concurrence of rapid changes in wetlands and both on- and offshore oil and gas industry activities led to this study. On one hand, oil and gas recovery has previously been extensive and will likely continue to be significant in the future. On the other hand, it was presumed that geologic rates (subsidence, sediment compaction, and sea level) will be constant in the near future. Hypotheses and questions were developed to understand the causes of wetland loss and what could be done to reduce wetland loss rates.

Determining Causes

Two basic questions about present wetland loss rates are: why does it happen at all, and why have the rates accelerated to dramatic proportions? Part of the answer is obvious. We need not have complete understanding of all mechanisms leading to plant death to conclude that some wetland plants will disappear during the dredging of a canal (a direct impact) or that permanent plant submergence and, ultimately, plant death (an indirect impact) will result when a marsh is completely impounded by spoil levees. We see the result clearly enough to accept that one is the result of the other, even though we do not know the proximal or final cause. For example, canal density may be directly correlated with wetland habitat change, but the conclusion that canals lead to wetland loss is incomplete without a mechanism to explain the correlations. Temporal and spatial changes, whether dramatic or not, may be used as guideposts to decipher mechanisms and challenge hypotheses but not necessarily to prove hypotheses. Thus, simple correlations are not sufficient, multiple approaches are desirable, and clear statements of hypothesized relationships and mechanisms are desirable to efficiently develop long-lasting conclusions.

Causal agents of wetland habitat change are many, and the significant ones are listed in Table 1-3 (see Chapter 3 for detailed discussion). The list reflects the presently diverse knowledge of how wetlands form, are sustained, and disappear. Wetland scientists generally agree that because wetlands, by definition, require water, plants, and some soil matrix, the direct or indirect alteration of wetlands affects their ability to adapt. It is the importance of the relationships between the parts and changes that is debated, not whether the list is complete. Some of these relationships are accepted as facts, others are suspected, and still others are being developed or challenged. This research effort was organized to test the strengths of these relationships and conclusions about their factual occurrence.

Some relationships are more prominent than others, and we wish to briefly draw attention to two of them.

(1) Are the relationships constant or dynamic? The former view (constant) is supported, for example, by the idea that geological processes alone drive the present 1% loss rates, that the modern delta is a proper analogue of the whole coast or that sediment supply strictly determines deltaic size and, therefore, plant distribution. The latter view (dynamic) suggests that changes in wetland hydrology significantly influence (1) the role plants play in wetland formation and maintenance, and (2) compensatory movements of sediments between marsh and bay or offshore and estuary.

(2) If saltwater vegetation composition increases, then it reflects saltwater intrusion. If vegetation changes, then wetland loss is more likely. These conclusions assume that the influence of physical factors on plant distribution is not superseded by biologic interactions through interspecific competition, consumer pressure or adaptation to physical disturbance.

Table 1-3. Causes and mechanisms of wetland loss in the study area.

Cause	Primary Mechanism
Direct habitat change	dredging, construction, filling in or over, erosion, prospecting machinery (marsh buggies)
Sea Level Rise	increased flooding of plants
Subsidence increases	
natural	net loss in vertical accretion without compensation
oil and gas withdrawal	accelerated net loss in vertical
soil drying	soil shrinkage, net loss in vertical position
Hydrologic Changes/Effects	
saltwater balance	physiological stress leading to plant community change or death
river levees	restricted sediment supply
sediment sources	decrease in sediment supply caused by less overbank flooding in rivers or marsh; delta switching
canals	change in sediment source and distribution, salinity and water levels; widening; channel theft
spoil banks	change in sediment source and distribution, salinity and water levels; water movement over and under marsh
hurricanes	marsh destruction
boat wakes/waves	bank erosion
Vegetation Changes	
quantity	change in physiological responses to salinity, sediment trapping, organic deposition, flooding
quality	change in organic deposition, sediment trapping, intraspecific competition
Pollutants (brine, drilling fluids and other)	death of plants
Other	
introduced pests	death of plants by parasitic insect (primarily on Alligator weed)
muskrat "eat-outs"	reduced vegetation cover leading to pond formation

Project Goals and Organization

There were three primary project goals for the study:

(1) Determine why the coastal marshes are being lost at a rate approaching 1% annually. Specifically, determine whether the high rate of coastal submergence is caused by an increase in wetland sinking (subsidence), a decrease in wetland building (sedimentation and organic peat accumulation) or a combination of these.

(2) Determine what impact OCS and onshore oil and gas development (particularly canal construction) have on wetland sinking and wetland building processes, and, therefore, what the indirect contribution of such development is to the rate of wetland loss.

(3) Determine to what extent wetland loss in south Louisiana is caused by the direct conversion of wetlands to open water or upland habitats by the dredge and fill activities of man.

As a means of better focusing project efforts toward these goals, the investigation was organized around five major questions addressing two important means of marsh loss: coastal submergence and direct conversion to open water.

(1) If land is sinking more quickly than land is building and the rates of each process are changing, to what extent is this disparity caused by changes in: (1) sediment supply reaching the marshes; (2) organic matter accumulation; (3) subsidence rates; and, (4) water level.*

(2) Do levee construction, canal dredging, and oil and gas production influence the rates of sedimentation, organic matter accumulation, and subsidence in coastal Louisiana? If so, do these impacts contribute to the high rate of coastal submergence?

(3) Are there spatial patterns of land loss, and, if so, what are reasonable cause-and-effect hypotheses to explain these patterns?

(4) How long does it take for a change in subsidence, sedimentation or accumulation to be expressed as wetland loss?

(5) What are the direct and indirect impacts of OCS activities on wetland losses in coastal Louisiana?

Working Groups

The project was subdivided into two broad analyses: direct impacts and indirect impacts. The direct impacts of OCS-related activities were assessed and compared with the direct impacts of other oil and gas and miscellaneous wetland-use activities on coastal wetlands in the study area. Indirect impacts were assessed by investigating how OCS activities affect the natural processes controlling wetland loss and by quantifying wetland loss that is indirectly the result of OCS activities.

Individual research tasks were aggregated into "working groups" to prepare the final selection of sampling sites and the final analysis. These groups have a commonality in their basic thrust and subject matter. A definition of each is listed below, along with the Principal Investigator (PI) directing the project at Louisiana State University. The major working hypotheses or issues for each task are identified below; all hypotheses or issues were addressed as thoroughly as possible with the time and support available.

I. Program Management: Methodology Development

Develop hypotheses, experimental design and methodology for other technical approaches; perform data management, archiving, report and budget coordination and consensus development; establish a Scientific Review Board (SRB) to serve as a technical review group and make recommendations to investigators and MMS concerning methodology, hypotheses, and experimental design (PIs: R.E.Turner and D. R. Cahoon, Coastal Ecology Institute, Center for Wetland Resources).

* The fourth term, water level, was added to the original three factors during the course of the project.

II. Direct Impacts Working Group

Determine the direct impacts of OCS pipelines, navigation canals, and support facilities by quantifying the areal extent of open water areas created, spoil deposits, and support facilities (PI: R. H. Baumann, Center for Energy Studies)

Issues:

A. How are the direct impacts of pipeline construction on wetland loss related to the pipeline diameter, habitat type, depth to the Holocene deposits, geologic age of the substrate, watershed location and pipeline age?
B. Is the initial impact of navigation channel construction on wetland loss directly related to the construction dimensions and techniques?

Ascertain historical, existing, and projected volumes of waterborne traffic moving through OCS navigation canals in order to identify major navigation channels. (PI: A. R. Reed, Ports and Waterways Institute, Center for Wetland Resources)

Issue:

A. Determine the OCS waterborne traffic portion of the total waterborne traffic in the major man-made channel/canal systems along the Louisiana coast.

Determine the direct impacts of non-OCS activities on wetland loss. (PI: R. E. Turner, Coastal Ecology Institute, Center for Wetland Resources).

Issues:

A. Determine the amount that onshore oil and gas activities (non-OCS) account for of all direct impacts resulting in wetland loss for the Louisiana coast.
B. Determine the total direct OCS impacts on wetland loss as a percentage of all wetland losses from all causes.

III. Saltwater Intrusion Working Group

Identify and quantify the degree and extent of saltwater intrusion with and without OCS-related canals that contribute indirectly to wetlands loss. (PI: F. C. Wang, Coastal Ecology Institute, Center for Wetland Resources)

Issues:

A. Does the rate of movement and areal extent of saltwater intrusion differ substantially for OCS versus non-OCS channels/canals?
B. How do the physical characteristics of OCS channels and pipeline canals differ from those of non-OCS channels and canals?
C. How is the rate of movement and areal extent of saltwater intrusion correlated with the physical characteristics of the canal including: tides and waves, wind stress, interfacial stress, bottom stress, major meteorological events?

Analyze the long-term (40 years) salinity records to determine the magnitude of change, relationships with climate and oceanographic forces, and residuals caused by geologic changes. (PI: W. J. Wiseman, Coastal Studies Institute and E. M. Swenson, Coastal Ecology Institute, Center for Wetland Resources).

Issue:

A. Determine how much, if any, salinity levels in Louisiana's coastal marshes changed in recent years in terms of average salinity, variability, and range.

Investigate the effects of increased salinity, increased submergence, and the interaction of salinity and submergence on the dominant plant species in three major marsh types by simulating saltwater intrusion and submergence under field and greenhouse conditions. (PIs: I. A. Mendelssohn and K. L. McKee, Wetland Soils and Sediments Laboratory, Center for Wetland Resources)

Issues:
A. Does an increase in salinity cause a significant decrease in plant growth?
B. Is the impact of higher salinity greater in marshes with increased submergence (i.e., marshes in which mineral and organic accumulation do not equal the local subsidence rate)?

Determine how salt migrates to the interior of the marsh, either by overland flooding or by migration through the interstitial water. A combination of field measurements and modeling were used to investigate groundwater flow by advection and diffusion. (PIs: W. J. Wiseman, Jr., Coastal Studies Institute and E. M. Swenson, Coastal Ecology Institute, Center for Wetland Resources).

Issues:
A. Are belowground water movements normally a significant proportion of total water exchange between marsh and estuary?
B. Do spoil bank levees significantly retard belowground water movements?
C. Is marsh soil salinity directly correlated to salinity levels in adjacent water bodies?

IV. Sedimentation/Subsidence Working Group

Estimate changes in wetland elevation and sea level in coastal Louisiana during the past few decades. (PI: J. N. Suhayda, Ports and Waterways Institute, Center for Wetland Resources)

Issues:
A. Determine to what extent fluid withdrawal has accelerated the consolidation of recent sediments in coastal Louisiana.
B. Estimate natural land subsidence rates.
C. Determine if recent (<50 years) relative sea level changes are dominated by wetland subsidence or sea level rise.

Assess the long-term changes in the sediment discharge of the Mississippi River. (PI: R. H. Kesel, Department of Geography and Anthropology)

Issues:
A. Has the sediment supply of the Mississippi River decreased substantially since 1870?
B. Has the suspended sediment supply decreased substantially since 1870?
C. Has the bedload sediment supply decreased substantially since 1870?

Examine the effect of man's alterations on sediment accumulation, peat formation, soil oxidation, and submergence. Three techniques were used: ^{137}Cs and ^{210}Pb (PI: W. H. Patrick and R. D. DeLaune, Wetland Soils and Sediments Laboratory, Center for Wetland Resources); stable tracers (PI: R. M Knaus, Nuclear Science Center); and, inert clay markers (PI: D. R. Cahoon and R. E. Turner, Coastal Ecology Institute, Center for Wetland Resources).

Issues:
A. Does construction of channels/canals with their associated spoil banks substantially alter marsh accretion processes (i.e., sediment distribution, mineral and organic deposition)?
B. Are OCS spoil bank impacts equal to non-OCS spoil bank impacts.
C. Is the rate of sediment accretion directly related to the age of the canal.
D. Do sediment accretion rates change significantly with distance from the spoil bank.
E. Is there a minimal level of sediment input and peat formation needed to sustain a viable marsh.
F. Are the impacts on sedimentation rate expressed as wetland loss 40 years later.
G. Are sediment distribution and accretion rates strongly influenced by the relationship between canal alignment and direction of surface water flow.
H. Is the rate of sediment deposition proportional to the height of the spoil bank.

V. Landscape Patterns Working Group

Conduct computer analyses of the spatial and temporal wetland loss patterns using remotely sensed data. (PIS: J. M. Hill, Remote Sensing and Image Processing Laboratory and Scott G. Leibowitz, Coastal Ecology Institute, Center for Wetland Resources).

Issues:
A. Is wetland loss primarily driven by man-made features?
B. Are erosion sites primarily located adjacent to canals or far away and less likely next to natural waterways.
C. What is the likelihood of an area undergoing wetland loss if it is near areas of previous wetland loss?
D. Are the spatial trends in wetland loss correlated with spatial trends in impact?

Determine the statistical relationships between wetland loss and the man-made and geomorphic features of the whole coastal zone. (PIs: R. E. Turner and J. H. Cowan, Jr., Coastal Ecology Institute, Center for Wetland Resources)

Issues:
A. Are wetland loss sites primarily located adjacent to canals rather than far away and are these sites located next to natural waterways?
B. Is the rate and spatial pattern of erosion correlated with marsh type and age?
C. Is wetland loss as ponds spatially clumped?
D. Is the size distribution of new ponds non-random?
E. How does canal configuration, as well as density, affect the conversion rate of wetland to open water.
F. Is wetland loss greater in impounded areas than in non-impounded areas?
G. Is the conversion of wetlands to open water related to the depth of the Pleistocene surface.

The following chapters are grouped by subject matter, beginning with background documents about climate (Chapter 2) and landscape changes (Chapter 3). Following that material are sections covering direct impacts, sediments, salinity, and landscape patterns. The results of a two-day consensus meeting to address the five major study questions is included in Chapter 23.

Chapter 2

COASTAL CLIMATE OF LOUISIANA

by

Robert A. Muller and Bruce V. Fielding
Department of Geography and Anthropology

Wetland loss is partially dependent on plant productivity which, in turn, is influenced by climatic influences, such as rainfall, temperature, and wind. Rainfall frequency and duration, as well as wind duration and direction, also determine marsh flooding regimes, water and soil salinities, and the destructive strength of storms. Long-term changes in sea level rise and geologic subsidence rates may be estimated from changes in water level if climatic changes influencing water levels are also known. Climatic events in the study area are summarized here to set the stage for interpreting annual and decade-long changes in estuarine salinity, water levels, marsh flooding, and plant population dynamics.

The coastal regions of Louisiana are part of the large, humid, subtropical climatic region that includes the southeastern United States, extending southward from New York City and the Ohio River valley on the north and eastward from central Kansas and Texas on the west. Within a global framework, similar climates occur in eastern China and southern Japan; in southern Brazil, Uruguay, and northeastern Argentina; in eastern coastal areas of the Republic of South Africa; and over some of the eastern coastal regions of Australia. The humid subtropical climate is characterized as having hot summers and mild winters with precipitation average greater than potential evapotranspiration.

Temperature And Precipitation: Averages and Extremes

The average climatic data for Morgan City, with about 56 km (35 miles) of wetlands between the city and the open Gulf, are used to represent the coastal phase of the humid subtropical climate of Louisiana. The temperature and precipitation data in Table 2-1 show 30-year averages between 1951 and 1980, the current "normal" or standard climatic period of the National Climatic Data Center (NCDC) of the National Oceanic and Atmospheric Administration (NOAA) (NCDC, 1985). The table includes mean daily maximum and minimum temperatures by months, the highest and lowest temperatures recorded in each month during the 30 years, and the average number of days with maximum temperatures of 32°C (90°F) or higher and minimum temperatures of 0° C (32° F) or lower. The table also includes mean monthly precipitation, the greatest and smallest monthly totals for the 30 years, and the greatest daily totals.

In January, for example, the "normal" temperature range is between 18° C (the mid-60s) and 7° C (mid-40s); in July it is between 33° C (the lower 90s) and 22° C (lower 70s). At the coast minimum temperatures are a little higher throughout the year than at Morgan City , but maximum temperatures are a little lower during summer; hence, the daily temperature range over wetland and coastal sites tends to be even less than at Morgan City.

Table 2-1. Average climatic data, Morgan City, Louisiana (1951 to 1980).

	Means		Temperature Extremes (C)		Mean Number of Days		Precipitation (mm)			
	Daily Maximum	Daily Minimum	Record Highest	Record Lowest	90 and Above	32 and Below	Mean	Greatest Monthly	Least Monthly	Greatest Daily
Jan.	18	6	29	-11	0	6	114	333	28	137
Feb.	19	7	30	-8	0	3	117	312	10	119
Mar.	22	11	31	-4	0	0	99	264	23	127
Apr.	27	15	33	4	0	0	119	394	05	231
May	29	18	36	7	5	0	135	401	08	203
June	32	22	38	13	20	0	130	406	23	183
July	33	23	39	16	24	0	206	516	74	142
Aug.	33	22	38	13	23	0	183	409	56	117
Sept.	32	21	37	8	14	0	157	457	25	142
Oct.	27	15	34	2	3	0	94	249	0	180
Nov.	22	10	33	-2	0	1	109	297	10	114
Dec.	19	7	29	-11	0	3	130	300	38	157

Arctic outbreaks are one of the climatic hazards of coastal Louisiana. Figure 2-1 illustrates the Arctic outbreak of the 1961-62 winter at Houma, Louisiana. The figure shows that the temperature fell to -11° C (12 F), but that maximum temperatures were well above 21° C (70 F) a few days before and after the outbreak. Three similar outbreaks have occurred in the 1980s, but temperatures this low normally occur only once every 10 to 15 years (McLaughlin, 1986). These severe Arctic outbreaks can be devastating to subtropical crops, plants, and landscaping, and marine life in the coastal estuaries, in addition to the economy, especially in terms of frozen and broken water lines in residential and commercial buildings and industrial complexes.

Table 2-1 also shows that the mean annual precipitation at Morgan City is 159 cm (62.5 inches). Analyses of the geographical patterns of rainfall show that Morgan City is located within a narrow east-west zone of maximum rainfall parallel to the coast, with annual averages in the zone also decreasing westward to 140 cm (55 inches) at Lake Charles. Although there are only very limited rainfall data near the coast, it appears that annual totals there may be as much as 5 to 10% less than in the zone of maximum rainfall. Mean monthly rainfall peaks during July and August when the precipitation mechanisms include frequent but very localized thermal showers and thundershowers from mid-morning to early evening, as well as more widespread rains associated with disturbed tropical weather systems from the Gulf. A secondary peak occurs during winter when the precipitation is caused by frontal activity associated with cold fronts sweeping across Louisiana and by midlatitude cyclones (low-pressure systems) moving generally eastward from the northwestern Gulf toward the Ohio River Valley or the Carolinas. Spring and late fall normally experience less rainfall because frontal weather and disturbed tropical weather occur less frequently and because thermal heating tends to be less effective.

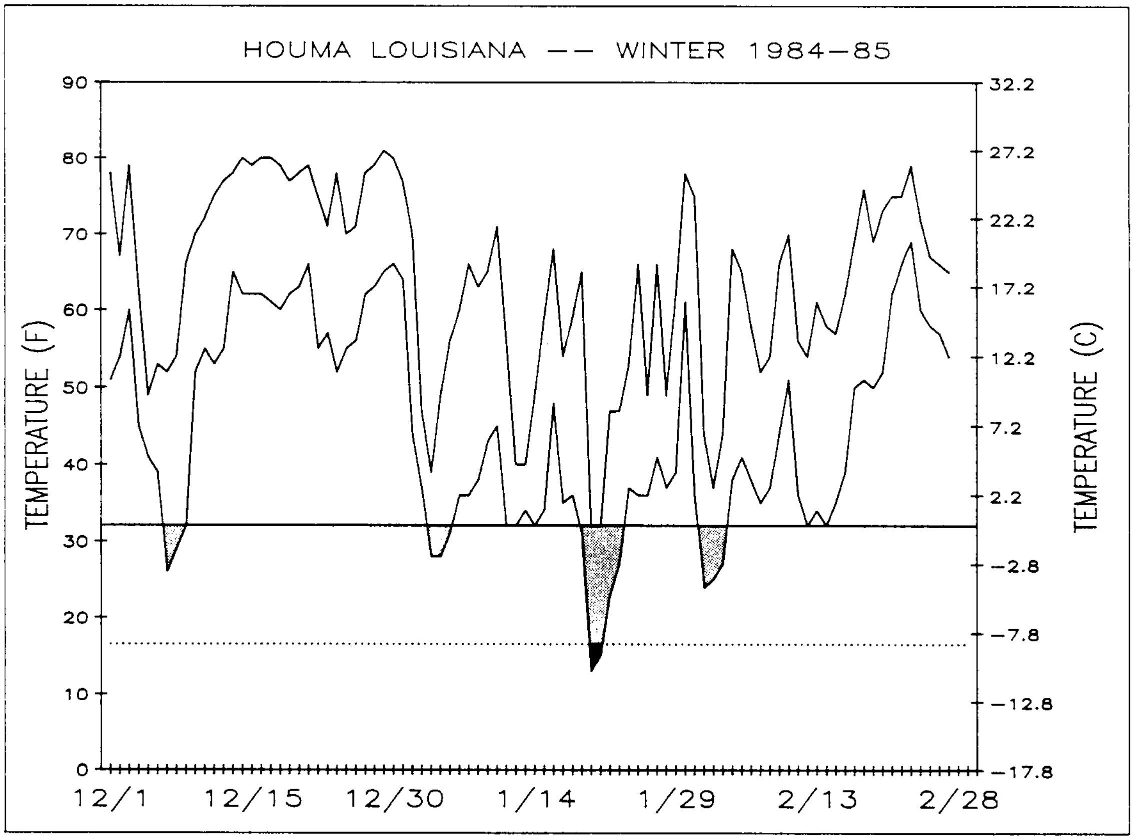

Figure 2-1. Daily maximum and minimum winter temperatures for Houma illustrate record-breaking Arctic outbreak in mid-January, 1985 (McLaughlin, 1986).

Table 2-1 also shows that the maximum monthly rainfall total for the 30 years at Morgan City was to 51.6 cm (20.3 inches) recorded during July, 1964; the minimum monthly total of 0 was recorded during October, 1952, and again in October, 1963. The maximum monthly rainfall recorded in Louisiana from 1900 to 1985 is 96.5 cm (38.0 inches) at Lafayette during August, 1940 (Fournerat, n.d).

The daily maximum rainfall of 23.1 cm (9.1 inches) for the 30-year period at Morgan City was recorded on April 13, 1980, during frontal weather. Climatic analysis suggests that the maximum 24-hour rainfall in the vicinity of Morgan City is expected to be about 13 inches, approximately once in a hundred years; similar values are to be expected westward along the coast to Houston, but higher estimates of up to 38 cm (15 inches) are shown for the outer Mississippi River delta country in the vicinity of Venice (Hershfield, 1961).

The maximum daily total recorded officially in Louisiana is 56 cm (22 inches) during disturbed tropical weather at Hackberry, southwest of Lake Charles, on August 29, 1962. Our preliminary analysis also indicates that the greatest single storm event rainfall of 76 cm (30 inches) was recorded at Lafayette between August 6-11, 1940, when a hurricane swept slowly westward along the Louisiana coast and then northward across eastern Texas. It should be noted that the excessive rains characteristic of southern Louisiana can be produced by both tropical and frontal weather systems, with the potential present throughout the year.

Synoptic Weather

The climate of southern Louisiana also has been organized into eight synoptic weather types expressing typical lower atmospheric circulation patterns that produce local weather and impact environmental, biological, and economic systems (Muller, 1977). Representative examples of the types relative to the weather at New Orleans are illustrated in Figure 2-2; brief descriptions of each type are given below.

Pacific High (PH): fair weather and mild temperatures on northwesterly winds following Pacific cool fronts.

Continental High (CH): fair weather and cool or cold temperatures on northerly to easterly winds in association with continental polar or Arctic air masses following cold fronts.

Frontal Overrunning (FOR): stormy weather, precipitation and cool to cold temperatures with mostly northerly or easterly winds with a cold front or stationary front to the south over the northern Gulf of Mexico.

Coastal Return (CR): mostly fair weather and mild temperatures with east or southeasterly winds.

Gulf Return (GR): mostly fair but warm to hot humid weather and scattered showers, especially in spring and summer with southerly winds.

Frontal Gulf Return (FGR): stormy weather, including showers, thunderstorms, and occasionally severe weather, with mostly southwesterly winds ahead of approaching cold fronts and south of quasi-stationary fronts over northern Louisiana.

Gulf High (GH): mostly fair weather with mild to hot temperatures and southwesterly winds.

Gulf Tropical Disturbance (GTD): stormy weather associated with disturbed tropical weather systems from over the Gulf including weak easterly waves and severe hurricanes such as Camille, with wind directions dependent upon locations and tracks of the disturbances.

These synoptic weather types were developed by interpreting daily synoptic weather maps published by the National Weather Service (NWS) and hourly observations at New Orleans International Airport published by NCDC in Local Climatological Data, New Orleans. Classification data began on January, 1961, and continues to date; 0600 hours normally represents the coolest time of the day and 1500 hours close to the warmest. Average properties of the weather types by months at New Orleans for 20 years between 1961 and 1980 have been compiled, and Tables 2-2 and 2-3 illustrate the average properties for January and July, respectively (Muller and Willis, 1983). For example, Table 2-2 shows that the average temperature at 0600 hours in January, during Continental High weather is about 2° C (35 F), but it averages in the low 60s during Gulf Return weather in the same month. The variation of average properties of the weather types in January, especially in conjunction with the curves representing the annual regimes of incoming solar radiation by weather types at Lake Charles in Figure 2-3, suggests the potential impacts on environmental, biological, and economic systems.

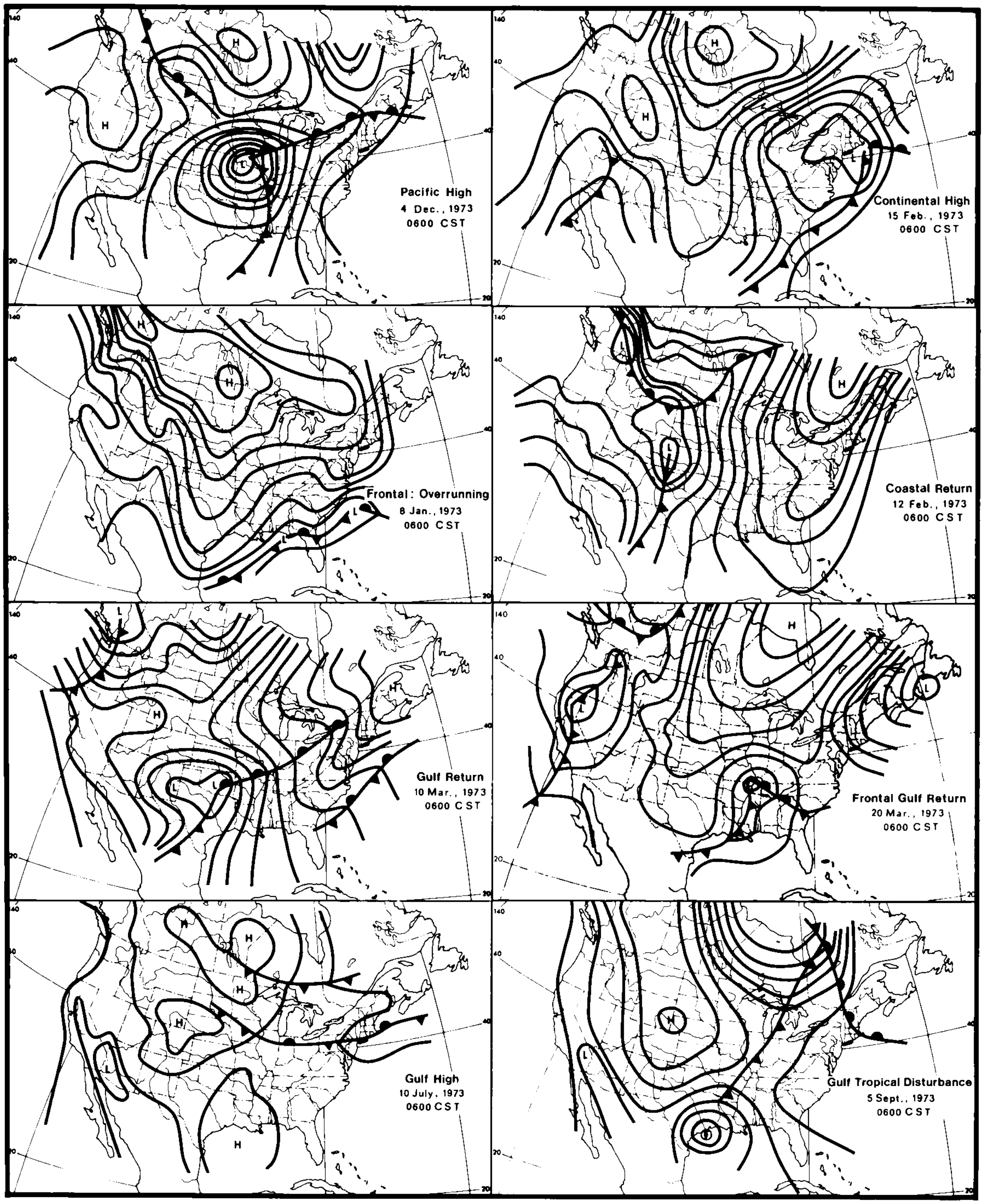

Figure 2-2. Eight synoptic weather types for Louisiana and the central Gulf Coast (Muller, 1977).

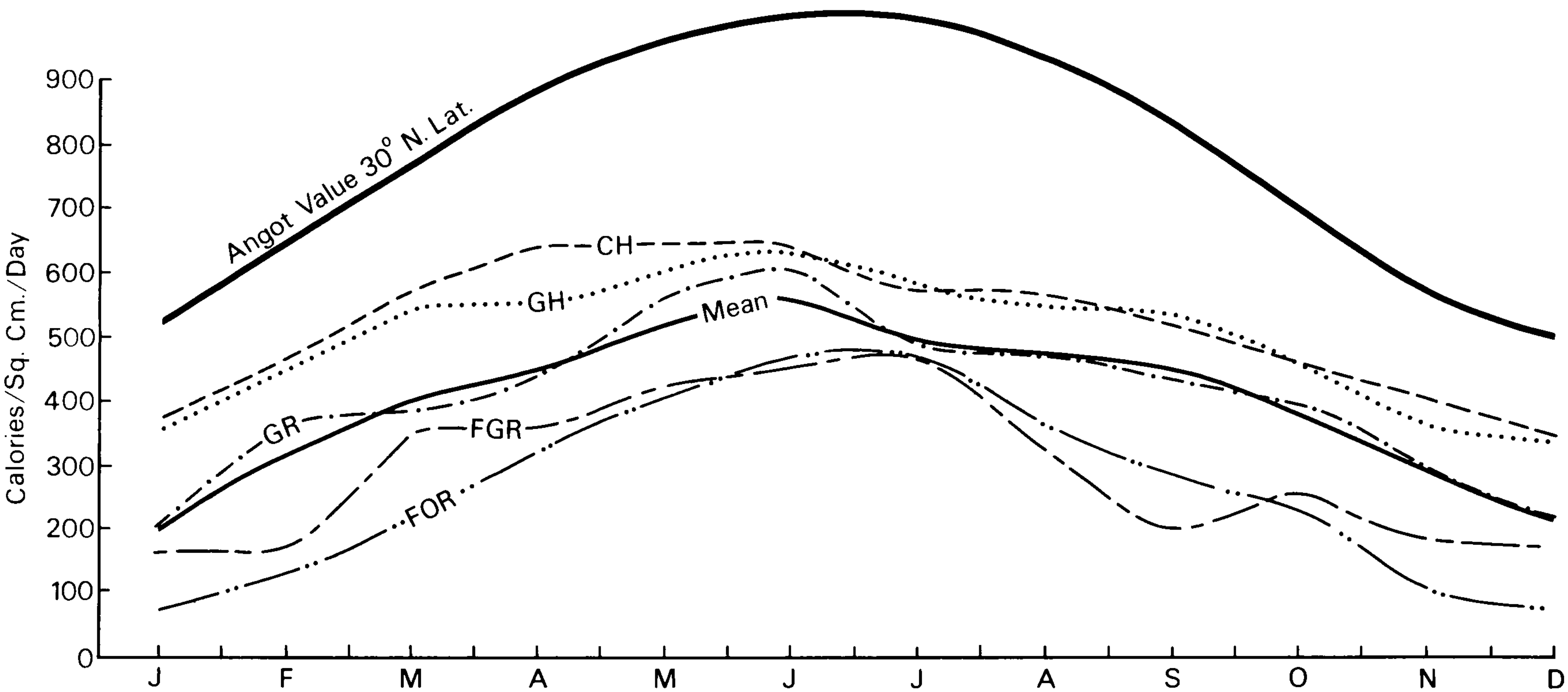

Figure 2-3. Mean monthly solar radiation over southern Louisiana by synoptic synoptic weather types (Muller and Willis, 1983).

Table 2-2. Mean properties of synoptic weather types recorded for January 1961-1980 (Muller and Willis, 1983).

0600 CST	PH	CH	FOR	CR	GR	FGR	GH	GTD
Number of Cases	22	143	246	40	67	75	27	0
Sky Cover 0-10[a]	3	1	10	5	9	10	3	0
Visibility (km)	13	16	11	10	6	8	13	0
Air Temperature (C)	8	2	7	9	16	17	4	0
Dewpoint Temperature (C)	6	-2	4	8	15	16	2	0
Relative Humidity (%)	86	76	82	89	94	92	90	0
Wind Direction[b]	32	02	02	09	15	18	23	0
Wind Speed (m/s)	2	4	4	3	3	4	1	0
1500 CST								
Number of Cases	25	132	234	44	72	90	23	0
Sky Cover 0-10[a]	2	2	10	6	7	9	3	0
Visibility (km)	16	18	11	13	13	11	16	0
Air Temperature (C)	18	11	11	17	22	21	17	0
Dewpoint Temperature (C)	7	-1	5	9	16	17	4	0
Relative Humidity (%)	51	46	73	61	70	77	44	0
Wind Direction[b]	30	36	02	09	16	18	26	0
Wind Speed (m/s)	5	5	5	5	5	5	4	0

[a] 0 (clear) to 10 (completely cloudy).
[b] tens of degrees from true north; 09 for east, 18 for south, 27 for west, and 36 for north.

Table 2-3. Mean Properties of synoptic weather types recorded for July 1961-1980 (Muller and Willis, 1983).

0600 CST	PH	CH	FOR	CR	GR	FGR	GH	GTD
Number of Cases	0	43	20	74	121	37	261	64
Sky Cover 0-10[a]	0	4	9	5	6	7	5	8
Visibility (km)	0	8	10	10	11	10	10	8
Air Temperature(C)	0	23	24	23	24	24	24	24
Dewpoint Temperature(C)	0	21	21	22	22	23	23	23
Relative Humidity (%)	0	87	87	93	90	91	91	91
Wind Direction[b]	0	02	01	06	15	23	27	08
Wind Speed (m/s)	0	2	2	1	1	1	2	2
1500 CST								
Number of Cases	0	38	19	69	131	47	245	71
Sky Cover 0-10[a]	0	4	8	7	8	9	6	9
Visibility (km)	0	13	13	13	14	13	14	13
Air Temperature (C)	0	31	29	31	29	28	31	29
Dewpoint Temperature (C)	0	21	22	23	23	23	23	23
Relative Humidity (%)	0	56	68	66	70	75	63	74
Wind Direction[b]	0	02	33	09	18	21	25	16
Wind Speed (m/s)	0	4	4	4	4	4	4	4

[a] 0 (clear) to 10 (completely cloudy)
[b] tens of degrees from true north; 09 for east, 18 for south, 27 for west, and 36 for north

Mean monthly precipitation by the synoptic weather types is given in Table 2-4 and shows that more than two-thirds of the annual precipitation is associated with the two frontal types, Frontal Gulf Return (41%), and Frontal Overrunning (30%). Gulf Tropical Disturbance weather produces an additional 10%, so that the stormy weather types together account for more than 80% of the annual precipitation. During late fall, winter, and most of spring, the two frontal types account for almost all of the precipitation. However, local thermal showers and thunderstorms generate rainfall in each of the types during summer.

Table 2-4. Mean monthly precipitation by synoptic weather type, in mm, Moisant Airport, New Orleans, Louisiana (Muller and Willis, 1983).

	J	F	M	A	M	J	J	A	S	O	N	D	Year	%
Pacific High	0	1	0	0	0	0	0	0	0	0	0	0	1	-
Continental High	0	0	0	0	0	1	0	1	1	0	0	0	3	-
Frontal Overruning	81	58	46	41	41	8	5	18	25	20	53	64	460	30
Coastal Return	0	0	0	0	5	13	18	25	15	8	0	0	84	5
Gulf Return	5	0	3	13	8	23	33	25	15	5	8	0	138	9
Frontal Gulf Return	61	66	74	58	74	46	30	36	30	33	53	76	637	41
Gulf High	0	0	0	0	0	18	33	25	5	0	0	0	81	5
Gulf Tropical Disturbance	0	0	0	0	5	13	38	23	58	10	3	0	150	10
All Types	147	125	123	112	133	122	157	153	149	76	117	140	1554	100

The average percent occurrence of each of the synoptic weather types by month is shown in Table 2-5. It is somewhat surprising that the two most common types, Continental High (23%) and Frontal Overrunning (18%), are normally associated with northerly to easterly winds and cooler temperatures. In contrast, the sultry Gulf Return (17%) and Frontal Gulf Return (13%) types represent the national image of Louisiana climate; the two types together peak in April when they each occur about 50% of the time.

In Table 2-5 some of the types have been combined into climatic indices. The Continental Index represents cooler air with winds mostly from northerly or easterly components, the Continental High and Frontal Overrunning weather types together. The Tropical Index represents the sultry maritime tropical air from the Gulf and includes the Gulf Return and Frontal Gulf Return weather year round, the Gulf Tropical Disturbance weather in season, Gulf High weather from May through September, and the Coastal Return weather from June through August.

The Storminess Index in Table 2-5 includes the three stormy weather types: Frontal Overrunning, Frontal Gulf Return, and Gulf Tropical Disturbance. The Storminess Index peaks in January when frontal weather over southeastern Louisiana averages a little more than 50%. The frontal weather declines to a minimum of 10% of the time on the average in July, but the Storminess Index shows a smaller secondary peak in September because disturbed tropical weather systems from the Gulf are most frequent at that time. Hence, the Storminess Index is lowest in early summer and again in October when frontal activity and disturbed tropical weather systems normally are not well developed.

Table 2-5. Percentage of hours, synoptic weather types, 1961-1980, Moisant Airport, New Orleans, Louisiana (Muller and Willis, 1983).

	J	F	M	A	M	J	J	A	S	O	N	D	YR
Pacific High	3	7	6	4	5	0	0	0	1	4	3	4	3
Continental High	22	25	20	19	19	19	6	16	28	46	32	26	23
Frontal Overrunning	38	27	23	13	13	7	3	6	14	15	25	31	18
Coastal Return	7	8	8	9	13	12	12	21	17	14	13	8	12
Gulf Return	11	11	21	34	26	25	20	16	13	9	12	10	17
Frontal Gulf Return	14	17	19	16	15	10	7	8	8	7	13	17	13
Gulf High	4	6	2	5	9	23	40	26	6	4	2	4	11
Gulf Tropical Disturbance	0	0	0	0	1	4	11	7	13	1	1	0	3
Continental Index	60	52	43	32	32	26	9	22	42	61	57	57	41
Tropical Index	25	28	40	50	51	74	91	78	40	17	26	27	46
Storminess Index	52	44	42	29	29	21	21	21	35	23	39	48	34

CH+FOR
GR+FGR+GTD+GH(May through September)+CR(June through August)
FOR+FGR+GTD

The climatic variability and frequency stormy weather types can be evaluated by the New Orleans synoptic weather type calendars from 1961 to date. Figure 2-4 shows the individual monthly departures of the Storminess Index from their respective monthly averages for 1961-1986, as well as a moving five-month running average to help highlight more stormy and quiet periods. The figure shows especially the more stormy run of years beginning with mid-1976 and ending about 1981, with much shorter peaks in 1966 and again in early 1975. The Index remained well below normal during 1962 and 1963, one of the driest periods over southern Louisiana in this century, and again for a much longer run between 1967 and 1974. There is no evidence that these anomalies are cyclical, but they are probably related to variability of the general circulation of the atmosphere and especially to "blocking patterns" and El Nino-Southern Oscillation phenomena (Douglas and Englehart, 1981).

The Storminess Index represents a synoptic atmospheric index of stormy weather contrasted with fair weather, and it is probably only moderately related to coastal erosion. Some of the stormy weather days include situations when the winds at New Orleans are only 3 mps (5 knots) or less, with great hurricanes like Camille bringing sustained winds of as much as 72 mps (146 knots) to the outer fringes of the Mississippi River delta and 39 mps (76 knots; peak gust of 49 mps (95 knots) at Lakefront Airport in New Orleans (National Climatic Data Center, 1969).

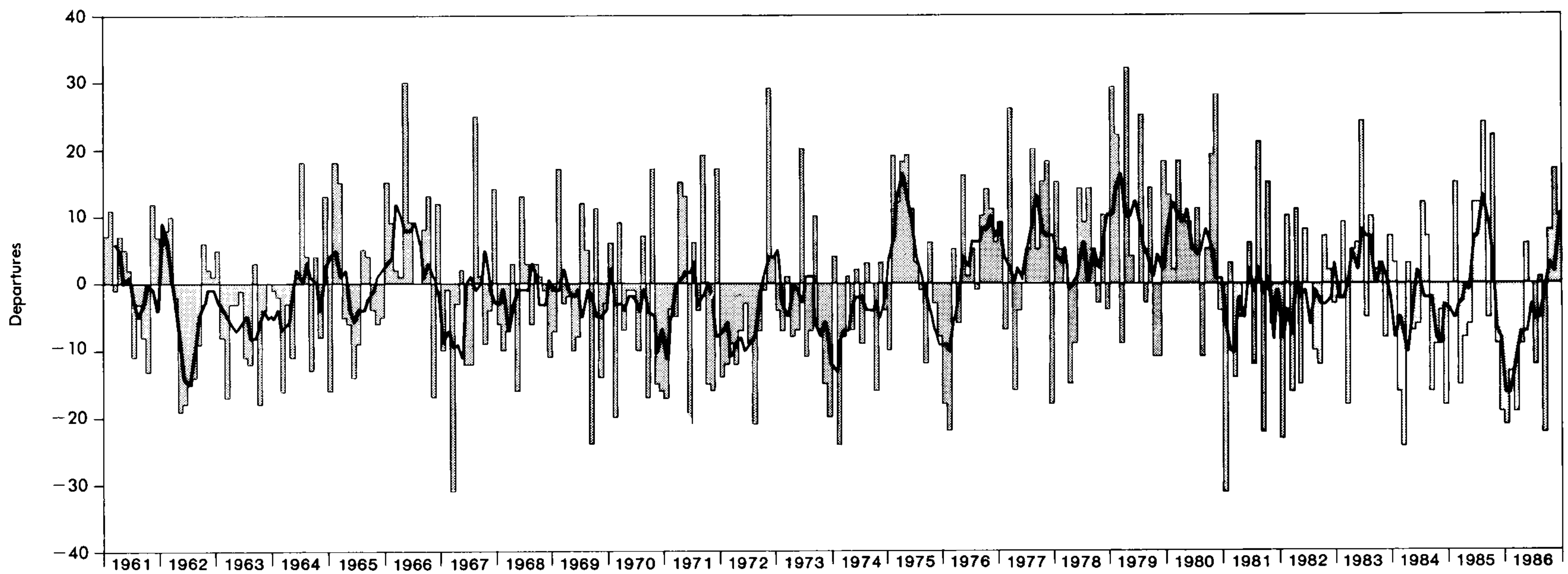

Figure 2-4. Monthly departures and a 5-month running average of the Storminess Index for for New Orleans (modified from Muller and Willis, 1983).

NWS wind data at New Orleans International Airport are published in *Local Climatological Data: New Orleans* by NCDC, and these data have been evaluated to provide some insight into the significance of winter-type mid-latitude cyclones relative to the indirect potential of summer and fall tropical storms for generating storm waves capable of erosion along the Louisiana coast. Table 2-6 is a listing of the number of hours of sustained winds greater than 8.8 mps (17 knots) at the airport by synoptic weather types for each month from 1961 through 1986; for the 26 years, winds equal to or greater than 8.8 mps (17 knots) occurred 1.4% of the time.

Table 2-6. Estimated hours sustained winds equal to or greater than 17 knots[a,b], New Orleans, Airport, 1961-1986.

	J	F	M	A	M	J	J	A	S	O	N	D	Total
PH	15	24	36	30	0	0	0	0	0	0	3	9	117
CH	126	72	51	42	3	3	0	3	3	24	63	75	465
FOR	222	213	147	27	27	3	3	3	6	15	48	123	837
CR	0	0	0	9	12	0	6	3	6	3	3	6	48
GR	15	51	84	210	63	15	6	6	9	12	33	72	576
FGR	54	168	252	174	54	3	0	3	3	18	105	141	975
GH	3	9	9	9	0	15	6	0	0	0	0	0	51
GTD	0	0	0	0	0	6	12	21	135	108	0	0	282
Total	435	537	579	501	159	45	33	39	162	180	255	426	3351

[a] 17 knots equals 8.7 m/s.

[b] based on sample of every third hourly report, for 8 observations/day; 2,920/year; 75,920/26 years.

Table 2-6 shows that 54% or greater of the 8.8 mps (17 knots) winds occurred during frontal weather (Frontal Overrunning or Frontal Gulf Return), and only 8.4 % of the winds above this threshold occurred during Gulf Tropical Disturbance weather. There were only 39 hours with gale-force winds greater than or equal to 14.4 mps (28 knots) and only 15 occurred during Gulf Tropical Disturbance weather: 3 during Hilda in 1964, 9 during Betsy in 1965, and 3 during Camille in 1969. The table is not a complete representation of the climatic potential for wave energy relative to erosion along the Louisiana coast. At the coast, wind speeds are normally higher than inland at New Orleans International Airport, and, at the same time, swells from storms elsewhere over the Gulf of Mexico can generate destructive waves along the Louisiana coast when local winds are less than 5 mph.

Tropical Storms and Hurricanes

Tropical storms and especially great hurricanes, Camille in 1969 for example, are traditionally treated as the most significant storm events along the Gulf Coast. In the synoptic weather-type section above, Table 2-5 shows that Gulf Tropical Disturbance weather occurs from May through November, with peak frequencies at New Orleans in September (13%), followed closely followed by July.

The NWS classifies disturbed tropical weather systems into categories, based on maximum sustained wind speeds in those systems. The most poorly defined systems are termed tropical disturbances; systems are called tropical depressions when maximum sustained winds are between 10.3 and 17.5 mps (20-34 knots); tropical storms have winds between 18 and 33 mps (35-64 knots); and, hurricanes occur when winds are greater than

64 knots. Hurricanes, in turn, are divided into the five-level Saffir-Simpson scale on the basis of maximum sustained winds relative to disaster potential.

Hurricanes have great potential for disaster because of the destructive force of high winds, flooding from excessive rainfalls, and especially flooding from storm surges. The latter represent significant sea level rises normally associated with the northeast quadrants of tropical storms and hurricanes approaching the Louisiana coast from the south (Louisiana Office of Emergency Preparedness, 1985). Figure 2-5 shows the destructive record storm surge of 6.89 m (22.6 feet) at Pass Christian, Mississippi, and the massive flooding of Plaquemines Parish associated with Hurricane Camille. Storm surges are associated primarily with the long fetch of onshore hurricane-force winds within the eastern half of these storms. Figure 2-6 shows estimates of the wind field of Hurricane Audrey prior to coming onshore at Cameron, Louisiana, on June 27, 1957; more than 500 people died partly because of the sudden intensity of the storm system. Figure 2-7 is an example of the precipitation pattern that produced some of the costly flooding during Hurricane Juan in October, 1985.

Appendix A is a summary of all of the tropical storms and hurricanes that have directly affected the Louisiana coast since 1900. The compilation has been developed from annual storm tracks plotted in an atlas of storm tracks (Neumann, 1981) and from annual maps in *Weatherwise* for the most recent years.

Although the Louisiana coast continues to be plagued with relatively frequent tropical storms and hurricanes, the appendix does show that hurricane-force winds are apparently uncommon at any one point along the coast. For example, for the 86 years of record, sustained hurricane-force winds are estimated to have occurred four times at Boothville, seven times at Morgan City, and five times at Cameron.

Long-term Climatic Variability

NWS and NCDC use 30 years of climatic data (by decades) to describe average or normal climate, and the data in Table 2-1 represent the current normals for 1951 to 1980 for Morgan City. Although many but not all climatologists expect some global warming from the increasing carbon dioxide content of the atmosphere over the next 50 years (MacCracken and Luther, 1985), scientists do not completely agree on the details of climatic variability on a regional basis in this century.

Winter temperatures are probably most critical from an environmental perspective in southern Louisiana, and Figure 2-8 shows a statistically smoothed estimate of winter-season temperatures for southeastern Louisiana from 1821 to date (McLaughlin, 1986). The gap in the 1860s occurs because the climatic data taken in Louisiana during the Civil War have been lost (personal communication, David Ludlum, 1985). Nevertheless, the figure shows the tendency for runs or clusters of warmer or colder years, including a remarkable winter cooling that began in 1957 and has persisted ever since except for a short break in the early 1970s. Some of the winters in the late 1970s and early 1980s are among the coldest winters of this century, not only in Louisiana but across most of the eastern two-thirds of the United States. Three mid-winter Arctic outbreaks in the early 1980s drove temperatures down to record lows for this century, causing widespread destruction of citrus orchards, as well as damaging subtropical crops and landscaping and marine nursery stocks in estuaries and shallow coastal waters.

Freshwater runoff is an important climatic control of the coastal wetland environments of Louisiana, and the amounts and timing of precipitation and runoff vary widely. Figure

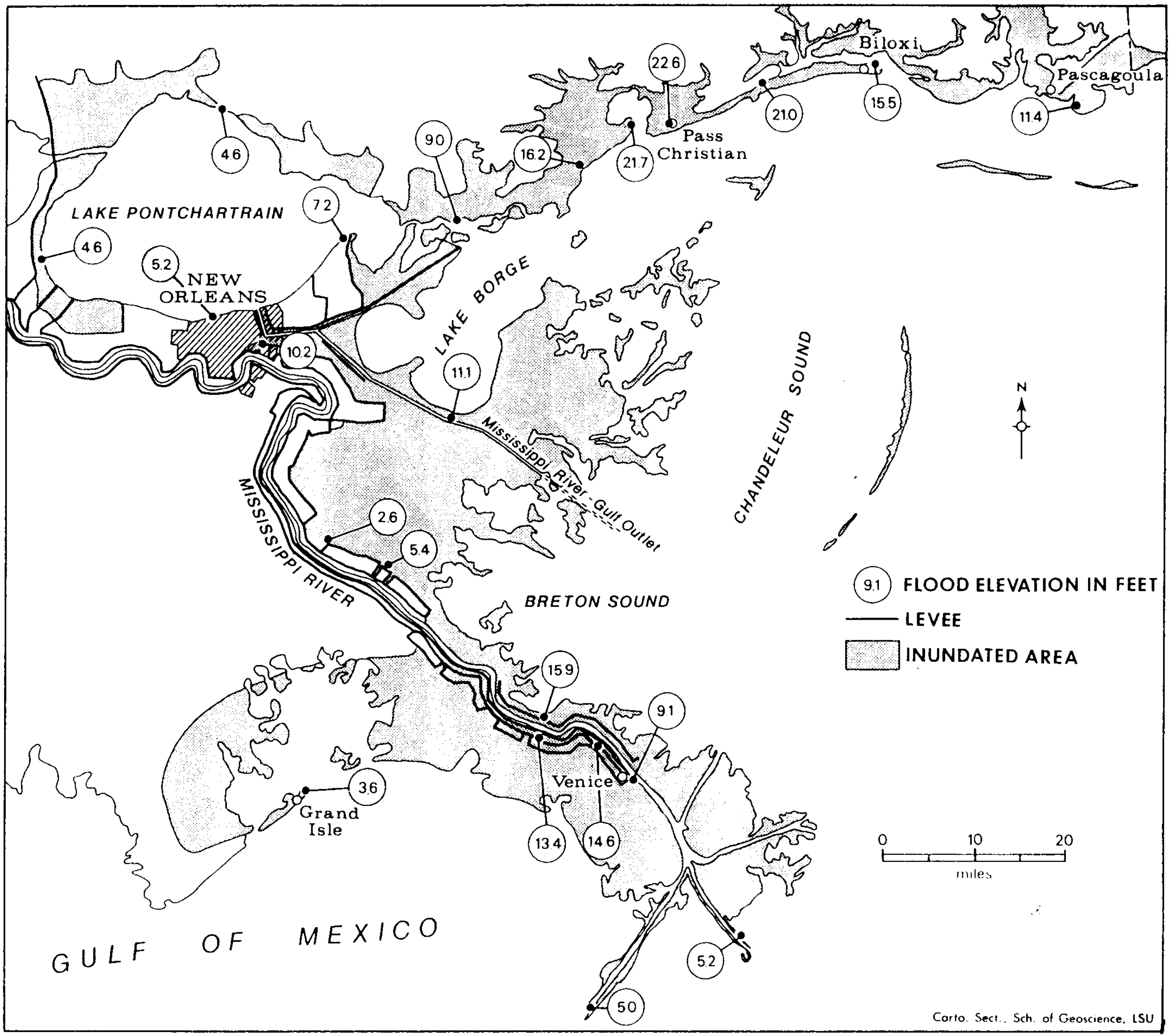

Figure 2-5. Storm surge in feet of Hurricane Camille in August 1969 (Muller and Willis, 1978).

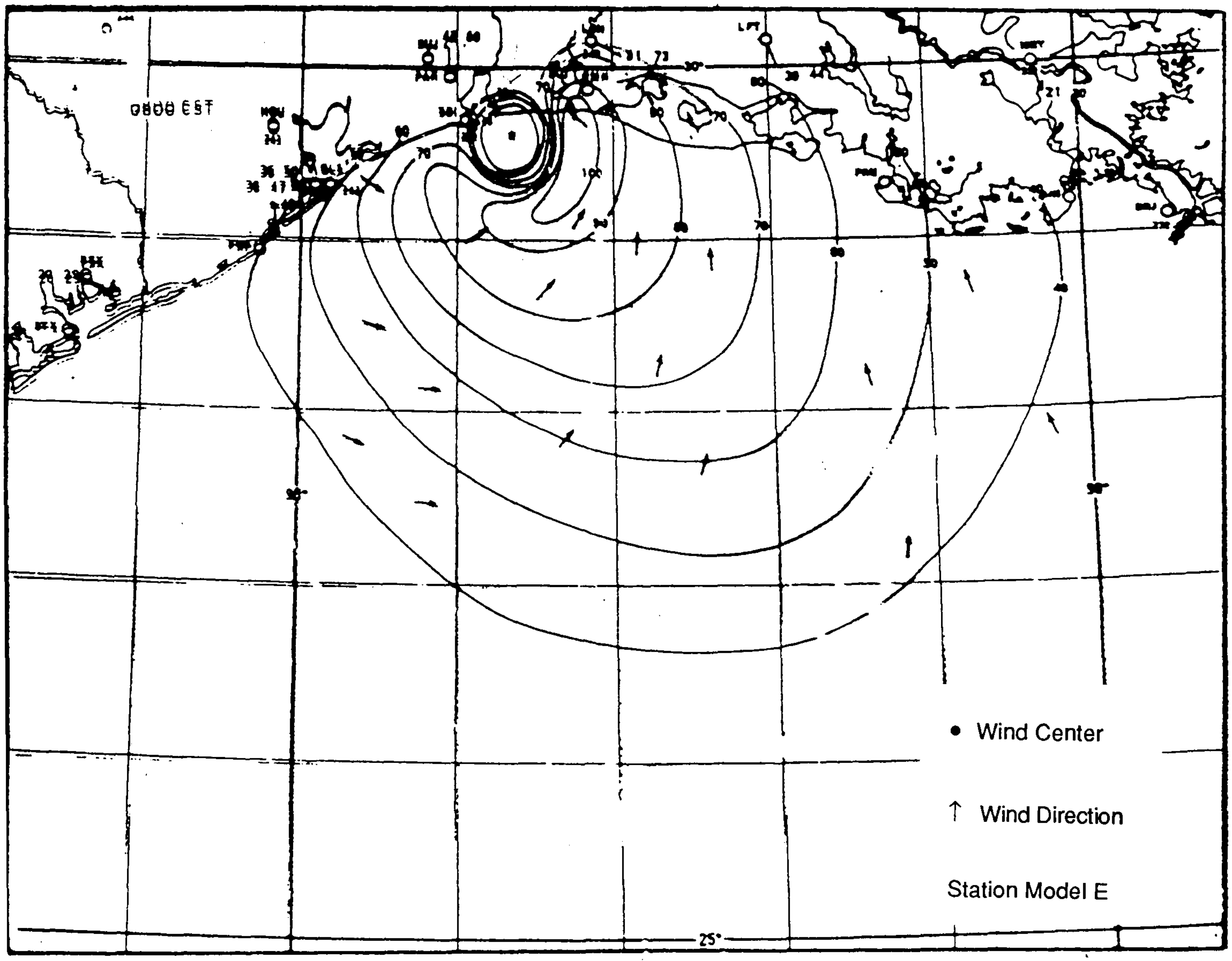

Figure 2-6. Estimated wind field of Hurricane Audrey at landfall in Cameron Parish, Louisiana (U.S. Army Corps of Engineers, New Orleans District, 1958).

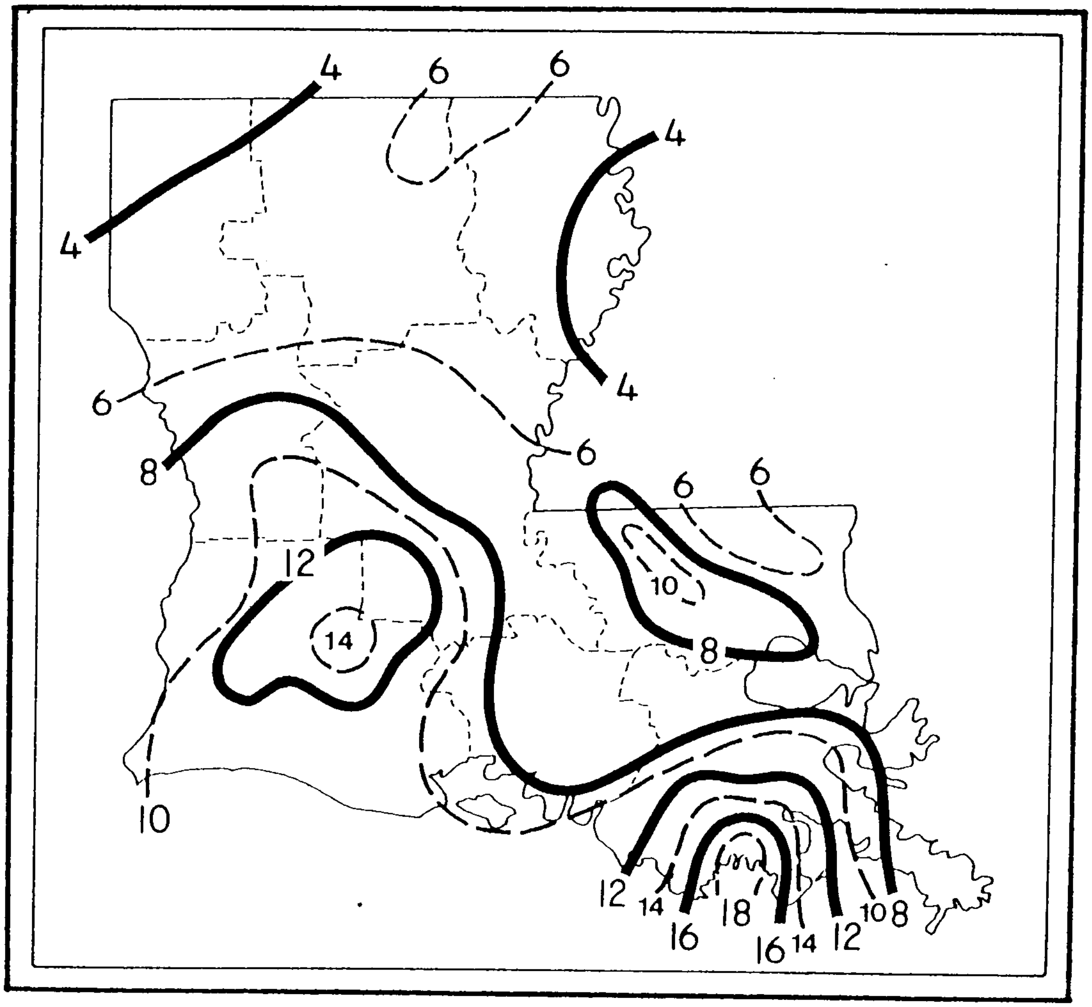

Figure 2-7. Storm precipitation (inches) during Hurricane Juan in October, 1985 (Muller and Willis, 1983).

2-9 estimates a water-budget model of water available for groundwater recharge and runoff for nearly 100 years over the East-central Climatic Division of Louisiana (Muller and McLaughlin, 1986). This climatic division extends east to west between the Mississippi and Pearl Rivers and north to south between the Mississippi border and Lake Pontchartrain. It represents the source area for much of the runoff into the Lake and the other adjacent brackish waters.

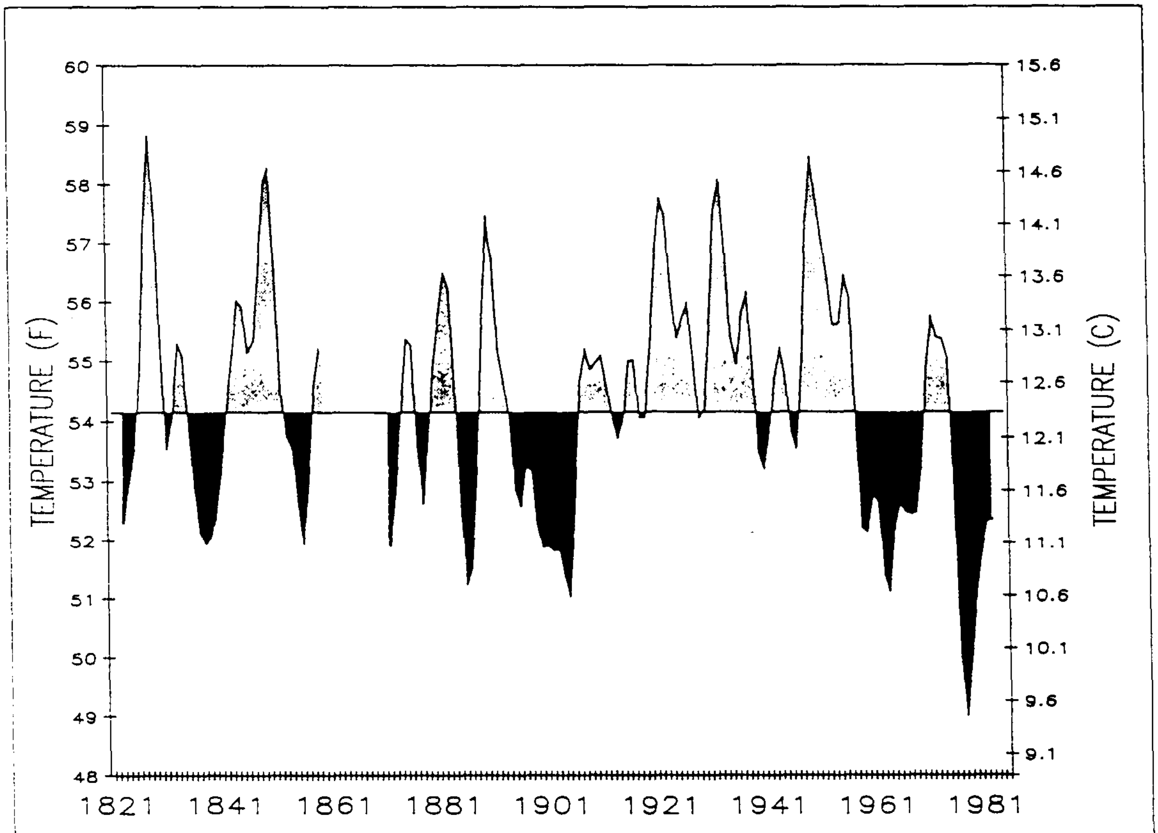

Figure 2-8. Five-year weighted average of winter temperatures in southeastern Louisiana (McLaughlin, 1986).

Figure 2-9 also shows that the mean winter-spring runoff, more than 80% of the annual runoff, averages about 51 cm (20 inches) per season, but the range has been from less than 8 inches in 1911 to more than 102 cm (40 inches) in 1983. The five-year weighted average in the figure provides an overview of the wetter and drier runs of years, with especially wet runs soon after the turn of the century, in the late 1940s, and again in the 1970s and early 1980s. The 1890s and the late 1930s and early 1940s were excessively dry for long periods. The figure shows that the extended wet period of the 1970s and early 1980s has been the longest extremely wet period of the entire record of almost 100 years, and it has been associated with record-breaking costly flooding in southeastern Louisiana (Muller and Faiers, 1984). Since 1983 the climate has returned to a much drier mode, with the exception of extremely wet Octobers in 1984 (frontal weather) and 1985 (Hurricane Juan).

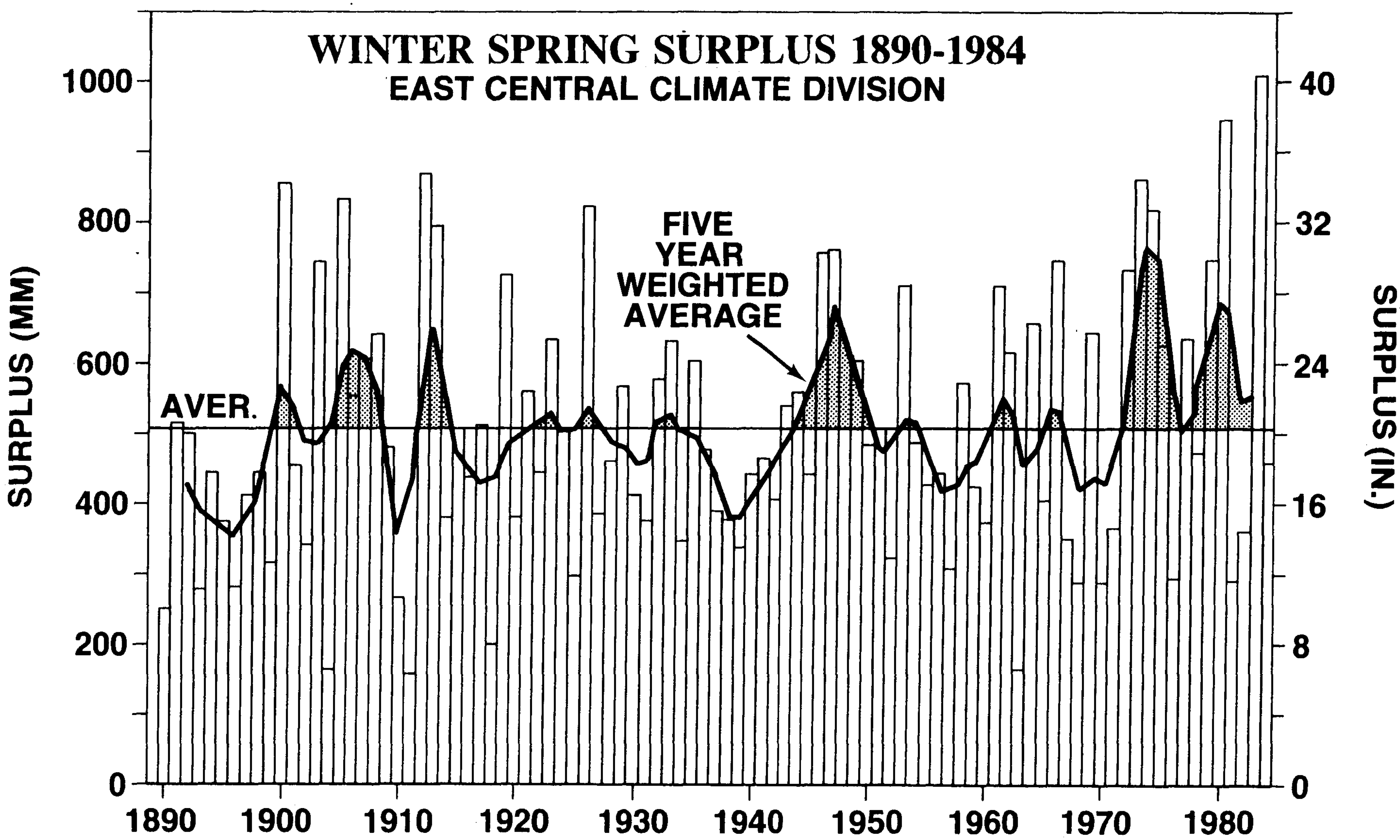

Figure 2-9. Winter-spring moisture surplus (precipitation avaliable for groundwater recharge or runoff) in the East Central Climatic Division of Louisiana, 1890 to 1984 (Muller and McLaughlin, 1986).

Chapter 3

LANDSCAPE DEVELOPMENT AND CHANGE

by

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This chapter introduces the study area in terms of (1) its recent geologic history; (2) the major modifications by settlers; (3) the recent regression from wetland to open water; and, (4) some proposed mechanisms contributing to wetland loss. Quantification of the interactions between the variables contributing to wetland losses are not made. Rather, evidence is presented for different points of view without intentional emphasis on one or more as dominant driving forces leading to conversion of wetlands to open water. Widely accepted points of view are given less coverage than those views generally less well-known. Evaluation of the relative importance of several possible interpretations about open water formation from wetlands is in the individual chapters and consensus review in Chapter 23.

The Study Area

The study area (Figure 1-1) includes mostly the Louisiana coastal zone; the Texas portion is an extension of the Chenier Plain of western Louisiana, and the Mississippi portion is a transition between the Mississippi River Delta and Mississippi Sound. Therefore the generalizations which follow are relevant to the entire study area. The following profile of the area was developed using the 1978 U.S. Fish and Wildlife Service (USFWS) habitat mapping data described by Wicker (1980) after minor corrections to the data base (made by C. Neill, S. Leibowitz, and J. H. Cowan, Jr., Center for Wetland Resources, LSU).

Of the 3.7 million ha of land and water shown in the Louisiana portion of the study area outlined in Figure 1-1, 58% is open water, 5.6% urban and agricultural, 32% wetlands, and 2.1% dredged canals and spoil banks. The wetlands may be combined into five types: salt marsh (15%), brackish marsh (47%), intermediate marsh (23%), fresh marsh (12%), swamp (<1%), and mangrove (<1%). In addition, about 1% is submerged grassbeds, which are sometimes considered wetlands. Louisiana wetlands represent 25 to 41% of all U.S. coastal wetlands, depending on the classification system (Turner and Gosselink, 1975; Alexander, 1985).

Two distinct geomorphic zones, the Chenier and Deltaic Plains, are cut by 11 major rivers, of which 2, the Mississippi and Atchafalaya Rivers, drain about 41% of the United States. New Orleans, the major metropolitan center and second largest U.S. port (by volume), is located on the Mississippi River and Lake Pontchartrain, an oliogohaline estuary. About 2 million people live in the 18 parishes that form the Louisiana study area.

Major economic activities in the study area include (1) oil and gas exploration, recovery, refining, processing, and related petrochemical industries; (2) fisheries; (3) tourism; (4) agriculture (primarily rice and row crops and sugarcane); (5) forestry; (6) fur and alligator harvesting; (7) cattle ranching; (8) transportation and port facility use; and, (9) light manufacturing.

Historical Development

Deltaic Plain

Deltas and their wetlands are the result of multiple interactions as described by Coleman and Wright (1971):

> The sediments composing the delta are, in a literal sense, the "gift" of the river system; but the landscape and environment of the delta are products of much more. The delta is a consequence of the conflict between the river and the sea. ... The delta morphology in detail reflects the totality of hydrologic regime, sediment load, geologic structure and tectonic stability, climate and vegetation, tides,winds, waves, density contrasts, coastal currents , and the innumerable spatiotemporal interactions of all these factors.

The present Louisiana coastal wetlands formed as a series of overlapping riverine deltas extended onto the continental shelf since the Pleistocene (Figure 3-1). The Mississippi River has shifted east and west across the coast seeking the increased vertical gradient and lower hydrologic resistance of a shorter route to the sea. The presently emerging Atchafalaya delta was preceded by at least 6 major deltas over the last 5,000 to 6,000 years. The older delta complex, the Maringouin, also known as the Sale-Cypremort, became inactive about 5,000 years ago when the river switched its position to the east. This process of delta growth and abandonment continued until the position of the modern bird-foot delta was reached about 200 years ago. A new delta is forming at the mouth of the Atchafalaya River, whose flow includes about 30% of the total Mississippi River discharge.

Growth and decline in a new delta is therefore cyclic but is not necessarily symmetrical. In the constructional phase, seaward progradation causes delta muds to be overlain by silts and sands which, in turn, are topped by delta marsh sediments, including organic deposits (Fisk 1960). Smaller subdeltas may deposit sediments up to 14 m (45 ft) thick and the entire delta sequence may be 150 m (491 ft) deep. In the destructional phase, the river abandons its channel in favor of a shorter route to the sea. As the upper layers erode, exposed sediments may be reworked, and marine transgression may occur. As the distributary channels become smaller and the delta is abandoned to be rebuilt elsewhere, the interdistributary ponds, levee flank lakes, and bays enlarge at the expense of the wetlands. The levee gradually sinks into the surrounding marsh which covers it, leaving only a reduced surface expression of its larger buried form.

The abandonment of the delta is not complete, however. Large-scale interlobe basins become isolated by delta lobe switching and gradually build seaward and in different forms from those of the smaller interdistributary bays. Kosters et al. (1987) point out that the physical environments of the two landforms are very different (Table 3-1). Whereas abandoned delta lobes and large-scale interlobe basins are located away from the daily effects of waves and tides and over relatively thin Holocene sedimentary deposits, the active outbuilding system is more likely to be affected by these processes. The swamps are wider, deeper, and over thicker organic deposits in the abandoned lobes, while the progradational delta has fewer large lakes (if any), its levees are closer together, and its sediment sources are directly riverine and often delivered through crevasses. Barrier islands are generally present only in abandoned deltas or seaward of interlobe basins.

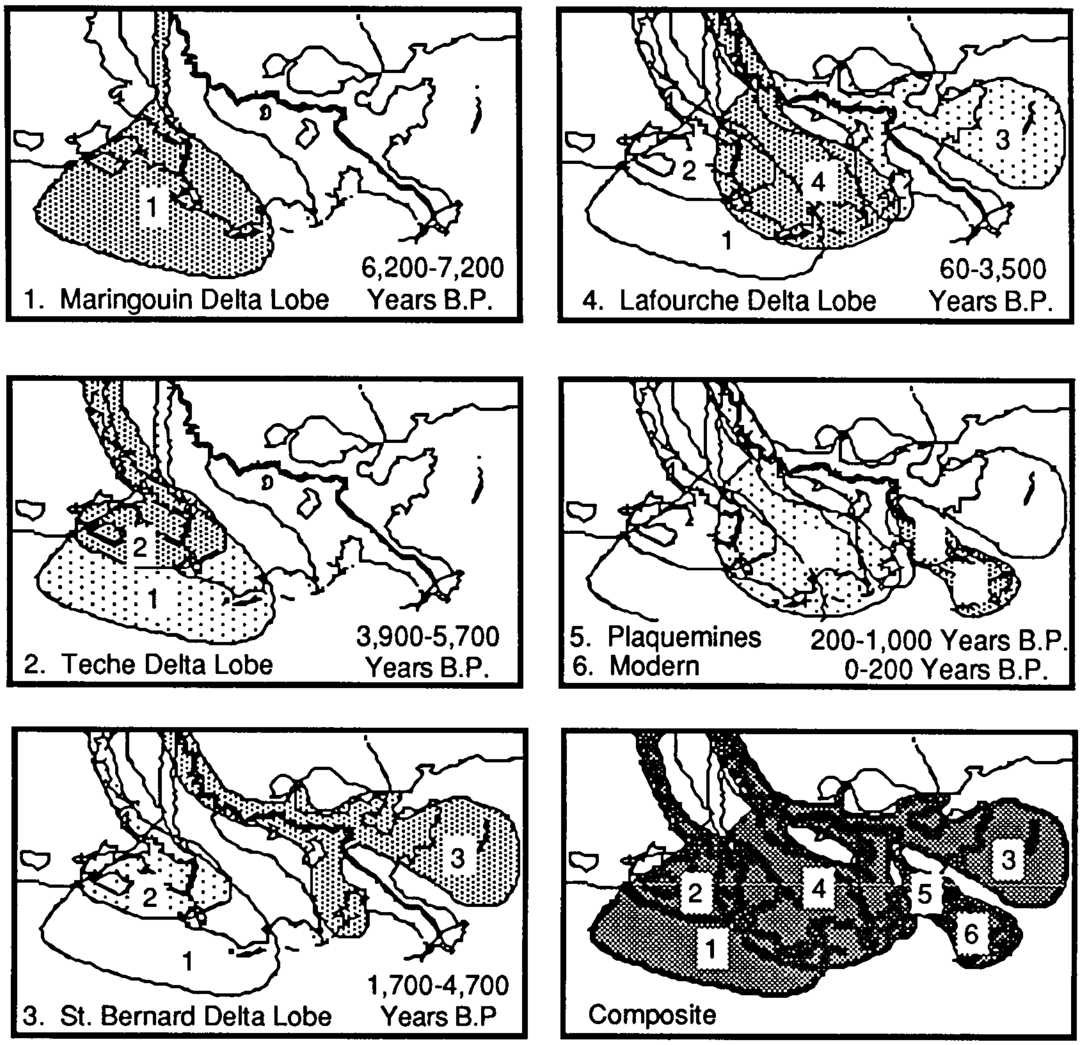

Figure 3-1. The major delta lobes of the study area (adapted from Frazier, 1967).

Table 3-1. Differences between large-scale interlobe basins and small-scale interdistributary bays in the Mississippi River Delta (adapted from Kosters et al., 1987, with additions).

Feature	Large-Scale Basin	Small-scale Bay
Distributaries	More than one deltaic system	One deltaic system
Distance between levees	Maximum of 75 km	Maximum of 20 km
Daily tidal influence	Small	Large
Daily wave influence	Small	Large
Lakes	Possibly long-lived and large	Small, if at all existent
Crevasses	Remnants	Many new ones
Evolution	Over 6000 years	800-1500 years
Sediment Sources	Resuspension, import through tidal passes and from local streams	Riverine
Peats	Extensive and shallow	Rare
Downwarping at Delta terminus	Varies from high to low	High
Barrier Islands	Often present	Absent
Swamps and Marshes	Large, extensive	Thin, restricted

The most recent delta lobe prior to 1950, is the Balize or bird-foot delta. Where the river breaks through the major channel sediments fill in the adjoining bays and built the delta over 200 years. Land gain and loss rates at individual crevasses rose and fell during that period, but the net change from 1820 to present has been a gain at all crevasses and for the total subdelta area (Figure 3-2); land area actually increased from 1972 to 1978.

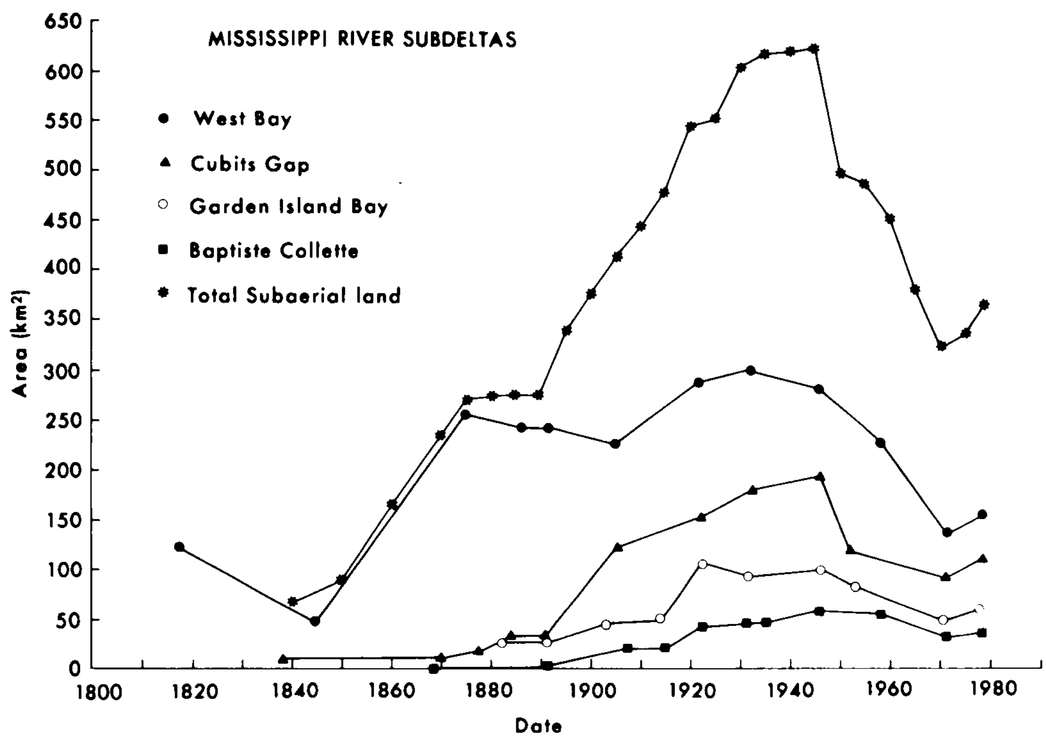

Figure 3-2. Subdelta growth at crevasses at the Mississippi River mouth (Balize delta; adapted from Wells et al., 1982).

Although there has been a net land gain for the last 5-6,000 years over the Deltaic Plain, smaller areas continually go through a cycle of growth and decay. Lake Pontchartrain, for example, has been enlarging since the St. Bernard Delta was abandoned, and the Chandeleur Islands and landward marshes are remnants of the St. Bernard delta abandonment, which started 4,000 years ago.

Chenier Plain

The Chenier Plain, located in the western portion of the study area, is a series of separate shore-parallel to shore-oblique ridges of shell and sand separated by progradational mudflats, or mudflats which are now marshes or open water. The mudflats grow during periods of deltaic abandonment as reworked sediments move westward with littoral drift; subsequent sediment re-sorting builds the ridges. The area is termed a "chenier" in reference to the oak trees (chene in French) growing on the ridges.

Barrier Islands

Louisiana's barrier islands form and evolve with delta growth and decay. However, unlike wetlands which form during the progradational phase of delta growth, barrier island growth is initiated when delta transgression predominates. Barrier islands are generally absent as deltas prograde onto the continental shelf. When the river abandons the main distributary and sediment sources diminish, erosional headlands form with flanking barrier islands (Figure 3-3). The transgressive delta front sands are reworked and move landward forming the barrier islands. At this point marshes are attached to the barrier islands on the landward side with a bay separating the islands from the mainland. Spits may form and some islands fragment to form flanking barrier islands. Without sediments to compensate for the overwhelming influence of subsidence and frontal erosion, the marshes between the islands and the mainland deteriorate and form a bay. Eventually, the island deteriorates completely and is far from the mainland (e.g., the Chandeleur Islands). Shoals are remnants of barrier islands (e.g , Ship, Trinity, and Tiger shoals). Evolution of one barrier island system is generally matched by others so that as one diminishes, another grows. However, evolution will only continue as long as the riverine deltaic cycle is allowed to initiate new deltas at the expense of sustaining others.

Gulf Coast Geosyncline and Subsidence

The large volumes of sediments deposited in the Gulf Basin created the Gulf Coast Geosyncline, and resulted in numerous depocenters whose location changed with glacial advance and retreat, sedimentation pattern, and depth. During the Pleistocene the major depocenter of the Gulf Coast Geosyncline was offshore of southwestern Louisiana. It moved eastward during the Holocene. The axis of the geosyncline is roughly parallel to the present coastline. Accumulated Holocene sediments within the study area range from 0 to 200 m and are generally thickest at the Balize delta lobe (Figure 3-4).

These sediments are mostly fine grained and highly organic. Following deposition they undergo consolidation, compaction, and oxidation. Volumetric changes occur as water is squeezed out from between soil particles. The soil is further reduced by biochemical oxidation of the attached and accompanying organic constituents. All of these processes are termed subsidence or the downward displacement of surface materials without a horizontal component. Vertical change resulting from oxidation and soil shrinkage caused by aeration, usually following drainage, is also defined as subsidence by agronomists.

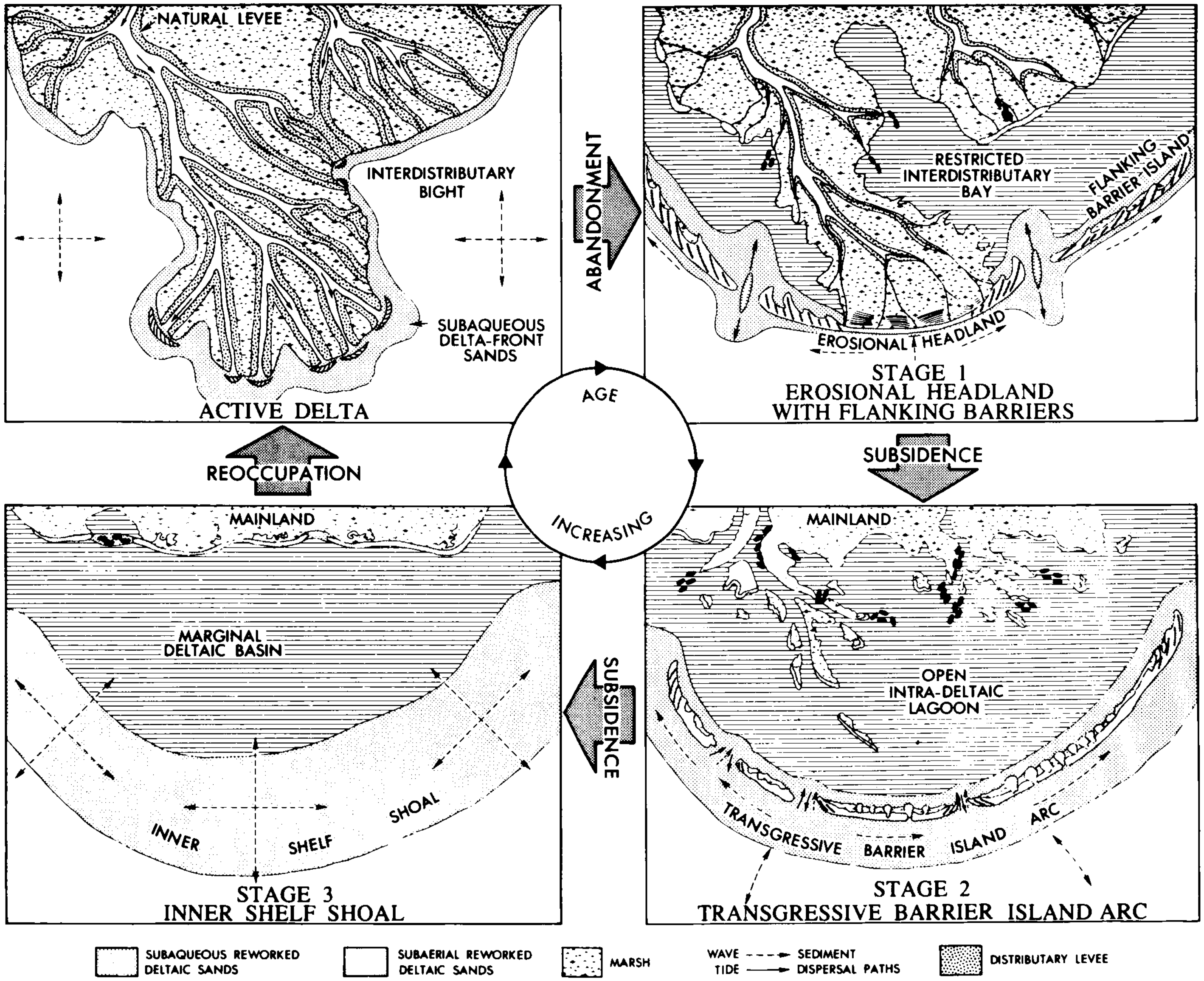

Figure 3-3. An evolutionary model for deltaic barrier island development (Penland et al., 1982).

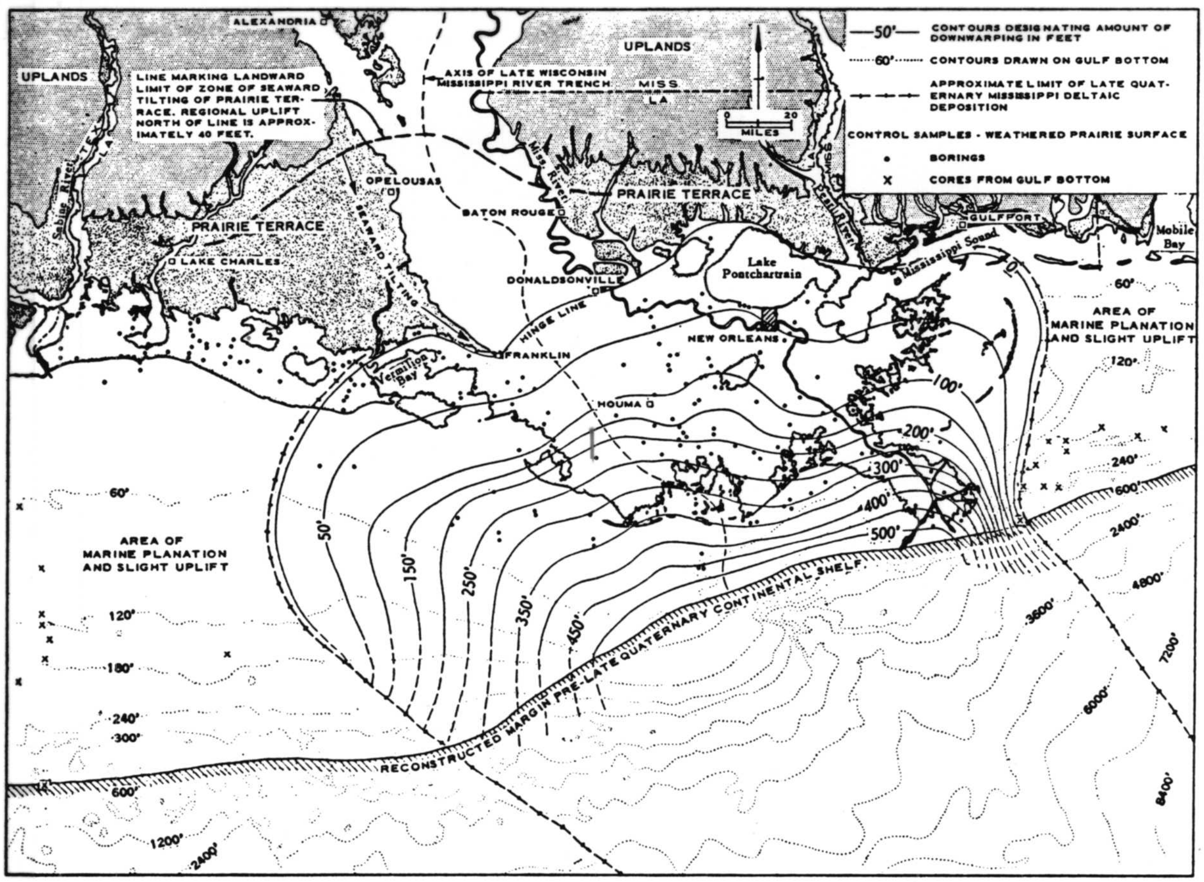

Figure 3-4. Elaborate coastal downwarping in the northern Gulf of Mexico (from Frazier, 1967).

Sea Level

Coastal water levels are not constant seasonally, annually or through geologic time. Water levels change when the total quantity of water in oceans fluctuates or when the spatial or temporal distribution of that water varies among the major water basins. Glacial advances and retreats, climatic events, and oceanic currents are examples of this absolute change in sea level or eustacy. Small changes in the water temperature of a large ocean may also change water volumes and therefore water level. During the Pleistocene, river valleys alternately filled in or were entrenched and the coastal wetlands retreated or expanded, respectively, depending on some natural balance between sediment supply, plant adaptation and growth, and inundation regime. In addition, the vertical position of land with respect to sea level is not constant. Relative sea level rises occur, for example, when the submerged soils compact, reduce through oxidation or removal or move when the underlying basement material changes position.

Absolute and relative sea level began rising from about 100 to 140 m below present levels following the last glacial advance (Late Wisconsin about 30,000 year, B.P.). Fisk and McFarlan (1955) describe sub-aqueous deltaic terraces along this coast at -183, -122, -61, and -30 m below present sea level, indicating that sporadic transgressions occurred when the most recent sea-level rise began 12,000 to 17,000 years ago. Sea level rise has decreased significantly since 5,000 to 6,000 years BP, when the present Chenier and Deltaic plains began to form in Louisiana and Texas.

Catastrophic Events

Shorelines are vulnerable to storm damage. Hurricanes are most likely to strike the coast during summer and fall (see Chapter 2). Hurricanes bring salt water and sediments inland with high water, increase flooding, and may erode land through physical action of waves, wind, and currents. Barrier islands are particularly susceptible to hurricanes. Hurricane Frederick topped Dauphin Island, Alabama, in September, 1979, with a 3.6 m storm tide. Shoreline retreat averaged 15 m there and up to 30 m at the Chandeleur Islands where the storm surge was only 1.3 m (Numendal et al., 1980; Boyd and Penland, 1981).

Recent Areal Changes

For the past 5,000 years there has been net land gain along this coast, together with periods and localized instances of wetland loss; now the rate of loss is approaching 0.8% annually or about 155 km^2 (60 mi^2) per year for the whole Louisiana coastal zone. Even more unfortunate is that the rate is apparently climbing geometrically, as discussed in Chapter 1.

Some segments of the Louisiana coast are eroding faster than others. Barrier island retreat and erosion are severe. From 1955/6 to 1978, the modern Mississippi River delta had one of the highest wetland loss rates in the state and parts of Barataria Bay and areas near Lake Pontchartrain were eroding as fast as in Plaquemines Parish (5% annually; Table 3-2). Although there was land gain in some areas, the overall loss rates are very significant in all of the hydrologic units within both the Deltaic and Chenier plains (Table 3-2). The highest rates of change were within the saline and brackish marshes in the Deltaic Plain and within the brackish and fresh marshes of the Chenier Plain.

Table 3-2. The average percent change from 1955/6 to 1978 in different hydrologic units along the Louisiana coast (adapted from Cowan et al., *in press*).

Hydrologic Unit	Percent Wetland Loss 1955/6-1978
Lake Pontchartrain/St. Bernard	15.4
Barataria	28.9
Terrebonne	28.9
Atchafalya	5.3
Chenier Plain of Louisiana	21.1

As the marsh turns into open water, the shoreline retreats inland. As the shoreline retreats, a positive feedback may develop. The amount of water moving into and out of the estuary increases with the enlargement of tidal inlets and bays following land loss; the higher flushing rate deepens the bay, leading to further erosion of the barrier islands. These barrier islands are retreating inland. The rate of retreat was 20% higher in 1954 to 1969 compared with 1932 to 1954 (Figure 3-5).

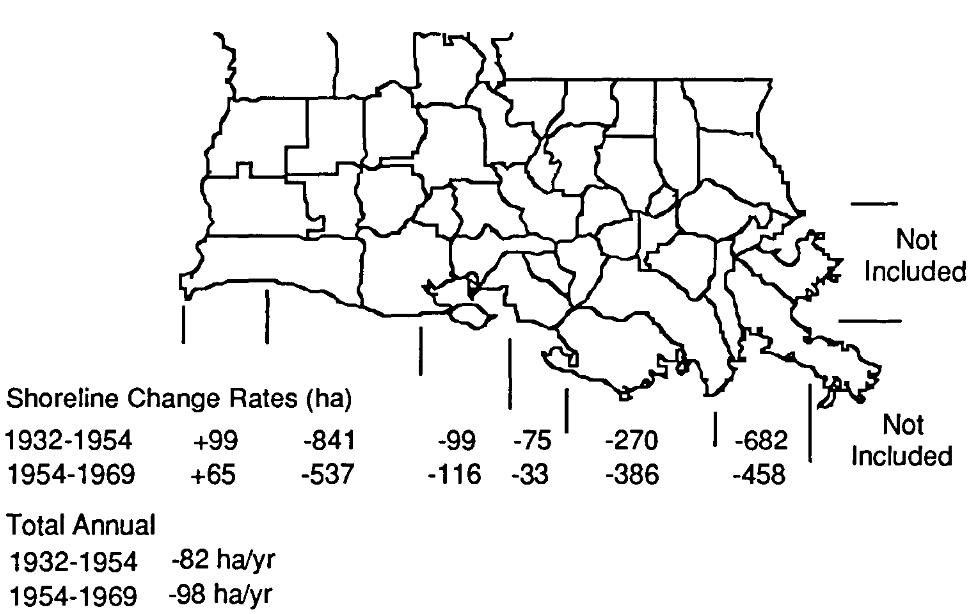

Figure 3-5. The land lost with the landward movement of the coastline from 1932 to 1969 (adapted from Morgan and Morgan, 1983).

In general, wetland loss rates are highest in young deposits near the coast and where recently accumulated sediments are thickest. The rates are lowest in older sediments far from the coast (Figure 3-6). Scaife et al. (1983) suggested several reasons for this. First, as a delta grows, overlaps, and extends seaward, the underlying deposits nearest the sea are necessarily the youngest, and the sediments within are sorted least and are less resistant to erosion. Second, the seaward edge of a delta is thicker, thus consolidation, dewatering, and downwarping are greatest there. Third, compared with landward, the seaward edge of a delta is more subject to wave attack, currents, and redistribution of sediments. Compared with younger deltas, older deltas have had more time to stabilize through consolidation, grain sorting or gravity.

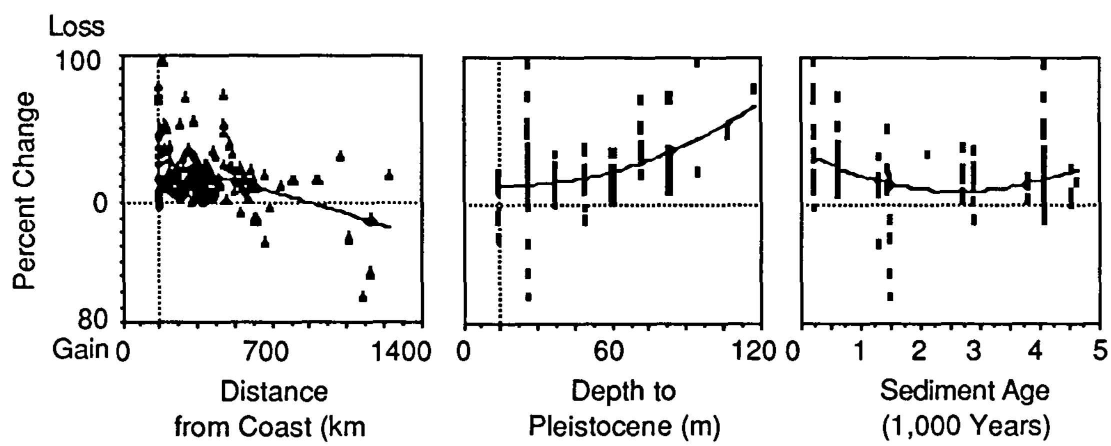

Figure 3-6. Wetland losses in 7.5 minute quadrangle maps from 1955 to 1978, as related to distance from the coast, depth to Pleistocene Terrace, and sediment age. Adapted from data originally prepared from Scaife et al. (1983) and additional changes resulting from this research. A polynomial regression is depicted.

Recent Changes in Natural Driving Forces

Major processes leading to the present high wetland loss rates include natural factors and those resulting from man's influences, including: (1) reduced sediment concentrations in the Mississippi River (Meade and Parker 1984), and heightened flood levees which prevent overbank flooding of the remaining sediments; (2) a complex hydrologic modification converting 7% of the wetlands into artificial channels and spoil banks, predominately oil and gas recovery canals and pipeline canals but also navigation waterways; (3) geologic subsidence; (4) sea level rises; (5) biological changes; (6) catastrophic events; and, (7) relative changes in sea level caused by absolute and relative sea level rise. "Are there recent changes in natural driving forces?" is a major question, which is addressed here.

Sea Level Rise

Determining changes in sea level in more recent times is difficult because of methodological and data limitations. Tide gage records are usually used to determine trends, and the tide gage itself is influenced by changes in substrate position, and seasonal variations. However, three stations around the Gulf of Mexico are assumed to be on relatively stable geologic platforms and, therefore, to represent eustatic sea level changes. From the 1920s to the 1980s gages at Pensacola, Cedar Key, and Key West, Florida had a mean annual water level rise of 0.22 cm/yr. Annual average water level at these three stations and globally (Barnett, 1984), and the rise is virtually identical (see Chapter 11). This indicates that water level changes at those Florida sites are steady. The best estimate of *regional* average eustatic sea level rise this century is therfore 0.22 cm/yr. Although clearly fluctuating over decades, the long-term rate of eustatic sea level in the Gulf of Mexico has not accelerated significantly since 1908 (Figure 3-7).

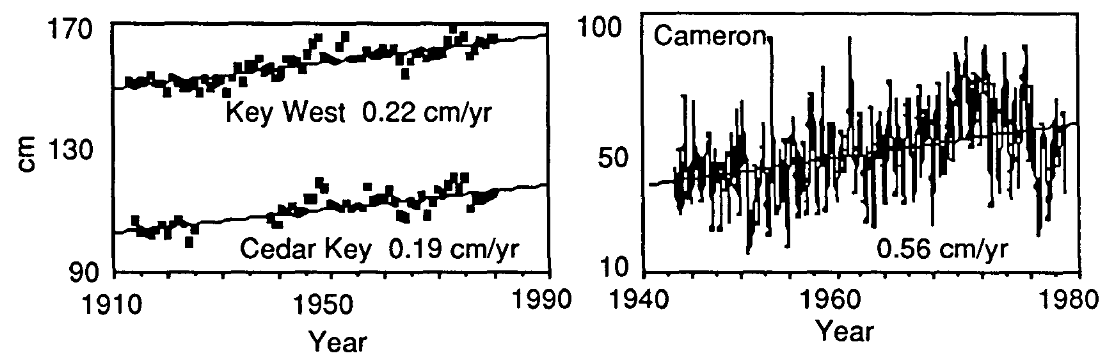

Figure 3-7. Water level changes at three coastal stations. Left panel: mean annual water level (cm) at Key West and Cedar Key, Florida. Right panel: mean monthly water levels at Cameron, Louisiana.

Although steady over decades, there are significant seasonal variations in eustatic sea level. Monthly mean sea levels for the Cameron, Louisiana tide gage stations are shown in Figure 3-7. The range of these monthly variations is much larger than the inter-year variations (up to 20 times higher), and there is generally a winter low and late summer peak. It is not obvious that the seasonal range has increased significantly in the last 50 years.

Global sea level rise may accelerate in the next century from the effects of increased atmospheric carbon dioxide concentrations, resultant temperature rise, glacial melting, and oceanic expansion. It has been suggested that a eustatic rise in sea level of 0.56 to 3.45 m is possible by the year 2100 (e.g., Hoffman et al., 1983).

Geological Subsidence

In addition to absolute changes in sea level, water levels change because of the compaction of sediments and downwarping of underlying depositions; the geosyncline depocenter has the highest rate of subsidence. Estimates of geological subsidence have been made by analyzing tide gage records. The most comprehensive and recent estimate is by Ramsey and Moslow (1987). In brief, they computed subsidence by subtracting the long-term eustatic sea level rise from the local water level record. Daily averages were computed to determine monthly and annual averages of the relative water level at that gage over a period of years. Ramsey and Moslow (1987) concluded from their analyses that geological subsidence increased during the period 1962 to 1982 compared with the period from 1942 to 1962. As discussed in Chapter 11, their analyses actually measured a relative increase in water level rather than subsidence. Only when the influence of seasonal changes in water level and eustatic variations are removed from the record can geological subsidence be estimated correctly. When these two corrections are made, it becomes clear that there is no measureable change in geologic subsidence evident in the tide gage records since the 1940's.

Although the rates of eustatic sea level rise and geologic subsidence have not accelerated significantly this century, the relative water level may have been rising. First, recent climatic fluctuations may result in longer and higher water levels during certain parts of the year. This aspect of water level rise is largely unexamined. Second, water management (e.g., for navigation purposes) may result in higher water levels if locks hold water in or pumping diverts water where it would not normally be.

Sediment Supply

According to Meade and Parker (1984), suspended sediment loadings abruptly declined in the mid-1950s following dam and reservoir construction on major tributaries of the Mississippi River (Figure 3-8). Suspended sediments fall out of suspension behind dams and in reservoirs, thereby filling the newly-formed basin and giving the basin a fixed, useful life for the purpose of the construction. The gradually more intensive land use throughout the Mississippi River watershed should have contributed to increased sediment loading but has apparently not yet filled the dams and reservoirs built there. This aspect will be discussed in Chapter 12. As a result, suspended sediment concentrations (but not necessarily bedload sediments) declined throughout the entire Mississippi River watershed in the 1950s, although the relative amounts are probably not well-represented by data from one station. The basins are apparently still filling since the suspended sediment load has remained at its present level since the period of major dam and reservoir construction.

The Old River Control Structure, upstream from New Orleans (Figure 3-9), diverts 30% of the Mississippi River volume into the Atchafalaya Basin and the Atchafalaya River which itself debouches south of Morgan City, Louisiana. Measurements of bed load show a shift to finer grains sizes at the bird-foot delta (Keown et al.,1981). In addition, the extensive levees along the Mississippi River delta (Figure 3-10) divert sediments downstream, close off crevasses, and prevent normal overbank flooding.

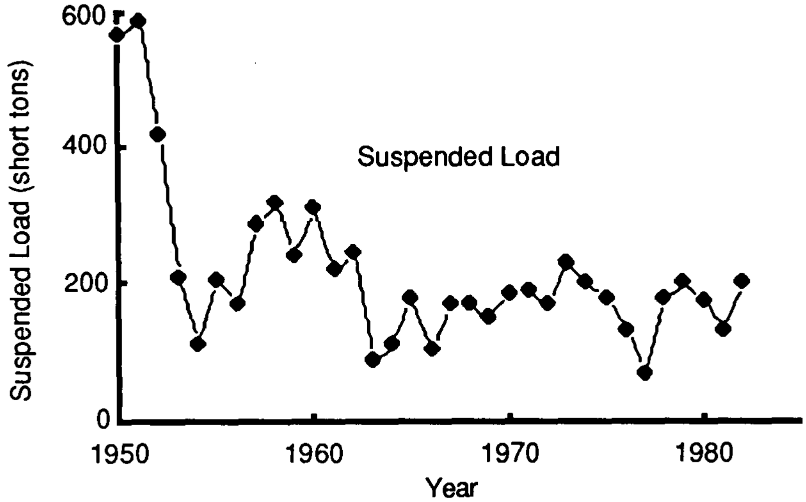

Figure 3-8. Annual discharge of suspended load in the Mississippi River at Baton Rouge, Louisiana, from 1950 to 1982 (adapted from Meade and Parker, 1985). The values were reported in short tons (1 short ton = 2,000 pounds, or 907 kilograms).

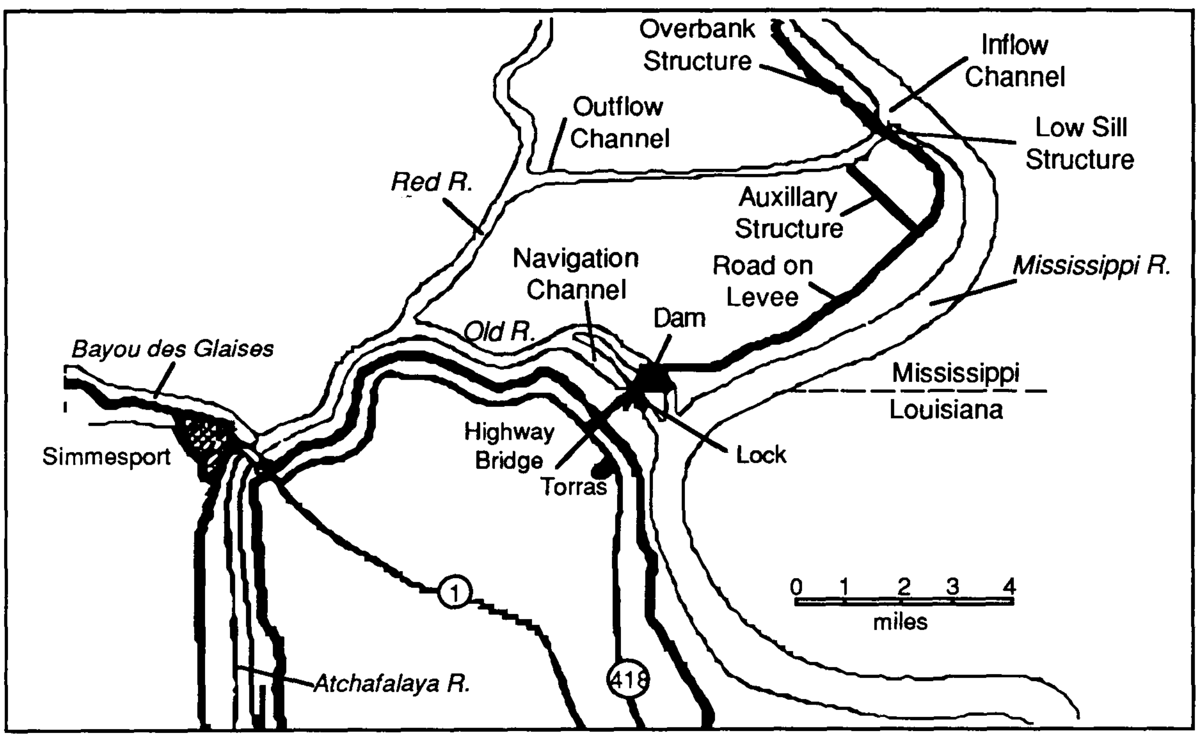

Figure 3-9. Old River Control Structure.

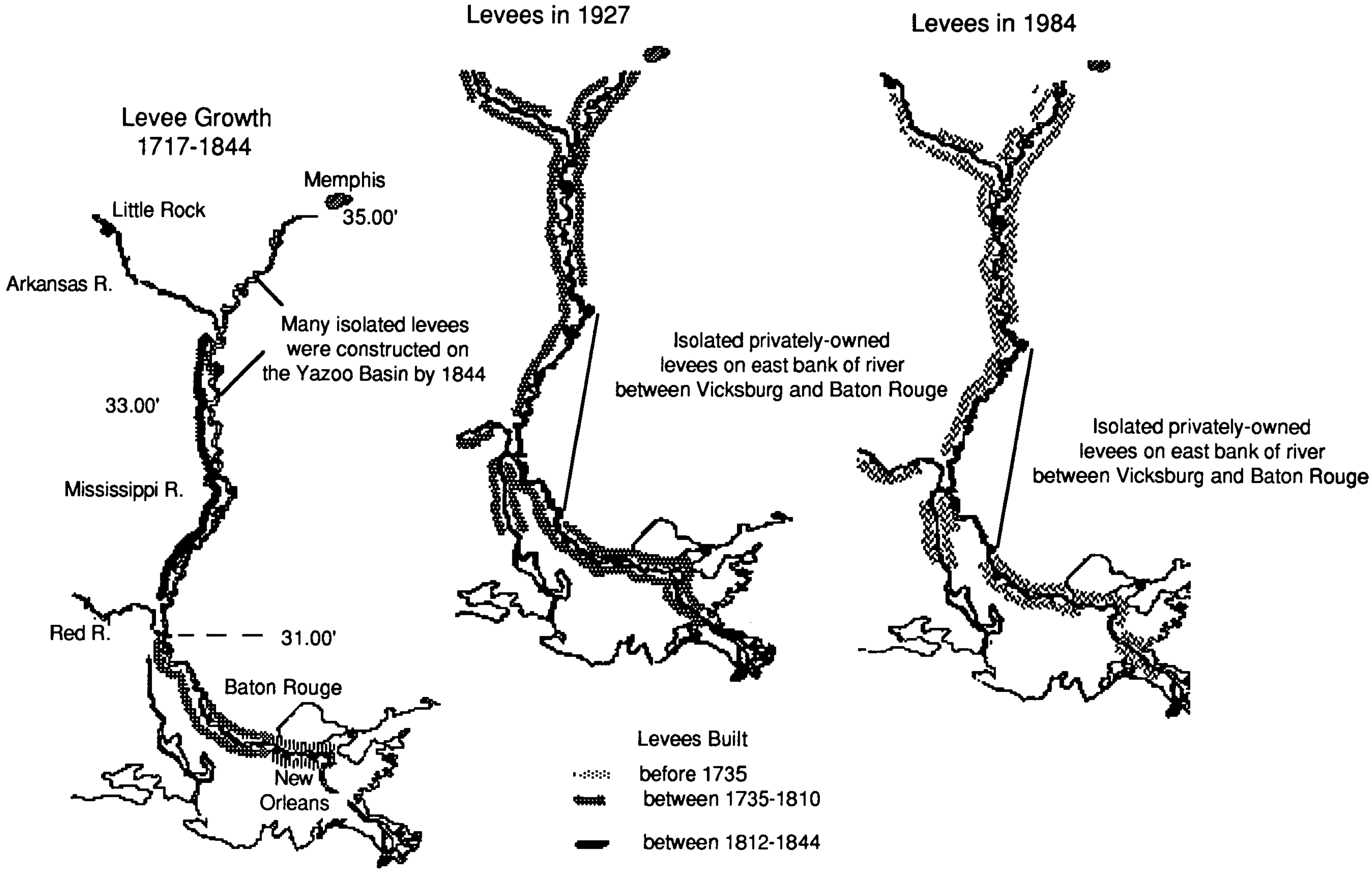

Figure 3-10. The development of the levee system on the southern Mississippi River from 1717 to 1984 (after Elliot, 1932; COE, 1984).

Man-made Influences on the Landscape

Early History

At the time of the Louisiana Purchase in 1803, the wetlands were already populated by hunters, trappers, and fishermen who occasionally modified wetland hydrology with navigation trails; a few farmers also tried to reclaim wetlands by using levees. With passage of the Swamp Lands Acts in 1859 and 1860, ownership passed from federal to state and then to private hands. More and larger wetland management followed, either directly or indirectly, either passive or actively. At least 43 large-scale agricultural impoundments were built by 1915; most of them failed (Turner and Neill 1984). Extensive cypress logging was conducted at the beginning of this century. Today one can still see the water trails formed when trees were dragged toward the dredged channels to be floated or loaded away to the mills.

Aquaculture ponds developed as early as the 1950s and are lcoated primarily in swamps and upland terraces. The Mississippi River was effectively walled in with levees from Vicksburg to the Gulf of Mexico after the unexpectedly high and disastrous floods of the late 1920s (Figure 3-10). However, the most intensive and extensive modification of wetlands results from the growth of canals dredged since the 1930s when offshore and onshore oil and gas recovery activities began in earnest.

Hydrology

Major hydrologic changes caused by man's activities result from navigation channels, canal and spoil bank construction, and the Mississippi River flood protection levees. The major navigation channels include the Intracoastal Waterway (running roughly east to west at the northern end of the study zone) and individual channels (running to south) connecting inland waters and the Gulf of Mexico (Figure 3-11). The amount of OCS transportation on these canals and their impacts are discussed in Chapter 4 and saltwater intrusion aspects in Chapters 6 through 8. An example of a primary hydrologic alteration of a deep navigation (shipping) channel is in Calcasieu Lake in the Calcasieu Basin during the mid-1940s. This shipping channel captured the main flow of fresh water from the Calcasieu River that, during periods of peak discharge, circulated throughout the basin (Alexander 1985). The deep channel now acts as a "salt pump" and brings saline waters further inland (Alexander 1985).

Canals in coastal Louisiana are built by various dredging methods to assist navigation, below-ground mineral recovery, pipeline construction, and trapping (for more complete discussion, see Allen and Hardy, 1980; Davis, 1973). Most canals are constructed to service the offshore and onshore oil and gas industry (Adams et al., 1978), which largely developed after 1940 (Figure 1-2). In some years, almost one-third of the U. S. Army Corps of Engineers (COE) dredge and fill permits are issued in Louisiana (Mager and Hardy, 1986). Each oil and gas field in the coastal wetlands has numerous canals and spoil banks. The canals are dug to transport drilling equipment, and the spoil banks are the residual dredging materials placed on either side of the canal, most often in a continuous line. Oil and gas annual production rates peaked about 10 years ago and have since declined, in spite of the deregulation of prices in the late 1970s. Consequently, fewer canals have been built in recent years, although the cumulative total canal area continues to climb. Presently, the surface area of canals is equivalent to 2.3% of the present wetland area. Every hydrologic unit has a significant area of canals that has increased greatly in the last 25 years. Overall, the total area of spoil bank levees plus canal surface is equivalent to about 7% of the present wetland area in the Mississippi River deltaic plain. The natural channel density in a natural marsh is about 8% to 10% of the marsh. The annual enlargement of some canals ranges from 2% to 14.8% or a doubling rate from about 5 to 35 years (Craig et al., 1980; Johnson and Gosselink, 1982). Enlargement of the existing canals now approaches the area of new canals added each year. The lifetime of spoil banks is measured in decades (Monte,1978).

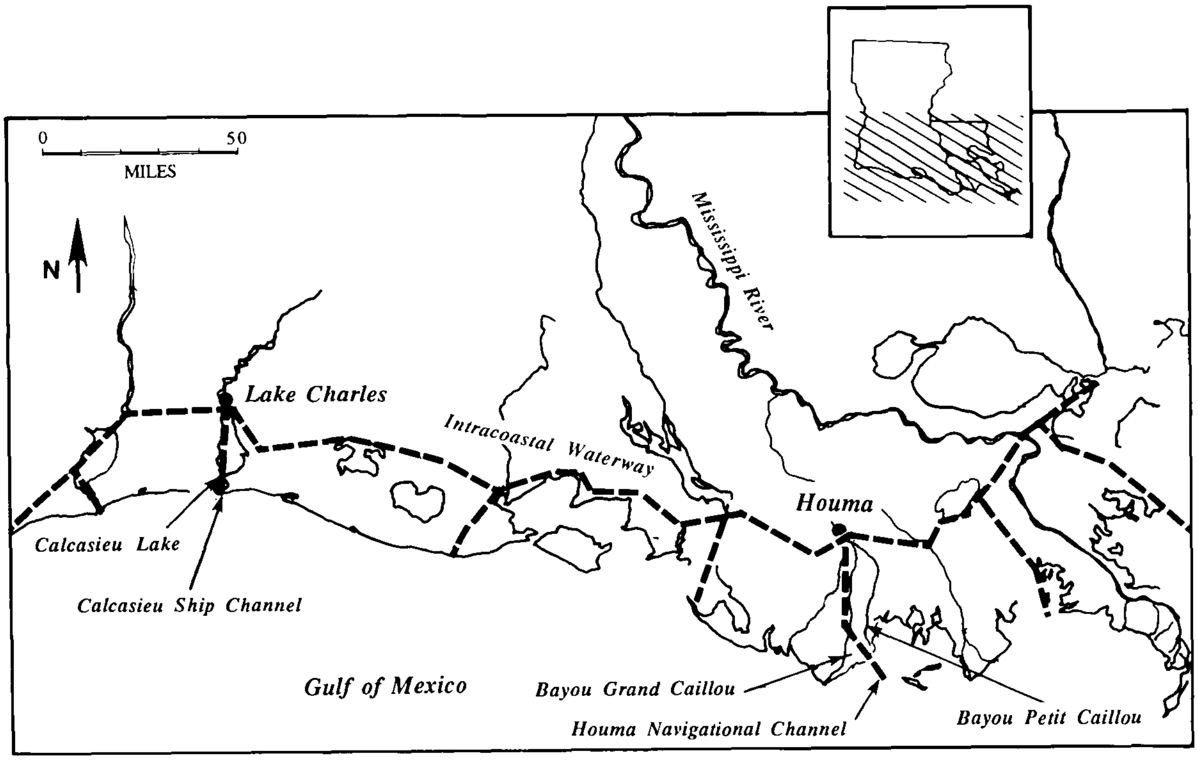

Figure 3-11. The major navigation channels in the study areas.

The weight of the levees themselves compacts the wetland soils beneath them. Nichols (1959) documented that below-ground water not only had a smaller cross-sectional area to pass through beneath a levee, but also a more impenetrable material. This finding indicates that the marsh is effectively hydrologically isolated from the nearby waterbodies from both above and below by a levee.

The indirect impacts of hydrologic modifications, however, have only recently been recognized and are, in general, poorly documented or understood (Allen and Hardy, 1980). One major indirect effect is to partially or completely impound marshes thereby reducing drying cycles and increasing flooding times (Table 3-3). Naturally formed streams often widen at the tips of the formerly smaller headwaters, whereas other streams disappear. Near Leeville, Louisiana, the area of natural drainage channels decreased exponentially with the linear increase in canals (Craig et al., 1980). As the canals and levees cross the channels, the latter often close off, silt in or erode into open ponds. Man-made features are spatially and temporally related to the "holes" in the marsh, wherein wetland emergent vegetation is converted to open water. Papers discussing wetland loss and causes, mitigation, and related management issues include Craig et al., (1979), Scaife et al. (1986), Sasser et al. (1986), Neill and Turner (1987) and Cowan et al. (in press).

Table 3-3. Changes in hydrologic regime of a semi-impounded salt marsh (from Swenson and Turner, 1987).

	Control	Semi-Impounded
Flooding		
number of events	12.9	4.5
event length (hours)	29.7	149.9
Drying		
number of events	11.6	4.0
event length (hours)	31.2	53.9
Mean Water Level (cm)	1.71	3.99
Volume Exchange (m3/m2 marsh surface)		
aboveground	0.15	0.06
belowground	0.09	0.04

Subsidence

Soil oxidation and subsurface fluid withdrawal may encourage subsidence in surface and deeper soils, respectively. Surface drainage has resulted in a lower water table and soil oxidation and has consequently lowered the land surface in New Orleans (Snowden et al., 1977; Traugher et al., 1979). Oil, gas, and production fluid withdrawal from deep layers has resulted in measurable increases in subsidence outside of Louisiana (Castle et al. 1969; Kesteren 1973). Local subsidence caused by oil and gas fluid withdrawal in Louisiana has been estimated to be 2 cm for the entire period of production (Martin and Serdengecti, 1984; Chapter 10).

Impoundments

Marsh management plans (MMPs) are an additional major change in the landscape. Almost all MMPs, either implemented or proposed, include structural marsh management practices that result in the impoundment or semi-impoundment of wetlands in Louisiana's coastal zone. Complete hydrologic isolation of a management area is an impoundment, while partial confinement is a semi-impoundment. Impoundments are hydrologically isolated, either naturally or artificially by a surrounding levee, and therefore they are disconnected from regional riverine or estuarine systems (Cowan et al., in press). The area of Louisiana impoundments, by category, is summarized in Table 3-4 and amounts to nearly 10% of Louisiana's coastal wetlands.

Table 3-4. The area, by category, of coastal Louisiana impoundments up to 1978 (adapted from Day et al., 1986; one ha = 2.47 ac).

Impoundments Category	Area (ha)
Agricultural	153,645
Crawfish	48,564
Fish and Wildlife	118,198
Urban	34,435
Unintentional - incomplete survey	46,289
Natural - incomplete survey	2,918
Unsuccessful[a]	34,435
TOTAL	438,484

[a] Original purpose of impounding not evident.

Fauna

The muskrat (*Ondatra zibethicus*) is a common herbivore in south Louisiana brackish marshes, and, when their populations are dense, they can decimate a marsh, particularly a three-cornered grass (*Scirpus olneyi)* marsh. The muskrat population, with reduced food, then either rapidly decreases or migrates. O'Neill (1949) noted that these "eat-out" areas normally revegetate after a few years. Gosselink (1984) mentions that the frequency of eat-outs is now much rarer than in the 1920s and '30s because of hunting pressure. Nutria (*Myocaster coypus*) were introduced to Louisiana at the turn of the century, and their population has expanded to the point where their fur harvest is a major economic activity in south Louisiana. Exclosure experiments at the Atchafalaya delta have demonstrated their dominant influence on vegetation colonialization in new deltas (J. G. Gosselink, Center for Wetland Resources, LSU, personal communication).

An introduced insect pest of the common freshwater plant alligator-weed *(Alternanthera philoxeroides*) invaded Louisiana wetlands between 1955 and 1978. The insect was introduced to control submerged aquatics in reservoirs and weakens stems causing reduced growth and physical collapse. Alligator-weed comprises 6.3% of all fresh marshes (Chabreck, 1972) and was particularly common in the Mississippi River delta. It is unknown just how severe the impact of the insect has been on wetland loss rates, but some alligator-weed is still present in the delta.

Wetland Plant to Open Water Conversions

The above discussion indicates some of the empirical and correlative relationships between changes in wetland hydrology and habitat changes. Our knowledge of why and how hydrologic changes affect habitat quantity and quality is incomplete, and it is the purpose of this study to enlarge the basis for understanding. Wetland plants are sensitive to alterations of the soil chemistry and hydrologic changes, and natural and man-made influences on soils and plants may significantly alter the balance between wetland gain or loss. The influences are discussed here.

Soil Properties

Soils respond rapidly to alterations in wetland hydrology because of the relatively quick growth of soil microorganisms and fast chemical reactions. The long-term development of soils can indicate the future health of plants, useage, and sediment accumulation. Marsh soils are often reducing environments where elements other than oxygen serve as electron acceptors. The proportion of oxidized-to-reduced components in the soil constitutes the redox potential (Eh) and can be measured in millivolts as an electromotive force about the electrode. Reduction occurs sequentially through a series of elements (oxygen, water, nitrate, manganese +4, iron +3, sulphate) from higher to lower Eh values. In general, reduction of one element begins only after reduction of the others is complete. Oxidized elements are reduced biologically by bacteria which have a definite selectivity for element type and Eh-pH range. For example, a specific bacteria group reduces sulphate to sulfide at -200 to -300 mv, only when nearly all the iron is reduced. Redox measurements thus give an indication of what elements are being reduced and the metabolically active organisms that are present. Decreased soil water movement and stagnation has been correlated with lower redox potentials and high concentrations of reduced compounds (particularly sulfide) in salt marsh soils (Mendelssohn and Seneca, 1980; Howes et al., 1981; King et al., 1982).

Plants, Sedimentation Rates, and Submergence

> Vegetation not only behaves as a geomorphic process through its stabilizing and binding effects on detrital sediments and by offering a continuous supply of organic material but also serves to disclose the roles of other processes such as tide and salinity regimes. (Coleman and Wright, 1971:64).

Plants contribute to vertical soil accretion by (1) trapping sediments; (2) accumulating locally and distantly produced organic matter; and, (3) depositing root material below-ground. Their distribution reflects influences of salinity, below- and above-ground hydrology, and interaction with insects and herbivores.

Vertical soil accumulation is therefore not simply the result of sediment supply but also of the interaction of plants and the prevailing hydrologic regime. For example, besides trapping mineral matter at the surface, plants add a substantial amount of organic material to the soil belowground in the rooting zone. Fresh marsh soils are mostly organic debris produced *in situ*. Even salt marsh soils may be composed of up to 50% organic matter. Second, as the organic material is laid down, mostly belowground, the weight per unit volume (measured as bulk density) decreases. This has been proposed as a major contributor towards sedimentation rates in mangrove coasts (Wells, 1981). Because of organic matter, marshes need less mineral matter than a bay bottom to maintain elevation in the face of a rising sea level or a sinking substrate.

Wetland plants are sensitive to soil chemistry, and they respond both metabolically and morphologically to altered hydrologic regimes (Linthurst, 1979; Mendelssohn and McKee, 1981; Mendelssohn et al., 1981). Because the reciprocal feedback loops are balanced in a stable marsh, the disturbance of any one of many factors may result in marsh loss or gain. Thus the soil pH-redox equilibrium, soil aeration, and plant water requirements are interrelated, but these relationships are not completely understood, even in laboratory settings (Sasser, 1977; Mendelssohn, 1979; Jakobosen et al., 1981; Mendelssohn et al., 1981).

The effect of excessive soil waterlogging on inland forms of "short" *Spartina alterniflora* is decreased productivity: the factors inhibiting plant growth are directly related to increased anerobic soil conditions and related factors leading to low Eh (Mendelssohn et al. 1981, 1982). Low soil Eh may result indirectly or directly in nitrogen deficiencies. Related factors include root oxygen deficiencies, toxins produced by soil anerobic respiration, reduced water movement causing nutrient depletion near the root, decreased root metabolism, and a smaller oxidized rhizosphere that buffers the plant against soil toxins. In general, salt marsh plants are very sensitive to change in the hydrologic regime (Table 3-5).

Table 3-5. Examples of the responses of coastal salt marsh communities to altered hydrologic regimes.

Experiment	Purpose	Result	Source
1. Streamside plants moved to lower elevation in the marsh (in pots; LA)	increase flood height and duration	lower standing crop, plant height and density	Mendelssohn et al., 1982
2. Inland plants moved to higher elevation in the marsh (LA)	decrease flood height and duration	higher plant biomass, height and density	Mendelssohn et al., 1982
3. Drainage tiles added to inland marsh (GA)	increase belowground water movement iron	higher plant biomass, lower sulfide, higher	King et al., 1982; Wiegert et al., 1983
4. Belowground water movement blocked (LA)	reduced belowground horizontal flow, increased soil flooding	reduced Eh and flower density	Turner et al., unpublished
5. Aboveground water movement blocked (LA)	very reduced sheet flow and increased flooding	variable Eh change; negligible sedimentation	Turner et al., unpublished
6. Above- and below-ground water movement blocked (LA)	very reduced waterflow and increased below-ground flooding	reduced Eh, sedimentation rate and tasseling	Turner et al., unpublished.
7. Above- and below-ground water movement blocked (NC)	very reduced sheetflow and increased below-ground flooding	reduced Eh, plant growth and sedimentation rate	Mendelssohn and Seneca, 1980
8. Impoundment levee broken (FLA)	increased above- and belowground waterflow	dying vegetation recovered	Gilmore, et al., 1981
9. Ditched marshes filled in (DEL)	re-establish original hydrology reversed	vegetation recovered soil subsidence	Stearns et al., 1940

The reduction status of wetland sediments influences the growth of plants, in part, through the plant rhizosphere by affecting the biological availability of the more important plant nutrients, such as nitrogen and phosphorous. Phosphorus is generally more available under flooded conditions (Patrick and Delaune 1977). Waterlogged soils high in organic matter may have a low Eh, resulting in the reduction of sulfate to sulfide (Gambrell and Patrick 1978). Sulfide may be toxic to the biota. Sulfide can leave the system as hydrogen sulfide gas or can precipitate with ferrous iron to form ferrous sulfide which is not toxic (Patrick and Delaune 1977). Goodman and colleagues (Goodman et al. 1959; Goodman 1960; Goodman and Williams 1961) have proposed that the increased soil sulfide concentrations occurring in standing waters may be implicated in the physiological demise of some marsh plants. A study in a Georgia salt marsh showed that where water movement was experimentally increased, plant productivity doubled (Wiegert et al. 1983);

where water movement and plant productivity were least, sulfide concentrations were the highest. Drainage of sulfides out of the area and inputs of iron into the area (from tidal flushing) may help keep the toxic effects of sulfide from lowering marsh productivity (King et al. 1982).

Salinity

Soil salinity has an immediate effect on marsh health through its effect on plant growth. There are many places across the coast where one finds dead cypress trees. This situation is commonly attributed to saltwater intrusion, but may also be the result of increased waterlogging. The potential effects of salt water were thought to be so significant to our understanding of the marsh that four separate studies on salinity were conducted as part of this project (Chapters 5 through 8).

Mosquito Ditches as Analogues: Three Examples

Although there are no published long-term, field-oriented studies in Louisiana of the management issues surrounding future impacts and alternatives, older studies of analogues of canal and canal spoil bank levees exist, which suggest that hydrologic changes have contributed to the present high landloss rates. Beginning at the turn of this century, mosquito ditches were constructed to reduce mosquito larvae habitat. Millions of coastal marsh acres were ditched to increase drainage. The resulting tensions arising amidst agricultural, health, wildlife, and conservation groups, among others, led to several studies of interest here. Mosquito ditches and their levees, as constructed then, were in similar proportion, but not scale, to most canals and spoil bank levees now being constructed in southern Louisiana. The ditches were often 0.6 m (1.8 ft) wide and deep. They criss-crossed the marsh and emptied into a larger water body. Canals, although often 40 m (130 ft) wide and 5 m (16 ft) deep and much longer, also have the dredged spoil material piled high on either side of the canals. Mosquito ditches and their levees are analogues of canals; a review of their impacts is useful to understand the influence of canals and canal spoil banks on wetland loss rates in Louisiana. Three major studies are reviewed here in the context of this study.

Mispillion River (Delaware)

Bourn and Cottam (1950) conducted a classic study of mosquito-ditching impacts on coastal marshes near the Mispillion and Herring rivers in Delaware. Vegetation changes in the Mispillion River from 1936 to 1946 were documented following ditching operations. Lesser et al. (1976) later re-examined the same area and attributed the vegetation changes observed during Bourn and Cottam's study to deepening of a navigation channel, not mosquito ditches. Daiber (1982) refuted the basis of the Lesser study's alternative explanation as unsupported by the available facts.

Figure 3-12 is a reconstructed map of the vegetation cover from before the growing season and ditching (1936) and after ditching for the next several years. The vegetation was initially 90% *Spartina alterniflora*, the same species occurring in the saline marshes of Louisiana. Following ditching, dead and dying plants appeared, particularly at lower marsh elevations in soft mud. By the end of the first growing season, shrubs re-colonized these sites. In later years shrubs invaded the areas between the ditches. Some ditches became clogged and supported the re-invasion of *S. alterniflora*. Within 10 years,

> ...the whole floral picture of the Mispillion marsh had changed....from that of a decade before, when *Spartina alterniflora* in pure stand covered 90 percent of the area. In 1946 the shrub *Baccharis halimifolia* was the

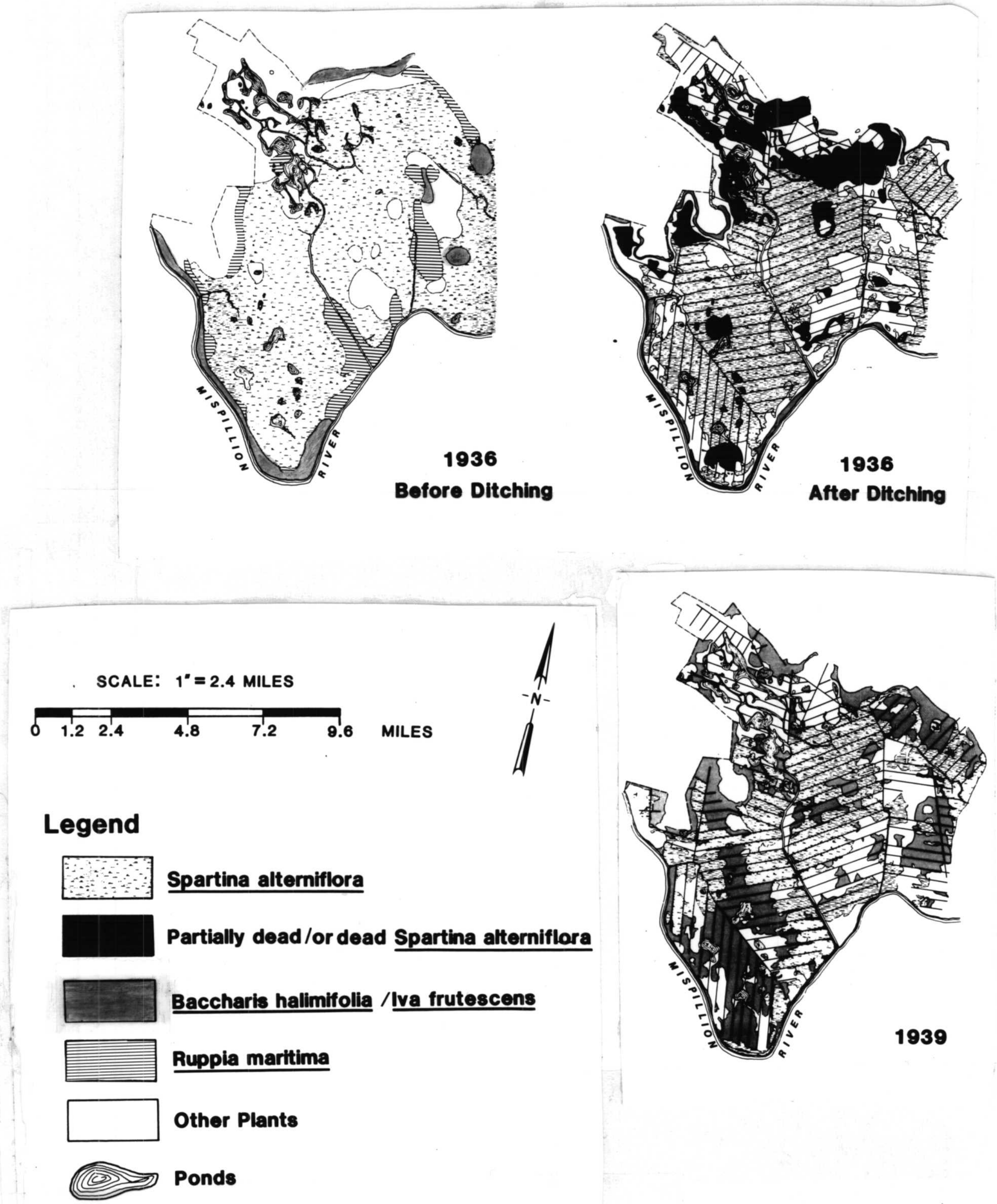

Figure 3-12. The vegetation changes in the Mississippi and Herring Rivers marshes following mosquito ditching (from Bourn and Cottam, 1950).

> dominant plant, with *Spartina alterniflora* limited to low areas near the center of the marsh. Even there, *Iva* and *Baccharis* had spread along the ditches, and in time, with the deposition of silt and the accumulation of organic matter, these species might be expected to take over the intervening spaces between the ditches. The spread of *Baccharis* was accelerated in the later years of the study by the erosion of the ditches near their points of discharge into the river.... Originally 20 inches wide and 20 inches deep, some of these ditches eroded until they were several feet in both width and depth near their mouths. (Bourn and Cottam 1950:6).

Submerged macrophytes, such as *Ruppia maritima*, were also replaced following drainage. In addition, Bourn and Cottam documented the marked reduction in invertebrate populations after ditching.

Two important processes occurred. First, some vegetation died following the manipulation of the hydrologic regime; other plants then recolonized the area. The plant response occurred within 1 year; it persisted for at least 12 years before the ditches began to fill, and the vegetation patterns clearly followed changes in the hydrologic patterns. Similar vegetation changes can be detected in aerial photographs of Louisiana marshes. Second, reinvasion by *S. alterniflora* occurred after the ditches were filled The importance of these observations becomes especially significant when one realizes that much of the vertical accretion in Louisiana salt marshes, and most of the accretion in fresh marshes, is dependent on the below-ground accumulation of organic matter produced by the plants. Therefore hydrologic changes affecting plant vigor may have a direct influence on sediment accumulation. The rate of subsidence in Mispillion marshes is much lower than the rate of approximately 0.7 to 0.9 cm/yr occurring in southern Louisiana. The loss of below-ground organic production for only a few years may mean the permanent conversion of wetland to open water because the vertical accretion deficit accompanying artificial drainage is greater than that to which the system can adjust.

Cleaver Marsh (Delaware)

Daigh, Stearns, and colleagues (Daigh et al., 1938; Stearns et al., 1940) conducted a careful study of the impacts of mosquito ditching on a Delaware marsh during the late 1930s and irregularly for several years thereafter. Their main interest concerned the muskrat population, but they also collected much information on vegetation and waterlevels. The marshes they studied are botanically similar to those found in Louisiana.

One of the principle impacts was the lowering of the water table and the ground level following ditching (Figure 3-13). Furthermore, when the ditches were deliberately filled with sediments after two years, both the water level and the ground level rebounded. They also documented an invasion of shrubs into the previously fresh marsh, a change in soil pH, and a negative impact on the muskrat population. The muskrat, reacting to both the lowering of the water table and the ground surface and to the vegetation changes, virtually abandoned the ditched area within one year. However, muskrat returned the year after the ditches were deliberately filled back in.

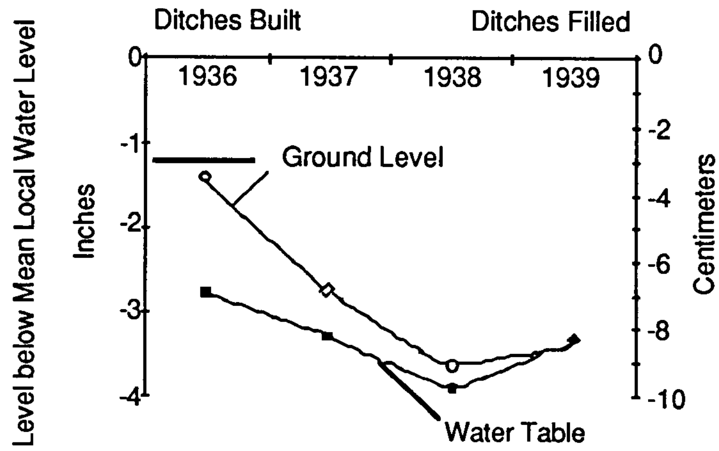

Figure 3-13. Water and ground level in the Cleaver Marsh before and after ditching (adapted from Daigh et al., 1938).

Two relevant points emerge from the Daigh et al. (1938) study. First, fresh marsh vegetation was affected by artificial drainage features with similar potential impacts as might occur in salt marshes. Second, the water table changes, which affected plant and animal distributions and habitat value, resulted in increased subsidence. The water table rose after the artificial drainage was reversed. Whether the same amount of change would occur in Louisiana following similar treatment is now unknown. Wetland hydrology, and hence soil accumulation, however, was clearly influenced by the ditches.

Wequetequock-Pawcatuck Tidal Marshes (Connecticut)

Miller and Egler's (1950) study of a ditched New England salt marsh is enlightening for the obvious parallels with Louisiana marshes. The sod line formed from the ditching operations is somewhat analogous to the larger canal spoil bank. Miller and Egler's description of ditch enlargement also corresponds to the erosion of canals observed in southern Louisiana, except for the scale and geographic location. Of particular interest is the way that the ditch-levees formed from the ditch turf line to convert the entire inter-ditch marsh into a panne:

> Such a panne then tends to hold the water and to produce the very kind of pool which the ditch was originally designed to drain. These new pannes are rectangular in shape, and alternate regularly with the ditches and their levees. (Miller and Egler 1950:168)

Miller and Egler (1950) also noticed that the ditches tended to enlarge and that vegetation patterns were altered, particularly near the levees.

One consequence of the impoundment was that the turf line trapped salt water washed onto the marsh during high tides. Following evaporation, additional pannes developed when the plants could not withstand longer periods of exposure to saltier conditions. Long ago ditches were dug to bring salt water further into the marsh than would normally occur and turned high marshes on the East Coast into coastal hay fields. Eliot described his successful approach as follows:

> "Last fall I began upon it and drew (dug) a Ditch of four FooT wide from a large Salt Creek, and carried it up in the middle of the Cove seventy Rods,

> in order to turn it into Salt Meadow, that being the best that I could do with it: It so far answers the design, that the Tide flows regularly into it, to the upper end of it; the Tide now flowing, where I suppose it never reached before." (J. Eliot 1748, as quoted by Nixon, 1982, p. 50).

In addition to results paralleling the other studies, one unique and relevant point emerges from Miller and Egler's study: panne formation, which is similar to the features that occur alongside Louisiana canal spoil bank levees, may sometimes be caused by entrapment of storm waters behind the levee. Plants that might otherwise adapt to short periods of higher-than-normal saltwater content die during long exposure to salty water. Saltwater entrapment (prolonged exposure to salt water) may be just as significant as saltwater intrusion (higher than normal salinities) into mostly freshwater marshes in determining vegetation community characteristics.

Marsh Plasticity

Wetlands are not all equal in composition or rates of areal gain or loss. They can make some adjustments to changes. For example, marshes can stretch vertically with tides and trap various amounts of sediments in apparent adjustment to constant sea level rise. Harrison (1975) measured 5 to 8 mm vertical displacement of marsh soils over approximately 2 hours as the tide level rose in a Connecticut salt marsh. Dead-end canals tend to fill over time, and backfilled canals may revegetate as do spoil banks when torn down (Neill and Turner, in press).

Examples of successful restoration from other areas indicate the importance of hydrologic changes on wetlands and how much they can recover. For example, when levees forming an impoundment in Florida were broken, the vegetation returned (Gilmore et al., 1981). When the levees were initially constructed, the resulting blockage of natural hydrologic flows resulted in dead and dying vegetation. When the hydrologic flows were restored, the vegetation recovered. Marsh vegetation also reappeared when dikes were breached in two 200-acre marshlands near San Francisco (Faber, 1982; Josselyn and Perez, 1982). Subsidence, as well as vegetation and groundwater level, was reversed when the ditches were refilled from the existing turf line created from the ditching.

Summary

The present Louisiana landscape, formed from the interaction of geologic, biologic and climatic influences over the past 5,000 years is extensive, nationally significant, heavily used by man, and changing at historically high rates. From a qualitative point of view, the probable causes of the present high wetland-to-open water conversion include both natural and man-made influences within and away from the study area. These causes are probably interactive and may be expressed through plant death or removal, substrate erosion, reduced vertical growth of a sinking soil substrate or all of the above.

On a coastwide basis, wetland loss rates are accelerating in recent times, whereas the rate of geologic subsidence and eustatic sea level rise appear stable. Important elements of climate and the rate of sediment supply have changed, and hydrologic modifications of the coastal zone are now extensive. It is generally accepted that increased sediment sources, maintenance of the natural hydrologic regime and sediment distribution, and lower rates of sea level rise contribute to wetland growth; we may therefore assume that reversing those patterns will generally result in less growth or even wetland losses. Salinity changes, regardless of origin, may or may not result in wetland losses. Less well-known effects include the long-term impacts of hurricanes, animal introductions, and fluid withdrawals.

Chapter 4

DIRECT IMPACTS OF OCS ACTIVITIES

by

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The goal of this task was to quantify the direct impacts of OCS activities within the study area and to determine which factors contribute to the variability of direct impacts within the region. We define direct impacts as those man-induced activities directly linked to the physical conversion of one habitat type to another. The primary human activities involved are dredge and fill, and the major habitat changes are emergent wetlands to open water, or spoil. The major onshore dredge and fill activities associated with OCS development are pipeline and support facility construction and the construction or enlargement of navigation channels. Tasks involved within this effort included the identification of OCS pipelines and facilities, the measurement and inventory of direct impacts, the determination of those factors that account for the variability in the degree of direct impacts of OCS activities, and the allocation of all direct impacts to OCS versus non-OCS activities.

This study was designed to assess several questions:

(1) Is total direct impact an important factor in accounting for wetland loss (habitat change)?
(2) Does direct conversion of wetland/habitat to open water and spoil by OCS-related activities account for a substantial part of all direct impact wetland loss?
(3) Is the degree of direct impact by pipeline construction on wetland loss directly related to the construction technique employed, pipeline diameter, age of pipeline, geologic region, and habitat type?
(4) Do direct impacts resulting from the construction of navigation channels substantially contribute to total direct wetland loss?
(5) Does OCS water-borne traffic comprise a significant portion of the total water-borne traffic in the major man-made navigation channels?
(6) Is the degree of initial direct impact of navigation channels directly related to the construction dimensions?

Our efforts also included investigations into several aspects of canal and navigation channel widening caused by erosion. Specifically, we wanted to determine whether OCS-canal widening was a significant factor in accounting for total wetland loss and whether the rate of canal/channel widening was related to the amount and type of water-borne traffic.

Background

Channelization of the central Gulf wetlands is an important factor when considering the acute wetland loss problem affecting this region. Estimates of the proportion of wetland loss attributed to channelization and canalization by both direct and indirect impacts range from 20% (Johnson and Gosselink, 1982) to 50% (Boesch et al., 1983) to as much as 90% in selected regions (Turner et al., 1982). The types of impacts that result include: (1) the direct conversion of wetland to open water and spoil and the indirect loss of wetlands from enhanced saltwater intrusion; (2) canal widening from boat-wake erosion; (3) pipeline breakages and leakages; (4) alteration in the sedimentary and hydrologic regimes by spoil banks and channels; and, (5) differential loading and compaction of sediments that result from the weight of spoil deposits.

Man-made channels in the region are constructed for varying purposes, not all of which are linked to oil and gas development. Drainage canals, trappers' canals (trainasse), navigation channels, exploration access canals, and pipelines are widely dispersed throughout the region and all have contributed to wetland loss. As related to OCS activities, navigation channels and pipelines are a primary concern.

Limited field-based investigations on the impacts of pipelines have shown that the quantity and quality of fish and wildlife habitat in the wetlands are adversely affected (Tabberer et al., 1985). Also, canal widening rates have been documented and attributed to boat traffic and edaphic factors (Johnson and Gosselink, 1982).

Total canal area is estimated to be 10% of the Louisiana coastal region in 1978 and directly accounts for approximately 6.3% of the total wetland loss from circa 1955 to 1978. However, a strong statistical relationship between canal density and total wetland loss indicates that the indirect impacts of canals account for a substantially larger percentage of total wetland loss (Chapters 19, 20, and 21). A comprehensive examination of canals by function (e.g., OCS versus non-OCS) and an accounting of the variable nature of impacts (e.g., pipelines) among similar canal types of the region has not been previously conducted. In fact, in the case of the latter, there is some lack of recognition that variability in impacts among pipeline canals is significant. For example, the Minerals Management Service (MMS) in its guidelines on pipeline impacts on coastal habitats in the Gulf of Mexico region states "Regardless of the emplacement technique employed, an approximately 61 m (200-ft) wide strip of marsh vegetation will be destroyed. This amounts to about 25 acres per mile" (MMS, 1983:121). In the same publication, however, MMS takes a more enlightened view and states "...the contribution of a given pipeline (to wetland loss) depends upon many factors, including type of wetlands, size of pipe, number and extent of open water areas crossed, method of emplacement, and method, if any, of attempt to restore the area" (p. 136). Data collection and analyses in this effort are principally devoted to supporting, refuting, and modifying these statements.

Materials and Methods

Total direct impacts resulting in wetland loss from 1955 to 1978 were estimated using data provided by Wicker et al. (1980, 1981), after corrections for mislabelling of a small percentage of the habitat labels in a large and detailed data set that was otherwise carefully prepared. These data are for 1:24,000 scale 7.5 minute quadrangle maps and describe wetland habitats using the Cowardin et al. (1979) classification system. Additional habitats include upland, agricultural, and urban zones. Habitat classifications were reduced to nine categories. Direct impacts were assumed to be caused by agricultural and urban expansion and by canal and spoil bank construction. OCS pipeline and support facilities could not be separated from all other direct impacts because of the way habitat categories were defined during the mapping effort. Calculations of site-specific conversions could not be made; only the net change in individual quadrangle maps were calculated. Only 10% or less of the total map area is upland. Visual inspection of hundreds of maps indicated that virtually all urban and agricultural expansion and canal and spoil bank construction are in wetlands. Therefore, the net change in these four development categories was assumed to be the result of direct impacts and at the areal expense of wetland habitat.

The identification of OCS pipelines and related facilities required review of many information sources, mostly maps and data bases. No single information source was entirely accurate or complete. However, most of the missing data were located and conflicts resolved by comparing the total information pool and verifying some of the original data sources. The key data sets and maps used for OCS identification included: (1) the MMS data base on OCS pipelines; (2) an unpublished map provided by John Chance and Associates of Lafayette, Louisiana, depicting the offshore location of pipelines, operator, and size of line; (3) a series of historical maps of offshore and onshore pipeline system development, including operator, product transported, and pipeline diameter as published by the Louisiana Geological Survey (LGS); (4) a copy of the most recent data base used by LGS to publish maps (includes some operator verification); (5) file data from the Louisiana Department of Natural Resources and the Texas General Land Office, which provided information on processing facilities located within the study area as well as the percent OCS versus non-OCS product transported by individual pipelines; (6) maps published individually by various operating companies depicting their own pipeline systems, and in some cases, the systems of other operators; and, (7) unpublished proprietary data from several Louisiana researchers that provided detailed impact assessment of several OCS pipelines.

Once identified, pipeline locations were transferred to the latest edition of USGS 1:24,000 quadrangle maps. High altitude, high resolution color infrared photography (NASA Missions 86-032 and 86-033, Dec. 6, 14, 1985) was used to determine the exact locations of pipeline routes. Next, 1:24,000-scale mylar habitat maps (Wicker et al., 1980, 1981) were overlaid on the pipeline route maps. This information base was then used to assess the direct habitat changes resulting from pipeline construction and processing facilities. The habitat maps covered about 90% of the pipelines sampled. For the other 10%, we used the 1:24,000-scale habitat maps that were developed to support the work of Gosselink et al. (1979) and the coastal wetland map of Chabreck and Linscombe (1978). We measured 157 of the known 225 OCS pipelines.

Lengths of pipeline impacts were planimetered by pipeline and habitat affected. The habitats included beach (including dunes), open water, salt marsh, brackish marsh, intermediate marsh, fresh marsh, forested wetland, spoil, and upland (including natural levees, cheniers, and Pleistocene outliers). The area of impacts, in most cases, could not be similarly measured because the error factor for measuring widths of most spoil deposits and pipeline canals at 1:24,000 scale is far too large. The widths of 72 pipelines were field-measured using a Leitz automatic level and metric stadia rod. Average widths of the measured pipeline canals and spoil banks were divided into six classes and assigned a width class, based on air photo comparisons of their apparent width in relation to canals of known width. Exceptions to this rule were made for most of the very large pipeline canals (usually a corridor containing multiple lines), and the widths of these canals were based on actual field measurements. Impact areas were then calculated using the planimetered lengths multiplied by the width class assigned. Impact areas and length were recorded by pipeline, habitat, age, parish, geologic region, and pipeline diameter.

Impact length and area for navigation channels and associated spoil deposits were directly planimetered from the 1:24,000 topographic maps. Field measurements verified that these impacts were sufficiently large enough to be measured at 1:24,000 scale with an error factor of approximately 6%. It should be noted that canal width measurements were generally larger than canal widths reported by the U.S. Army Corps of Engineers (COE).

General linear models were developed to determine the significance of the factors measured in accounting for direct impact variability. Pipelines were treated separately from navigation channels. Analysis of variance was conducted for all categorical factors (e.g., geologic region, habitat type, construction type) including one-, two- and three-way models. Analysis of covariance was performed for the continuous factors (e.g., age, width, and diameter) and included all possible interactions of the factors.

All the major navigation channels support OCS and non-OCS activities and at least 44 out of 225 pipelines transport both OCS and non-OCS hydrocarbons. Allocations of impacts for those multiple use channels and pipelines needed to be determined. For pipelines, the percentage of OCS product flow to total product flow is reported by operators on a monthly basis. To our knowledge, the data are only available in raw form and are not centralized in any single, easily accessible form. Data for January, 1978, are available in summary form within a proprietary report prepared for the State of Louisiana by the Gulf South Research Institute for use in a tax/fee issue. From these data, we allocated OCS impacts, based on the percent OCS product being transported for January 1978. For pipelines constructed post-1978, the same method of allocation was used, based on the Operator Production Audit reports filed for the early months of 1984.

Table 4-1. Percent use of coastal waterways for OCS activities: alternative high/low estimates, based on data from Waterborne Commerce Statistical Center (WCSC) (Appendix B).

Name of Waterway (In order of WCSC-based % of OCS per total)	WCSC-Based Data	Alternative Estimates High	Alternative Estimates Low
Bayou Terrebonne & GIWW 14-59	46.7	60.2	7.9
Bayou Lafourche, LA	21.3	36.2	3.6
Empire, LA Waterway to Gulf	21.2	36.1	3.6
LaLoutre/St. Malo/Yscloskey, LA	19.3	33.6	3.3
Vermilion Bay & GIWW 159-160	10.3	58.8	1.8
Houma/LeCarpe/Gr. Caillou/Petite Caillou/60-78	9.4	26.3	1.6
Bayou Dupre, LA	9.1	18.2	1.5
Freshwater Bayou & 161-193	8.0	16.2	1.4
Atchafalaya River & GIWW 79-95	7.7	28.1	1.3
Mermentau River, LA	6.1	12.7	1.0
Mississippi River & Passes	3.3	7.2	0.6
Bayou Tech, LA	2.1	4.7	0.4
Gulf via Bayou Barataria Bay	1.9	4.3	0.3
Bayou Casotte, MS	1.8	4.1	0.3
Innerharbor Navigation Canal	1.3	3.0	0.2
Lake Pontchartrain, LA	1.2	2.8	0.2
Calcasieu River	1.1	5.2	0.2
Mississippi River Gulf Outlet	0.9	1.9	0.1
Sabine Pass Harbor, TX	0.7	1.5	0.1
Beaumont, TX	0.3	0.7	0.1
Petit Anse/Tigre/Carlin bayous	0.1	0.2	0.0
Merm/Nezpique/Des Cannes	0.1	0.1	0.0
Biloxi & Gulfport, MS	0.1	0.1	0.0
Pascagoula Harbor, MS	0.0	0.0	0.0

Allocation of direct impacts of navigation channels was based on vessel count, size, and destination data provided by the Waterborne Commerce Statistical Center (WCSC) and the Performance Monitoring System (traffic at navigation locks), both of which are part of the COE. The data are problematic because a substantial amount of vessel traffic goes unreported.

To account for the missing data, we used three separate methods (details are provided as Appendix A) that yielded three traffic count estimates and three estimates of percent OCS destination for each navigation channel (Table 4-1). These percentages were used as multipliers to determine the OCS allocation of the total direct impacts. Many of the navigation channels used by OCS traffic were not included in the direct impact analysis because there was either no impact (use of natural channel) or the direct impacts could not be determined accurately. For example, the lower Mississippi River is a natural waterway that undergoes frequent maintenance dredging. The spoil areas are usually not well-defined and, in fact, some of the spoil is used to create marsh. In addition, we did not include the Gulf Intracoastal Waterway (GIWW) in assessing direct impacts. We recognize that the GIWW is an important navigation route for OCS-destined vessels, and it does lie within the geographic boundaries of the study area; however, its construction predates OCS development, and justification for its continued maintenance is not predicated on service to OCS.

A fourth allocation estimate for direct impacts of OCS activities on navigation channels was derived by allocating 100% of the direct impacts of the Mermentau to Gulf Channel, Freshwater Bayou, Bayou Boeuf/Chene, Belle Pass, and the Houma Navigation Channel, plus using the highest estimate of percent OCS traffic on the remaining channels as a multiplier. This provided us with the highest allocation to OCS. We justified 100% allocation of the above channels to OCS on the reason for their construction or the COE justification for taking over jurisdiction.

Results and Discussion

Total wetland loss (OCS and non-OCS) for the period 1955/56 to 1978 *for the Louisiana portion of the study area* amounted to 288,414 ha (0.85%/yr). Of that total, a maximum of 73,905 ha of wetlands were lost because of direct impacts, of which a net of 18,110 ha were converted to canals and 28,245 ha were converted to spoil (Table 4-2). Direct impacts (spoil, canal, urban, and agricultural areas) accounted for a maximum of 25.6 % of the total wetlands lost during the 23-year period.

OCS pipelines and navigation channels that support OCS activities (excluding the GIWW) accounted for 19,010 ha (12,150 ha of canals and 6,860 ha of spoil) of the total wetland loss that occurred along the Louisiana coast from 1955 to 1978. This represented 25.7% of the wetland loss attributable to total direct impacts, 67% of wetlands converted to canals, 24.2% of wetlands converted to spoil, and 6.6% of the total wetland loss.

After allocating the *total direct impacts* of navigation channels to *OCS versus non-OCS* (see discussion below), OCS direct impact activities account for 11,589 to 13,631 ha (4.0 to 4.7%) of the total wetland loss that occurred in the Louisiana coastal zone from 1955/6 to 1978.

Habitat change attributable to the direct impacts of OCS activities for the *entire study area* since construction of the first OCS pipeline (1951) has amounted to 12,070 to 14,897 ha (Table 4-3). Of that total, some 12,012 ha are attributable to OCS pipeline and processing facility construction, and 58 to 2,885 ha are attributable to the OCS allocation of navigation channel construction (the range is derived from the different percent allocations used). Subsequent discussion treats navigation channels and pipelines separately because of inherent differences in their respective functions and impacts.

Table 4-2. Area (ha) by habitat type in Louisiana's coastal zone and the change in areal extent at each habitat type from 1955/56-1978.(derived from Wicker et al., 1980, 1981).

A. Habitat Description	Area (hectares) 1955/56	1978	Change 1955/56-1978
Agriculture	139,823	123,017	-16,806
Beaches and Dunes	4,758	3,081	-1,677
Bottom Land Hardwood Forest	37,502	29,003	-8,499
Brackish Marsh[a]	a	557,153	a
Canal	18,483	36,593	+18,110
Cypress Tupelo Swamp	172,243	145,120	-27,123
Fresh Aquatic Bed	533	8,096	+7,563
Fresh Marsh	543,654	270,773	-272,881
Fresh Open Water	66,766	40,356	-26,410
Fresh Shrub/Scrub	6,348	13,945	+7,597
Mangrove	63	2,955	+2,892
Mudflat	6,008	8,066	+2,058
Nearshore Gulf	76,505	72,7,21	-1,784
River, Stream, Bayou	34,528	35,782	+1,254
Estuarine Aquatic Bed	4	10,626	+10,622
Estuarine Open Water	1,797,600	2,011,917	+214,317
Saltmarsh[a]	[a]721,350	[a]181,394	[a]-539,966
Spoil	15,588	43,833	+28,245
Upland Forest	42,171	43,605	+1,434
Urban/Industrial	44,475	88,831	+44,356
	3,728,402	3,728,867	
B. Habitat Description			
Marsh	1,265,004	1,009,320	-255,684
Swamp	209,808	177,078	-32,730
Forest/Upland	48,519	57,550	+9,032
Aquatic Grass Bed/Mudflat	6,545	26,788	+20,243
Canal and Spoil	34,071	80,426	+46,355
Open Water	1,975,399	2,162,776	+187,337
Urban/Agriculture	184,298	211,848	+27,550
Beaches and Dunes	4,758	3,081	-1,677
	3,728,402	3,728,867	

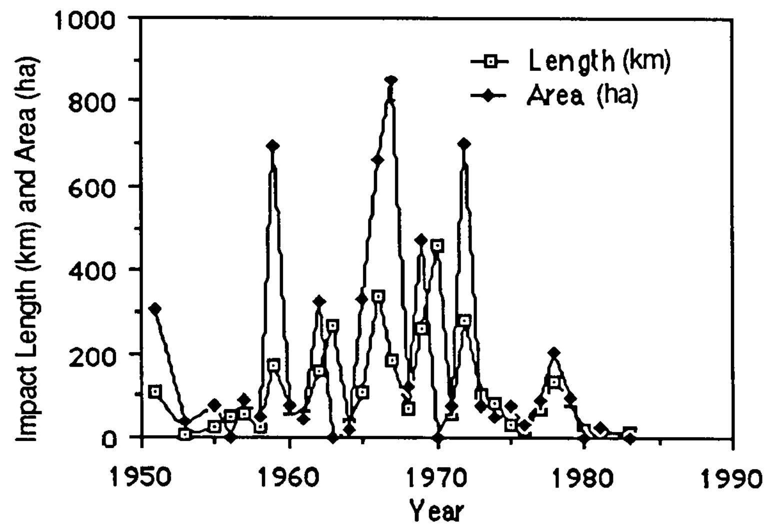
[a] Brackish marsh was not delineated as a separate category on the 1955 habitat maps but as "non-fresh marsh".

Table 4-3. Direct impacts attributable to OCS activities in the East Central Gulf of Mexico wetlands.

		Pipelines	Navigation Channels	Totals
Canal:	Length (km)	4,440	331	4,771
	Area (ha)	8,507	34-2,005	8,541-10,512
Spoil:	Length	849	242	1,091
	Area	3,466	23-880	3,489-4,346
Facilities:	Length	11.3	-	11.3
	Area	38.5	-	38.5
Totals[a]:	Length	4,827	331	5,158[a]
	Area	12,012	58-2,885	12,070-14,897

[a]Totals are not cumulative, e.g., pipeline can have both spoil length and canal length along the same section of line. Facility area can occupy spoil area.

Figures 4-1 and 4-2 depict direct impacts that result from OCS pipelines. Impact length and area through time generally reflect the periods of peak OCS pipeline construction activity. Impact length and area by habitat type depict some relative differences. Most notably, wetland habitats have a higher impact area relative to length, whereas the opposite is true for non-wetland and open water habitats. This implies that there are differences in the impacts per unit length by habitat type.



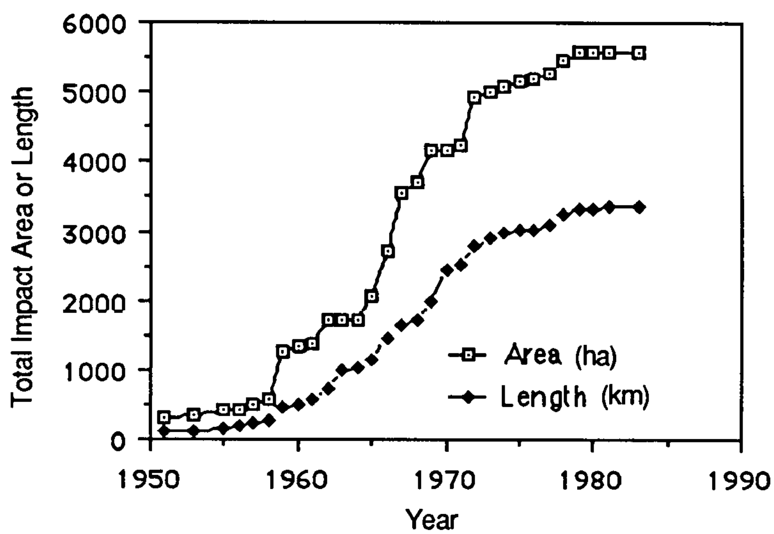


Figure 4-1. OCS pipeline impacts over time (70% sample). The upper panel shows annual impacts. The lower panel shows the cumulative impacts.

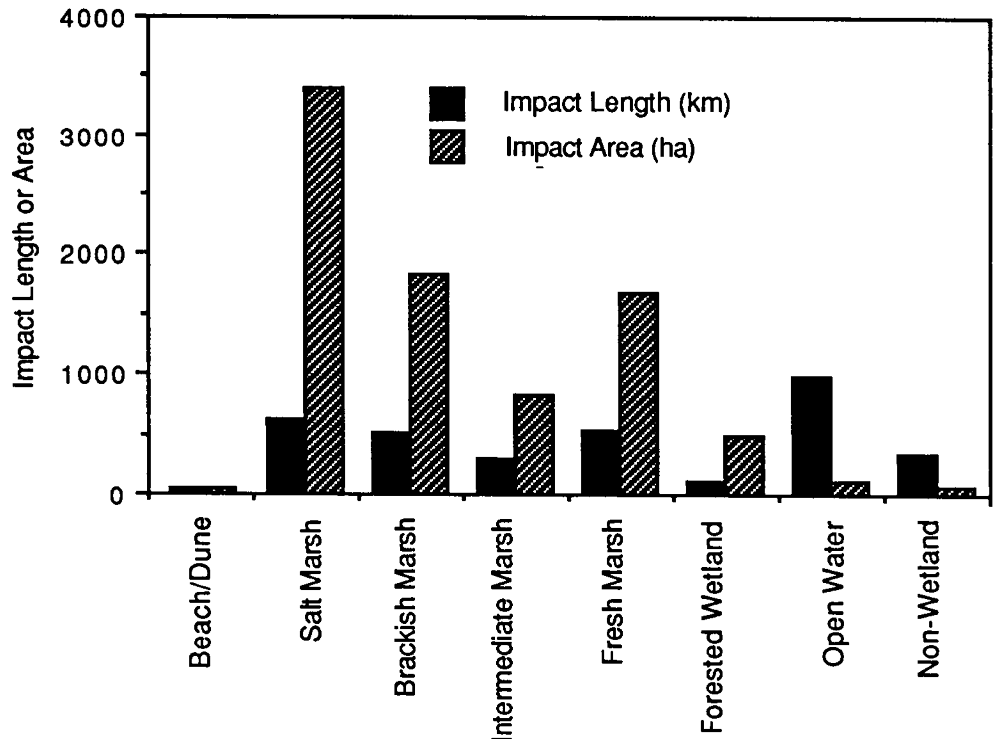


Figure 4-2. OCS pipeline impacts by habitat (70% sample).

If we assume that pipeline locations are distributed rather randomly across the study area, then we should expect significantly greater total impact in the Deltaic Plain than the Chenier Plain, simply because of the large areal differences in the two regions. Similarly, we would expect greater impacts in brackish marsh compared with forested wetlands. To normalize the areal differences in habitats and geologic regions we converted all impacts to impact area per unit length (ha/km) of pipeline. The remaining discussion on factors affecting the direct impact variability of OCS pipelines is based on comparisons of impact area per unit length.

In all models, construction type (backfilled versus non-backfilled) is a highly significant factor ($F<0.01$). Mean impacts for backfilled pipelines is 0.91 ha/km compared to 3.70 ha/km for non-backfilled (overall mean is 2.49 ha/km); 77% of the total length of pipelines are either backfilled or traverse through open water with no visible direct impacts. Thus, 55% of all the direct impacts attributable to OCS pipelines are accounted for by 23% of the pipelines (i.e., non-backfilled pipelines).

Geologic region (Chenier Plain versus Deltaic Plain) was also a highly significant factor ($P<0.01$) in determining variability. The interaction between geologic region and construction type is not significant ($P=0.47$). The Chenier Plain does contain a larger percentage of backfilled canals than the Mississippi Deltaic Plain. We were therefore concerned that the differences between the impacts for the two regions could reflect the greater prevalence of backfilled canals in the Chenier Plain rather than implied differences in geology. Backfilling pipelines in the Chenier Plain is more successful for reducing impacts than in the Mississippi Deltaic Plain (mean=0.68 and 1.05 ha/km, respectively; $P=0.04$), but the impacts for open pipelines are not significantly different for the two regions (mean=2.99 and 3.74 ha/km for

Chenier and Deltaic Plains, respectively). In the latter analysis only nine samples were available for the Chenier Plain.

Habitat type is also a highly significant factor in the variability of direct impacts on an area per unit length basis. Habitat type also has a highly significant interaction with construction type and geologic region, and the model with all possible interactions is highly significant (P=<0.01; $R^2 = 0.62$). The impacts by individual habitat for construction type and geologic region are shown in Table 4-4. Construction type does not significantly affect the beach/dune or upland/ridge habitats because almost all are backfilled. The effect on open water is highly significant only because non-backfilled canals cause spoil deposits above the water line. In some cases, these spoil deposits result in a net wetland gain, whereas backfilled canals have no visible direct impact; thus, an anomalous case results in which non-backfilled canals could be viewed as beneficial. All of the wetland habitat types experience significantly lower impacts in backfilled situations.

Table 4-4. Significance of construction type and regional geologic unit on pipeline impacts by individual habitat type. All numbers provided are mean values expressed in ha/km and are significant at the 95% level of confidence. NS = not significant; BF=backfilled, NBF=non-backfilled; CP=Chenier; Plain; DP=Deltaic Plain.

	Construction Type		Geologic Region		Interaction
Habitat Type	BF	NBF	CP	DP	
Salt Marsh	1.10	4.38	NS		NS
Brackish Marsh	1.47	4.06	1.17	3.20	NS
Intermediate Marsh	2.12	3.95	1.33	3.15	NS
Fresh Marsh	1.66	4.06	1.26	3.02	NS
Forested Wetland	2.16	3.84	NS		NS
Open Water	0.00	0.89	NS		NS
Beach/Dune	NS		NS		NS
Ridge/Upland	NS		NS		NS

The impact of geologic unit on individual habitats is highly significant only for fresh, intermediate, and brackish marshes. Impacts to upland/ridge and wetland habitats at the two geographical extremes (gulfward and landward) are not significantly related to geologic region.

The two continuous variables, time (age of pipeline) and diameter of pipeline, were modeled against the categorical factors using analysis of covariance. Comparing the effects of construction type, with impacts as the dependent variable and time as independent, showed that the intercepts (age=0) for backfilled and non-backfilled are not significantly different (P=0.80), but the slopes of the lines are highly significant (P<0.01) and are significantly different from one another with increasingly higher impacts for non-backfilled canals with increasing age. The R^2 for this model is 0.37.

Using the same analysis as above, but substituting regional geologic unit for construction type, we find that impact increases with age for both units (P <0.01); however, the rate for the Deltaic Plain is higher than that of the Chenier Plain. The R^2 for the model is 0.08 (P<0.01).

Using both of the preceding models, but substituting diameter for age, we find that the intercepts are significantly different for geologic units (the Chenier Plain has a lower initial impact) but not for construction type. There are highly significant differences in the slopes of the lines with greater impact with increasing diameter for the Deltaic Plain and for non-backfilled canals.

The above analysis of covariance models indicates that impact increases with pipeline age and diameter of pipeline and also that the rate of increasing impact is greatest for the non-backfilled canals in the Deltaic Plain. Surprisingly, the initial impact of backfilled canals is not significantly different from non-backfilled for our models; we expected non-backfilled to have a higher initial impact. Low R^2 values for the linear model indicate that impact area per unit length appears to increase with pipeline age and diameter in a non-linear fashion. New regulatory procedures were introduced in the early 1970s to reduce impacts primarily by encouraging that pipelines be backfilled. We believe that the abrupt reduction in impacts per unit length after the early 1970s accounts for most of the non-linear relationship with respect to pipeline age.

While impact area per unit length increases with age, we do not know if this reflects additional indirect impacts (e.g., erosion) or if the initial direct impact was greater for the earlier lines and the industry simply improved pipeline construction techniques or a combination of both. Our available evidence is only partially conclusive. We know that the trend does not merely reflect increased backfilling through time and thereby reduced impacts per unit length because the impacts of both backfilled and non-backfilled canals are independently related to pipeline age. With further reduction by geologic unit, we find that non-backfilled canals in both geologic regions have an increasing impact with age and that impacts in backfilled canals increase with age in the Deltaic Plain but not significantly in the Chenier Plain. The latter result is probably explained by the differences in geology. We conclude that the impact of backfilled pipeline canals in the Chenier Plain will not increase with age within the confines of the original pipeline construction right-of-way (e.g., subsidence of fill). In contrast, the impact of backfilled pipeline canals in the Deltaic Plain, constructed at the same time, with the same diameter, passing through the same habitat mix, increases with age within the confines of the construction zone. This strongly suggests that indirect impacts account for part of the increased impacts with age for the backfilled canals in the Deltaic Plain.

We compared dammed versus undammed non-backfilled pipeline canals to support or refute the hypothesis that boat traffic contributes to erosion and pipeline canal widening. We found no significant difference in direct impacts between the two. Thus, (1) boat traffic is not significant in pipeline canal widening; or (2) the dams do not work; or (3) the dams may have been constructed after, and as a result of, canal widening. This is a difficult problem to address because there are very few pipeline canals open to navigation. We could only positively identify six. Given that relatively small number, we doubt whether boat erosion of pipeline canals accounts for very much of the total wetland loss, even if it contributes significantly to the total impacts of those lines in which it occurs.

The total measured direct impact area of navigation channels with OCS-related traffic (with the exception of the GIWW) is 16,902 ha. Direct impacts averaged 50.81 ha/km and, as such, they are 20 times greater than the direct impacts resulting from pipelines on a per unit length basis. Spoil accounted for 11,379 ha, while an additional 5,523 ha of wetland and upland habitats were converted to canal area (open water). MRGO accounted for 45% (7,503 ha) of the direct impacts of the navigation channels measured. The ratio of spoil area to canal area for navigation channels is substantially higher than that for the pipelines (approximately 2:1 versus 0.4:1, respectively). The fact that navigation channels are not backfilled probably explains this difference. Furthermore, navigation channels are dredged to much greater depths than pipelines and produce more spoil deposits per unit length of dredging.

Allocation of direct impacts by individual navigation channel is provided in Table 4-5, and impacts by habitat type are provided in Table 4-6. Total direct impacts from navigation channels are concentrated in brackish and salt marsh habitats, whereas pipeline impacts are dispersed more evenly in all habitats. Because the habitat identification database is considerably more recent than the construction dates of many navigation channels, habitats may reflect indirect impacts of the navigation channel (e.g., increased salinity resulting in greater relative abundance of more saline wetland habitats).

Table 4-5. Allocation of direct impacts (ha) by navigation channel. Data have been rounded to nearest whole number. Primary data are largely derived from U.S. Army Corps of Engineers Reports to the Chief Engineer. See Materials and Methods section for allocation methodology.

Waterway/Allocation	Method 1	Method 2	Method 3	Method 4	Total Direct Impact[a]
Beaumont/Sabine Pass	3	21	45	45	3,003
Calcasieu Ship Channel	6	35	164	164	3,146
Lower Mermentau River	3	19	40	313	313
Freshwater Bayou	8	46	93	576	576
Vermilion Cut Off	2	9	53	53	90
Bayou Boeuf/Chene	9	52	188	670	670
Houma Navigation Channel	12	70	196	744	744
Belle Pass	5	31	53	146	146
Barataria Waterway	2	13	30	30	706
Empire Canal	1	1	2	2	7
MRGO	8	68	143	143	7,503
Totals	59	365	1,006	2,885	16,902

[a] Non-allocated. Includes 100% of direct impacts measured.

Table 4-6. Allocation of direct impacts (ha) by habitat type for navigation channels. Data have been rounded to nearest whole number. See Materials and Methods section for allocation methodology.

Habitat/Allocation	Method 1	Method 2	Method 3	Method 4	Total Direct Impact[a]
Salt Marsh	15	95	234	517	6,393
Brackish Marsh	16	104	301	744	6,172
Intermediate Marsh	11	65	168	738	755
Fresh Marsh	2	12	32	123	123
Forested Wetlands	2	22	45	161	161
Dune/Beach	<1	2	4	10	11
Open Water	4	23	77	77	2,745
Ridge/Upland	7	41	145	517	543
Totals	58	364	1,007	2,885	16,902

[a] Non-allocated. Includes 100% of direct impacts measured.

Variability in the relative differences of direct impacts of navigation channels is affected by the project design. Deeper draft channels are also wider, and the greater cross-sectional area results in larger spoil deposits.

The actual widths of navigation channels are invariably greater than project design; in some cases, actual width is approximately twice that of design widths.* These increases have been widely attributed to differences in initial construction dimensions and boat-wake erosion (Johnson and Gosselink, 1982). We compared the difference in actual width versus project design width against time and traffic count and the interaction of time and traffic and found no significant relationships. We recognize that there are problems with the data for this type of analysis and that the preceding factors, along with substrate type may, indeed, account for some of the variation.

More specifically, our traffic count data are from a single year; its quality is the best available, but its accuracy has serious shortcomings, and therefore it may not be representative of traffic over the long-term in which channel widening occurs. In addition, the reported channel design widths may be bottom widths and not surface widths. Finally, substrate type, for which sufficient properties data are unavailable, may be an important factor in accounting for the variation in widening rates among reaches of the same channel. A more accurate assessment of navigation channel widening rates requires a long-term field monitoring program, including boat traffic and type, and substrate variability, as well as accurate data on initial construction dimensions.

Conclusions and Recommendations

1. Total direct impacts accounted for an estimated 25.6% of total net wetland loss within the Louisiana portion of the study area from 1955/56 to 1978. Of the total direct impacts of 73,905 ha, OCS-related activities accounted for 11,589 to 13,631 ha of the wetland loss during the same time interval. Although this is a substantial areal loss, it represents only 4.0 to 4.7% of the total Louisiana wetland loss from 1955/56 to 1978, and 15.7 to 18.4% of direct impacts.

2. Direct impacts from OCS pipelines averaged 2.49 ha/km and totalled 12,012 ha. Direct impacts are variable and are related to construction technique, geologic region, habitat type, age and diameter of pipeline, and other factors that were not examined. Management for least impacts should include the following principles: (1) the pipeline should be backfilled; and, (2) wetland habitats should be avoided in favor of open water bodies and topographic highs (levees, cheniers).

3. Direct impacts from backfilled OCS canals in the Chenier Plain are not significantly related to age of pipeline; therefore, future indirect impacts within the pipeline right-of-way for this situation are not expected to result in a significant increase in pipeline impact. Direct impacts from non-backfilled canals in both regions and from backfilled canals in the Mississippi Deltaic Plain are positively related to the age of pipeline.

4. There is a significant relationship between impact and pipeline diameter, although the relationship is non-linear and the effect of diameter appears to be substantially less than that of other factors examined. Therefore, the best strategy is the installation of a larger diameter pipeline to allow for future expansion of product flow rather than a repetitive smaller diameter

* Data provided by K. Wicker, Coastal Environments, Inc., from a companion study currently being conducted for MMS. Primary data is largely derived from U.S. Army Corps of Engineers Reports to the Chief Engineer.

installation. In other words, reduce the number of pipelines rather than the size of individual lines.

5. The concept of using a corridor approach containing several pipelines, rather than a random distribution of individual lines to reduce impacts, appears valid for non-backfilled canals. However, no significant difference in direct impacts for corridor versus random distribution was found for backfilled canals. Backfilling reduced direct impacts by 75% and, therefore, is the preferred construction technique over corridor construction.

6. The current MMS published guidelines on the impacts of pipelines on Gulf of Mexico coastal marshes estimate that a pipeline will destroy about 6.28 ha/km (25 acres/mi). Our data show that the average impact for all OCS pipelines traversing an average habitat mix within the coastal zone is 2.49 ha/km. Because backfilling is now a standard procedure, a new pipeline will probably result in an average direct impact of 0.68 ha/km in the Chenier Plain and 1.05 ha/km in the Deltaic Plain. Even using a worst case scenario (salt marsh with no open water, non-backfilled canal in the Deltaic Plain), the average direct impact is 4.38 ha/km, a value substantially less than the published guideline.

7. Widening of OCS pipeline canals does not appear to be an important factor for total net wetland loss in the coastal zone because few pipelines are open to navigation and, for the examples found, the impact width was not significantly different than for open pipelines closed to navigation. Individual lines, however, may widen at locally significant rates.

8. Navigation channels account for a minimum of 16,902 ha of habitat change. Of the total change, 13,615 ha resulted in the loss of wetland and beach habitats. The maximum amount of habitat change attributable to OCS activities was 2,885 ha (17%) of which 2,293 ha (16.8%) was the loss of wetland and beach habitats. OCS traffic appears to comprise a relatively small percentage of the total commercial traffic using navigation channels; thus, the allocation of navigation channel impacts to OCS activities is small. Of the total habitat change, 13,652 ha (81%; Table 4-2) are attributable to MRGO, Calcasieu Ship Channel, and Beaumont Channel/Sabine Pass, all of which have very low OCS destination usage (Table 4-1).

9. Direct impacts per unit length of navigation channel average 20 times greater than pipelines. The dominant factor controlling the impacts per unit length is the project design. However, surface channel widths are substantially greater than design widths. A detailed long-term field investigation would be required to determine the validity of the commonly-held belief that channel widening is some function of tonnage, speed, and frequency of vessels, as well as edaphic factors.

PART III

SALTWATER INTRUSION

Saltwater intrusion has, in recent years, been referred to as a major cause of wetland loss in coastal Louisiana. The gradual encroachment of saline water is thought to occur in Louisiana as the Mississipi River Deltaic Plain subsides and sea level rises (Morgan, 1977). Vegetation maps indicate a northward movement of saline marsh types from 1968 to 1978 (Chabreck and Linscombe, 1982). Straight line canals are belived to accelerate the penetration of salt water into brackish and fresh marshes that would not normally be subject to such a change in salinity. However, there is not a comprehensive data set supporting the hypothesis of saltwater intrusion nor has there been sufficient documentation in the refereed literature.

The major task of the saltwater working group was to examine the saltwater intrusion issue. Because of the complexity of the question, a multidisciplinary approach was taken. There are several questions that must be answered before any statement regarding the importance of saltwater intrusion as a factor in land loss can be made. The saltwater working group was comprised of four seperate technical approaches, each designed to answer one of the basic questions listed below:

(1) Do man-made canals and channels promote saltwater intrusion?

(2) Is there evidence of increasing salinities in the historical record?

(3) What is the relationship between the salinity in the bayou and the salinity in the adjacent marshes?

(4) What are the salinity levels that adversely inpact vegetation?

The following chapters present the analyses and results of each technical approach designed to answer the above questions. The questions serve only as an outline, and the reader will find that individual researchers have added to the basic questions as needed. For example, the question of saltwater effects on vegetation could not be addressed without also addressing the effects of submergence, because both of these may occur.

Following the individual chapters is a concensus chapter that synthesizes the results of each chapter in light of the basic question of whether saltwater intrusion (either natural or man-induced) is a major factor in land loss.

Chapter 5

Saltwater Intrusion Modeling: The Role of Man-made Features

by

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The objectives of this investigation were: (1) to identify the behavior of saltwater intrusions in the major channels supporting OCS oil and gas activities along the Gulf of Mexico in Louisiana; (2) to estimate the length of saltwater intrusion under various physical forcing functions (river discharge, tidal amplitude, and wind speed and direction); and (3) to simulate the patterns of salinity distribution in channels for various channel dimensions.

General Types of Salinity Profiles

The flow regimes in coastal Louisiana areas are governed by various forces that determine the degree of stratification and the shape of salinity patterns. The primary forces (Ippen, 1966) are: (1) the gravitational force caused by density variations between fresh water from an upland drainage basin and saline water entering from the Gulf of Mexico to produce a net Gulfward transport of fresh water; (2) the frictional forces at the interface and at the channel bottom. The boundary conditions imposed on the flow regime are: (1) the freshwater inflow from the upstream reach of the channel; (2) the tidal force resulting from the oscillatory tidal velocity at the channel entrance; and, (3) the surface wind force generated by weather activity.

In coastal Louisiana, the depths of estuaries and bays are quite shallow, usually less than 10 m. Under these conditions, fresh and salt water appear to be well-mixed. In the upstream navigation channel, vertical salinity profiles range from partially to highly stratified.

The major physical forcing functions that influence the length and shape of salinity distribution in coastal channels are river discharge, tidal amplitude, and prevailing winds. Four types of salinity profiles, depending on the channel dimensions and the relative magnitude of the above forcing functions, have been characterized by Stommel and Farmer (1953) as shown in Figure 5-1. The four types are well-mixed, partially stratified, highly stratified, and saline wedge.

A well-mixed water column occurs in a shallow channel, when the tidal current dominates and the river discharge is low. The salinity profile is quite uniform in the water column (Figure 5-1a). The velocity profile follows a simple logarithmic distribution.

A partially stratified channel occurs when the river discharge is moderate and tidal range is high; there is a difference in salinity between the surface and bottom. The salinity gradient is apparent in the water column (Figure 5-1b).

A highly stratified channel occurs when the river discharge is high and tidal range is moderate. The salinity difference between the surface and bottom becomes large, and a sharp salinity gradient is formed in the water column (Figure 5-1c).

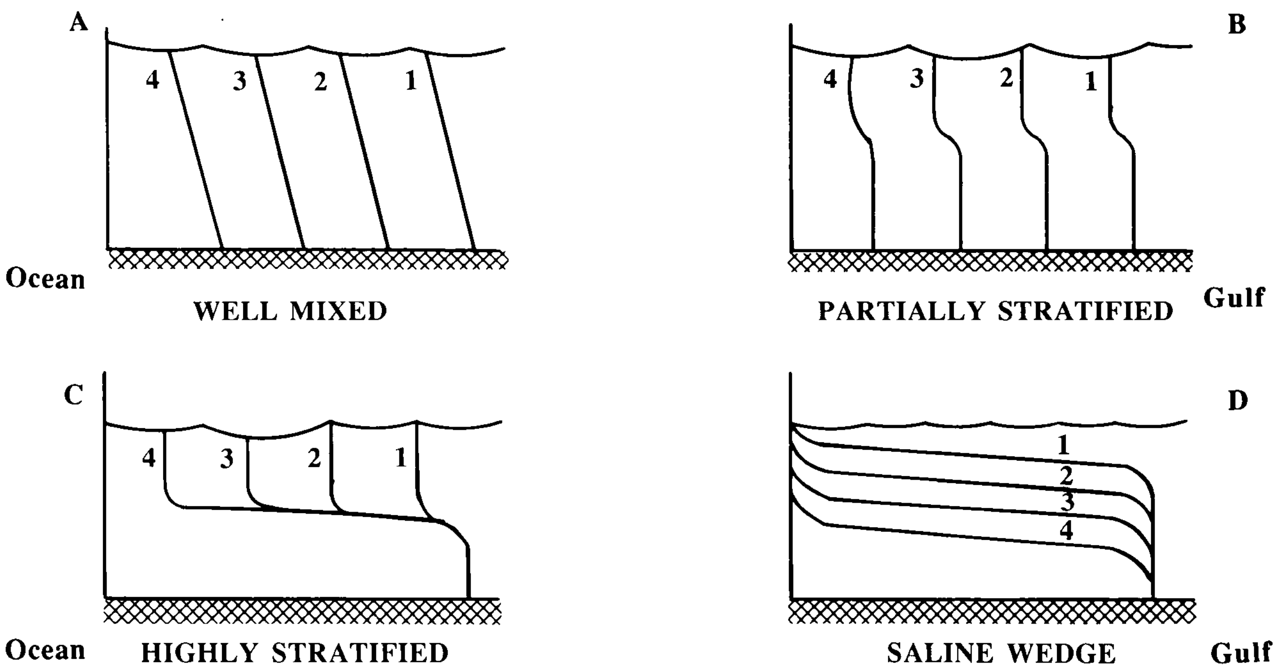


Figure 5-1. General types of salinity profiles (after Stommel and Framer, 1953).

A saline wedge occurs in a deep channel, when the river discharge is high and tidal range is low. The salt water intrudes beneath the fresh water, and a salt wedge is formed. As the advancing phase of front ceases, the saline wedge becomes stationary or arrested (Figure 5-1d).

Field Measurements of Salinity Distribution

Three coastal channels, Houma Navigation, Bayou Petit Caillou (a natural channel), and Calcasieu Ship Channels, were selected for this study. They were selected based on their locations (Figure 5-2), the dimensions of channel width and water depth, and their relative significance to offshore oil and gas activities.

Houma Navigation Channel

Houma Navigation Channel, one of the 25 OCS-related channels identified in Chapter 4, connects Terrebonne Bay with the Intracoastal Waterway near Houma (Figure 5-3). The 40 km channel is well-maintained for navigation by dredging, hence the channel depth and width are quite uniform along the entire channel. The water depth is 6.6 m, and the channel width is about 100 m, with a typical width to depth ratio of 15. Figure 5-3 shows that the Houma Navigation Channel is not an isolated channel but has many branches that connect to Lake Boudreaux via Bayou Grand Caillou. Lake Boudreaux may act as a sink or a source of salt to adjacent water bodies during the high or low flow season of the year. Field measurements were made at selected locations depending on the salinity distribution in the channel during the time of the sampling period.

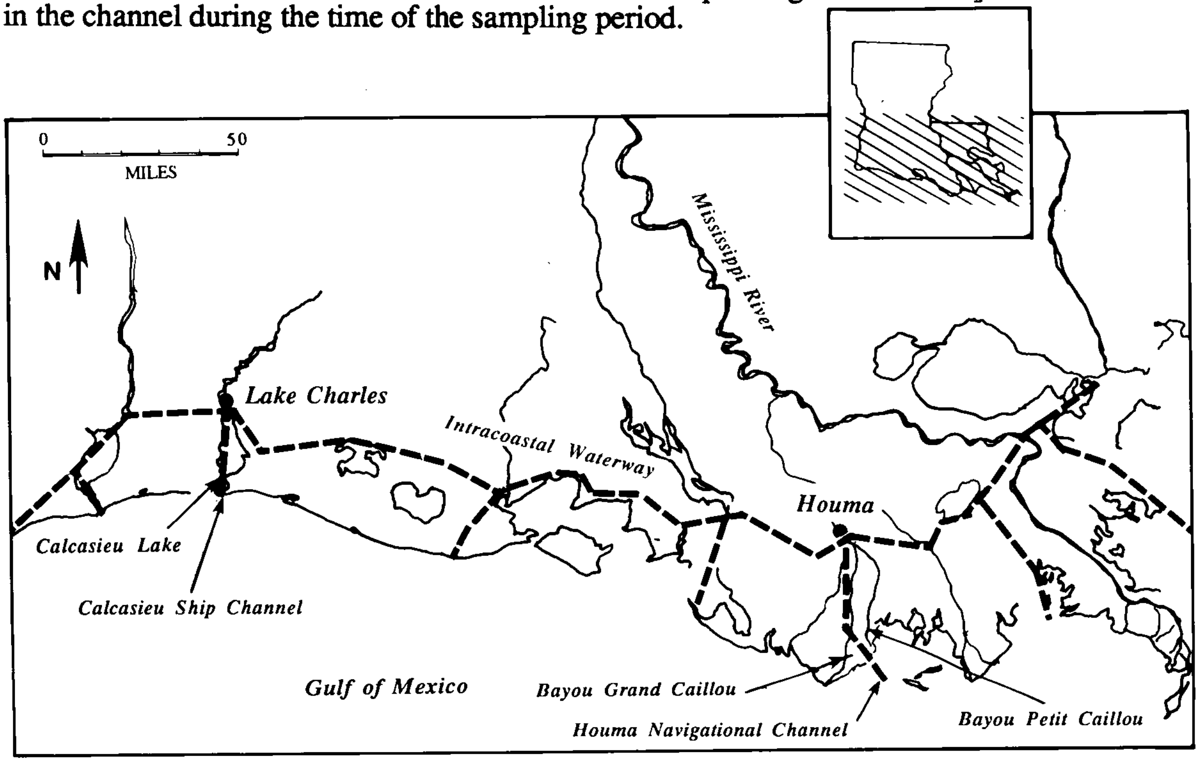


Figure 5-2. Map showing the locations of three selected channels: (a) Houma Navigation Channel; (b) Bayou Petit Caillou; (c) Calcasieu Ship Channel.

Two field trips were made to the Houma Navigation Channel site. The salinity distributions from the two surveys are presented in Figure 5-4. The first trip was conducted on September 20-21, 1986, when the freshwater discharge measured at the upstream reach was relatively low, about 0.5 cms/m. The saltwater locus (5 ppt) reached north of Houma about 40 to 50 km from the channel entrance (Figure 5-4a). A fairly high salinity, 18 to 19 ppt, was observed in the first 10 to 15 km northward from Cocodrie, Louisiana (Gulf side). Fifteen km upstream, the salinity decreased to 15 ppt.

The second trip was made on October 17-18, 1986. This time, the salt water (1 ppt) only reached to Celestin (25 km from the channel entrance) because of the relatively large freshwater discharge of about 1.7 cms/m from upstream (Figure 5-4b). Half of the channel length was freshwater.

The velocity profiles measured at the upstream reach of the channel, near Houma (40 km from the channel entrance), are shown in Figure 5-5. The first survey shows that water cannot move at the bottom part of the channel because of the salinity gradient. The second survey shows a logarithmic velocity profile because of a high influx of fresh water from the upstream drainage basin. The salinity is close to 0 ppt north of Celestin, and there is no saltwater front beyond that. Figure 5-5 clearly demonstrates that the salinity gradient does exert a significant influence on the vertical velocity distribution.

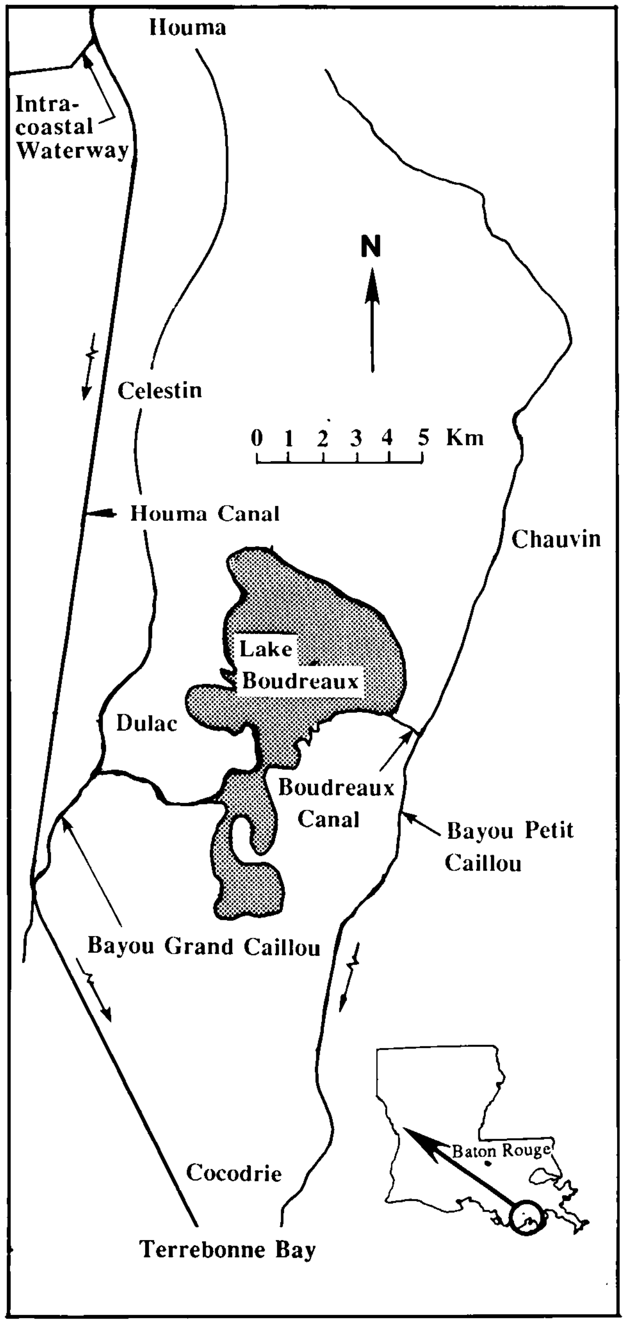


Figure 5-3. Location map of Houma Navigation Channel and Bayou Petit Caillou.

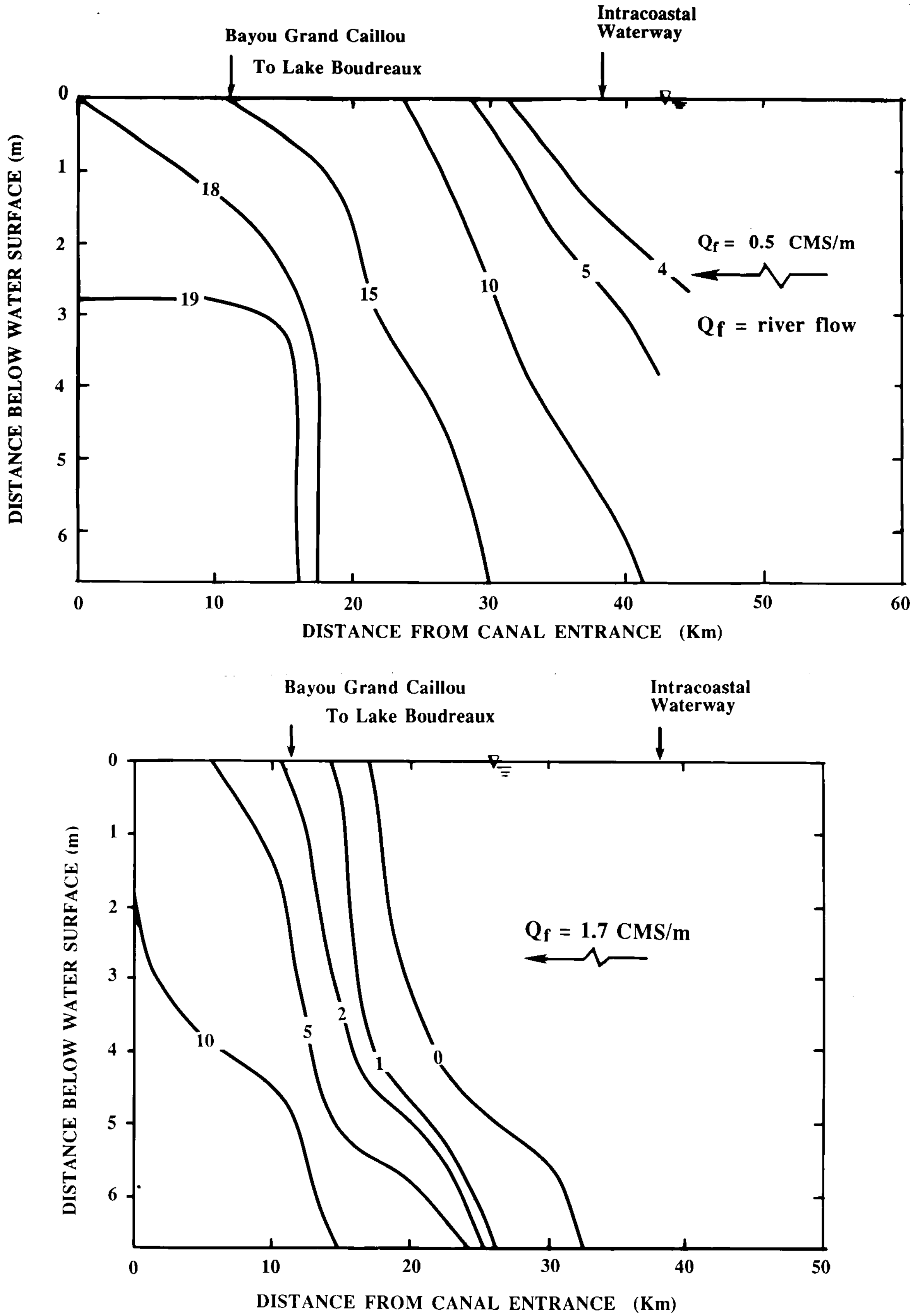


Figure 5-4. Salinity distribution in the Houma Navigation Channel: (a) field measurements on September 20-21, 1986; (b) field measurements on October 17-18, 1986.

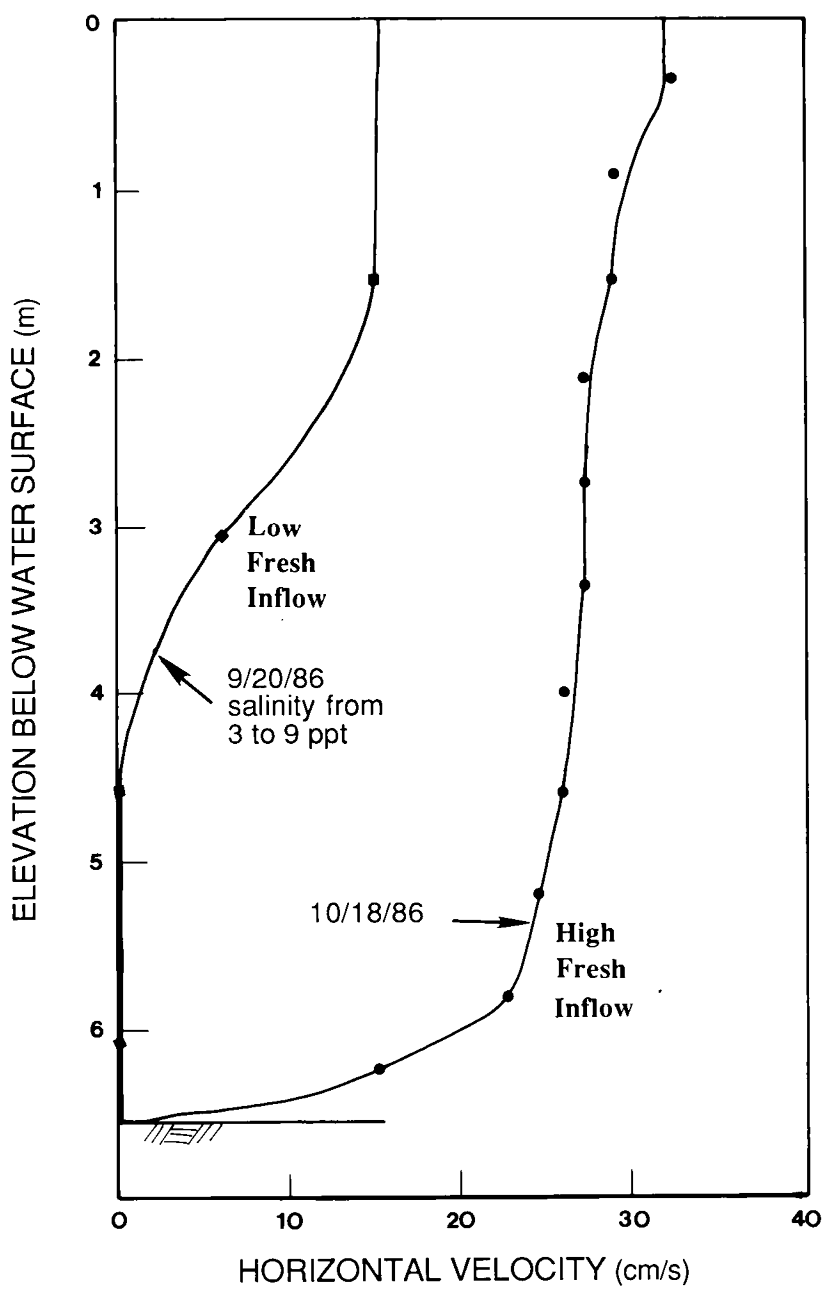


Figure 5-5. Velocity profiles in the Houma Navigation Channel near Houma: (a) field measurement son September 20, 1986; (b) field measurements on October 18, 1986.

Bayou Petit Caillou

Bayou Petit Caillou, a relatively shallow and narrow channel, typical of a natural bayou, is used primarily for small boat navigation. It is located about 10 km east of Houma Navigation Channel (Figure 5-3) and runs into Terrebonne Bay near Cocodrie. The water depth for the first 20 km from the channel entrance is only about 3 m, and it becomes shallower further upstream. The channel width is about 50 m with a typical width to depth ratio of 15. Similar to Houma Navigation Channel, Bayou Petit Caillou also connects to Lake Boudreaux via Boudreaux Canal (Figure 5-3).

A field trip was made to this site on October 19, 1986. Field measurements were conducted at selected locations depending on the salinity distribution in the channel at the sampling time. Figure 5-6 shows the salinity distribution derived from field measurements: uniform (10 to 14 ppt) from Cocodrie to Boudreaux Canal (about 15 km from the bayou entrance) and then decreasing rapidly to 1 ppt at an upstream distance of 10 km (25 km from the bayou entrance).

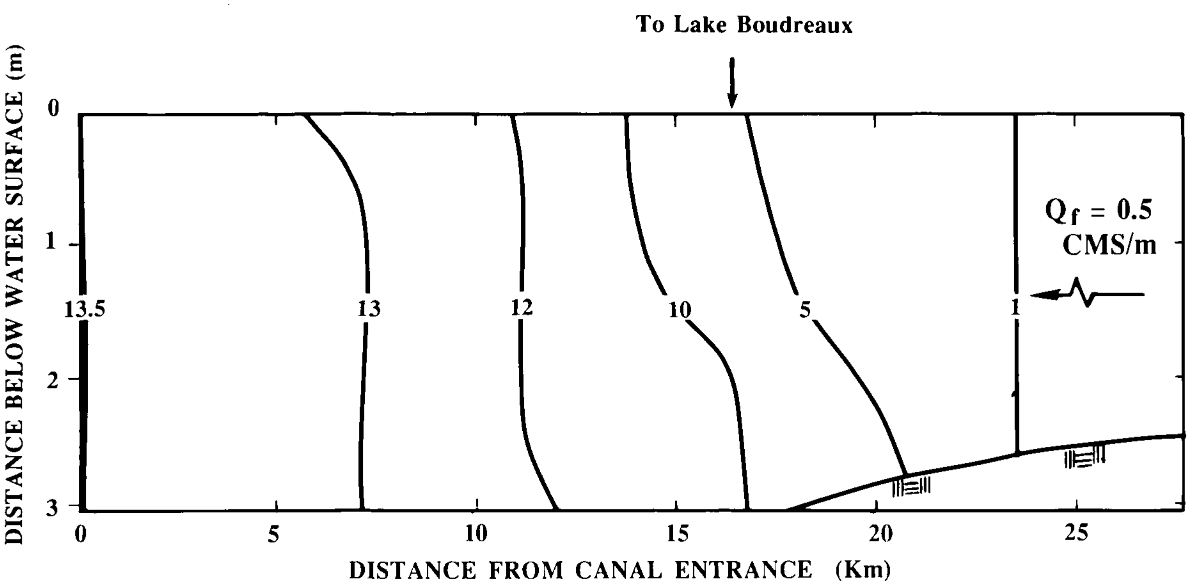


Figure 5-6. Salinity distribution from Bayou Petit Caillou field measurements on October 19, 1986.

Bayou Petit Caillou is closely connected to Lake Boudreaux via Boudreaux Canal, a short (1.5 km) and relatively deep channel (3 m). The lake has a salinity of about 10 ppt, and it supplies much of the salt water to Bayou Petit Caillou. Hence, a fairly uniform salinity (10 to 14 ppt) is maintained for the first 15 km of the bayou. Further upstream, salinity decreases quickly because of the lack of the saltwater supply.

Calcasieu Ship Channel

Calcasieu Ship Channel, located on the southwestern coast of Louisiana (Figure 5-2), plays an important role in the waterborne commerce activity and a relatively minor role in OCS-related activities. The channel is oriented from north to south and runs into the Gulf of Mexico near Cameron (Figure 5-7). It passes through Calcasieu Lake and then connects to the Intracoastal Waterway, about 40 km from the channel entrance.

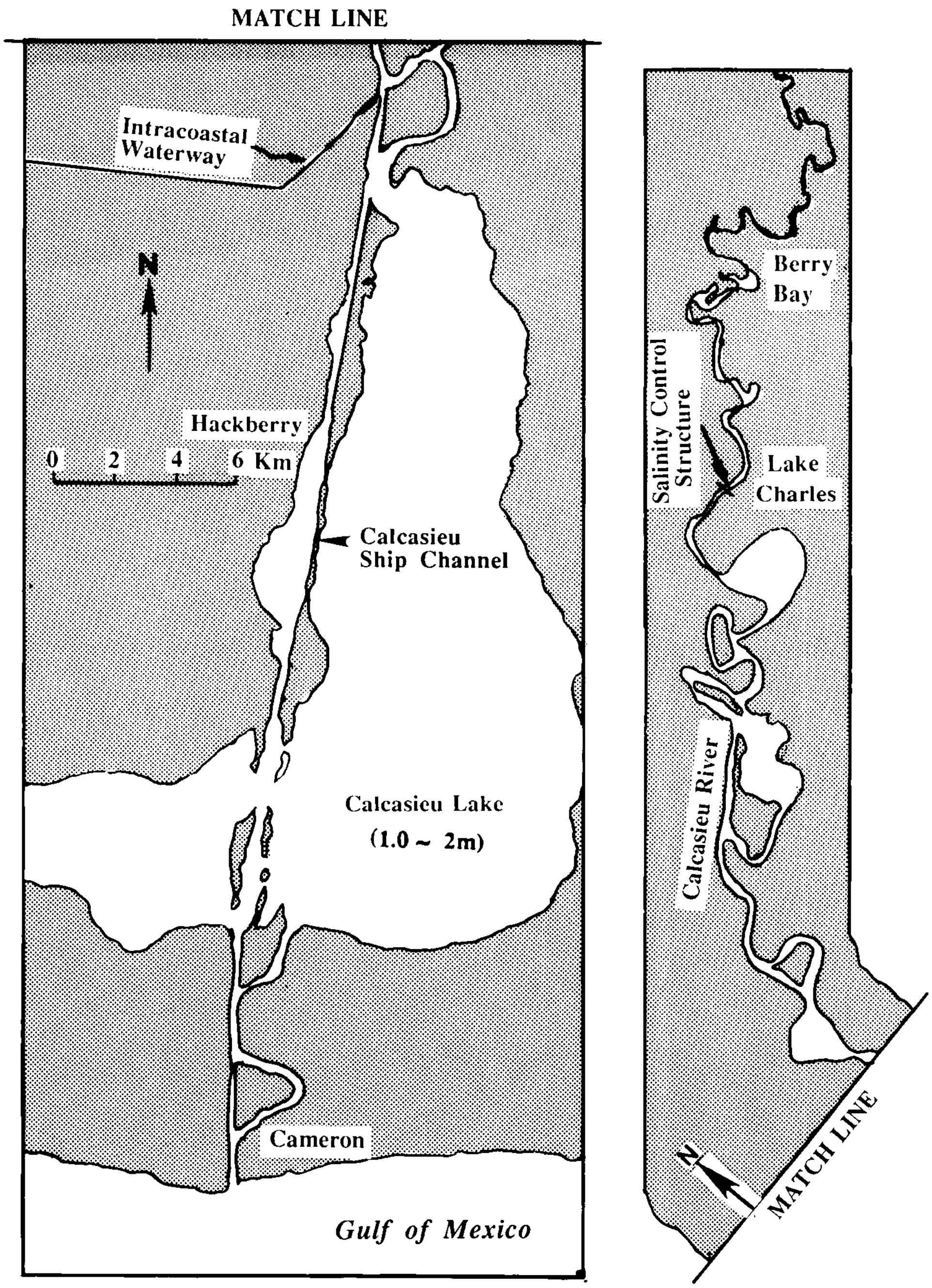


Figure 5-7. Location map of Calcasieu Ship Channel in southwest Louisiana.

From Cameron to Lake Charles (60 km), the channel is well-maintained by dredging for larger and more modern vessels traveling through the ship channel and to the Gulf. The channel has a typical water depth of 12.5 m and a width of 200~240 m, representing a width to depth ratio of 15~20. North of Lake Charles, the water depth decreases to about 10 m. While the total length of the ship channel is about 60 km, the natural river extends northward for more than 100 km.

Two field trips were made to this site. Field measurements were taken at selected locations along the channel, depending on the salinity distribution during the sampling period. The first trip was conducted on November 15-16, 1986. At this time, the freshwater discharge from the upstream reach was very low: only 0.1 cms/m. The salt water locus (5 ppt) reached Berry Bay (75 km from the channel entrance). The second trip was on March 2-3, 1987. This time the saltwater (less than 1 ppt) was only detectable near Hackberry (25 km from the channel entrance) because of the large freshwater discharge, 6.7 cms/m, from upstream.

Two salinity distributions are shown in Figure 5-8. In Figure 5-8a, during the low freshwater inflow period (0.1 cms/m), the saltwater front was expected to reach beyond Berry Bay. Note that Calcasieu Lake (dashed line in Fig. 5-8) is relatively shallow, only about 1.5 m compared with the water depth (12.5 m) in Calcasieu Ship Channel. The lake does not significantly affect the overall salinity intrusion; it only reduces the surface salinity caused by lateral diffusion (Figure 5-8a). Figure 5-8b shows the salinity distribution during the high freshwater inflow period (6.7 cms/m). Half the length of the channel is dominated by fresh water. A slightly high salinity regime 10 to 19 ppt confined at the channel bottom was observed. This unstable behavior of stratified flow might be caused by the increased mixing process in the channel bottom (Wang, 1975).

Velocity and salinity profiles were measured at the upstream reach of the channel, near a salinity control structure (70 km from the the channel entrance) during the November 1986 trip (Figure 5-9). The profile shows the behavior of saltwater intrusion at low freshwater discharge. The velocity profile indicates the reverse direction of flows from south to north about 4 m below the water surface.

In the second measurement on March 2-3, 1987, velocity and salinity profiles were measured near Hackberry (25 km from the channel entrance; Figure 5-10). Because of the high freshwater discharge (6.7 cms/m) during this sampling period, we observed a rapid change in salinity profile during a tidal cycle. The reverse velocity in the opposite direction near the bottom was largely induced by the salinity gradient.

Summary of Field Measurements

Table 5-1 summarizes the field trips conducted for the three selected channels during the study period. For Bayou Petit Caillou field data indicate that salinity is well-mixed on the Gulf side. Houma and Calcasieu Ship Channels exhibit partially stratified salinity distribution during low flow conditions. The degree of stratification is less under moderate to high flow conditions.

Figure 5-11 displays the longitudinal salinity gradient and vertical salinity profiles at the Houma Navigation Channel study site obtained from the September, 1986 field trip. In Figure 5-11a, both the depth-averaged salinity gradient and the surface salinity gradient gave a value of 0.0006 ppt/m.

The salinity wedge, arrested or stationary, in the lower Mississippi River, as reported by Balloffet and Borah (1985), has not been found in these three channels. The Mississippi River has a large drainage basin of 3.22 x 106 km^2 (Coleman, 1981), an average river discharge of 15.6 x 10^3 m^3/sec, and a much deeper channel at the river mouth (15 m). These environments are favorable for the formation of an arrested saline wedge. For other coastal channels in Louisiana, however, the salt water in the water column exhibits partially stratified behavior, in general.

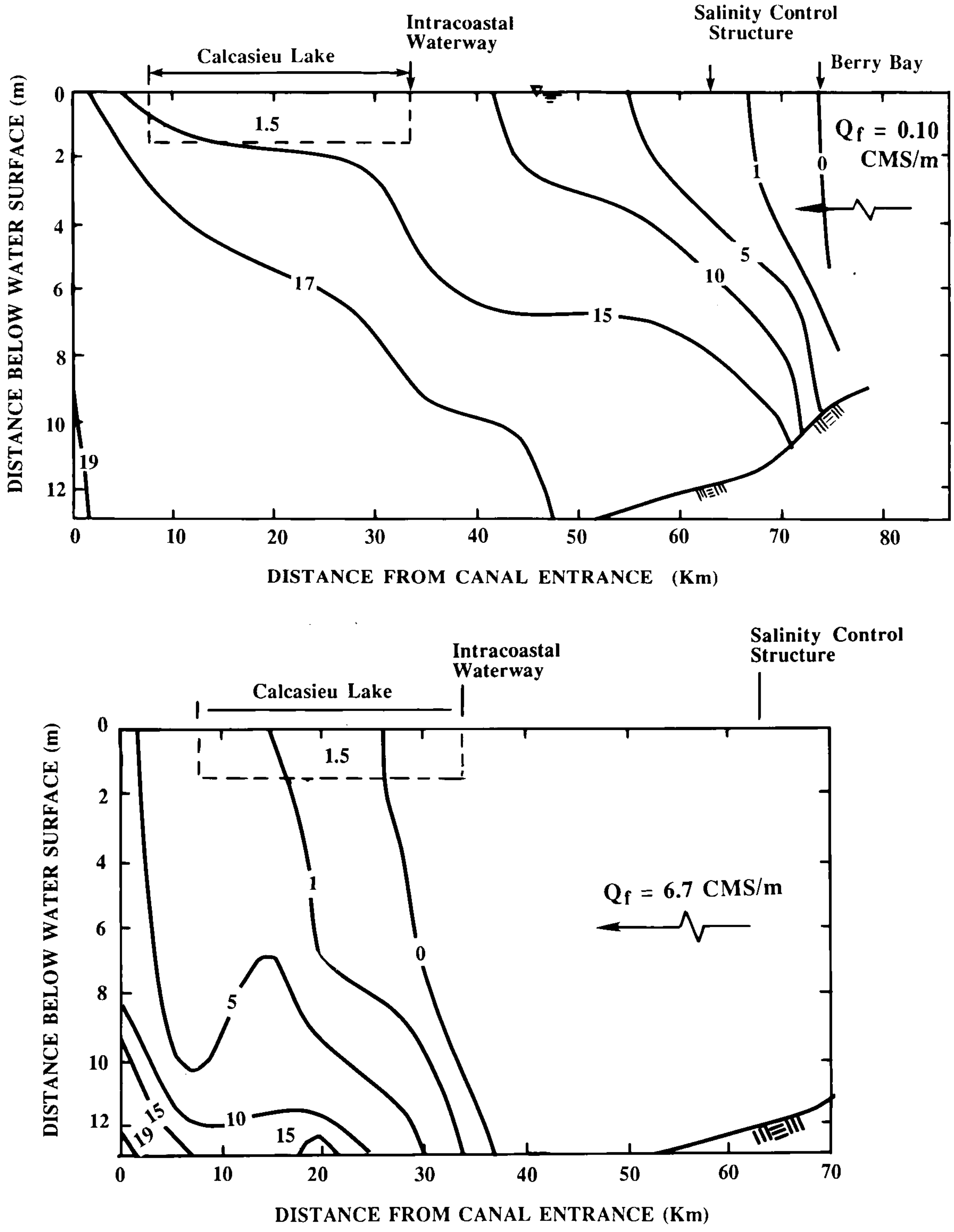


Figure 5-8. Salinity distribution in the Calcasieu Ship Channel: (a) field measurements on November 15-16, 1987; (b) field measurements on March 2-3, 1987.

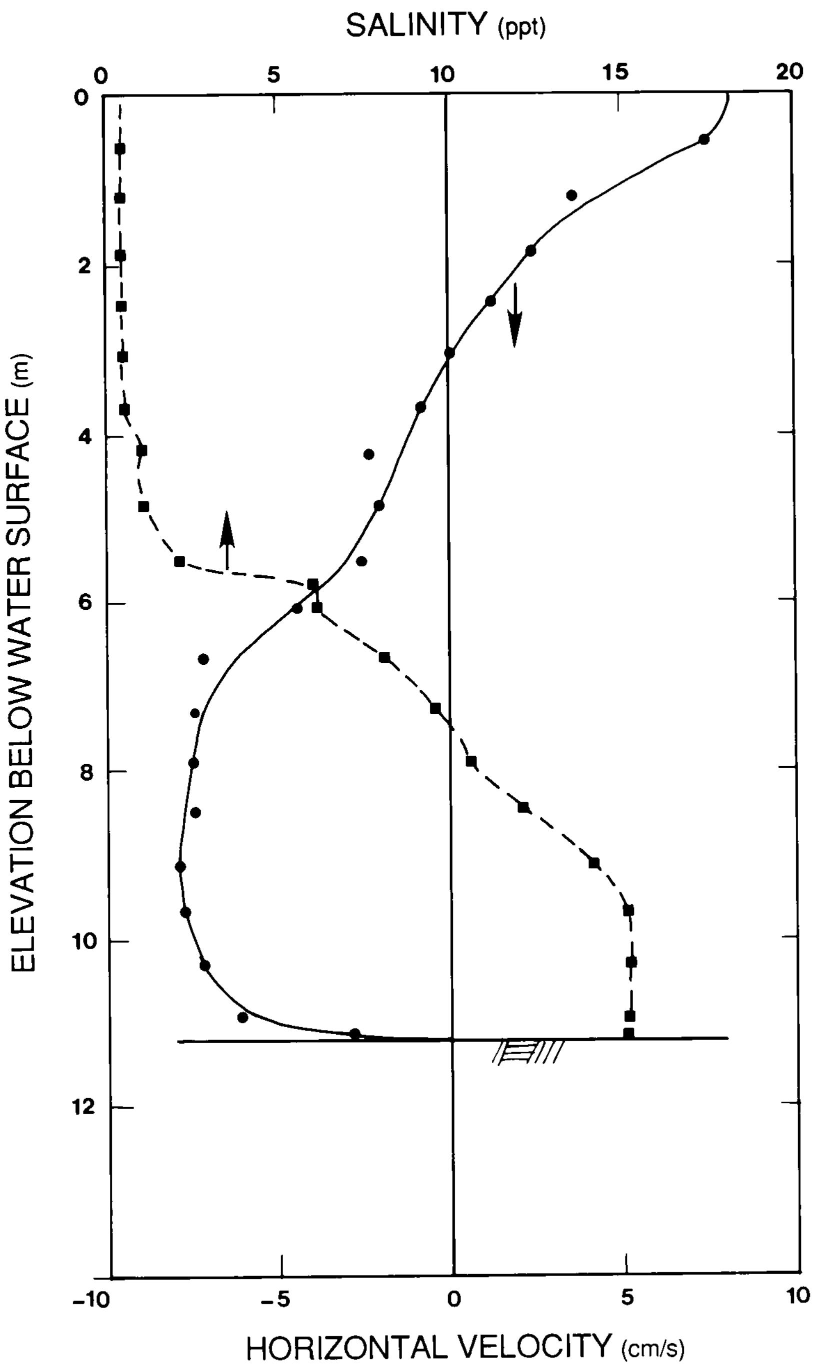


Figure 5-9. Velocity and salinity profiles in the Calcasieu Ship Channel near the salinity control structure (70 km from the channel entrance); field measurements on November 15-16, 1986.

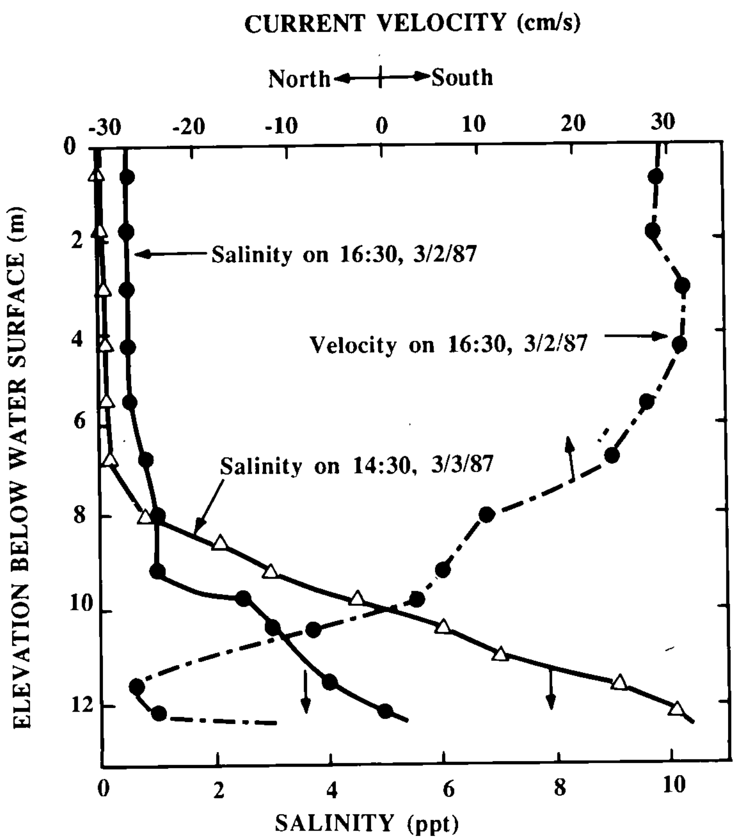


Figure 5-10. Velocity and salinity profiles in the Calcasieu Ship Channel, near Hackberry (25 km from the channel entrance); field measurements on March 2-3, 1987.

Table 5-1. Summary of field trips conducted for selected channels during the study period, 1986-1987.

	Physical Forcing Functions			Channel Dimensions			Length of Saltwater Front	
	Fresh Water (cms/m)	Tidal Amp (m)	Local Wind (m/s)	Water Depth (m)	Channel Width (m)	Channel Length (km)	Detectable Front (km)	Salt Locus (km)
Date of Field Sampling	Q	A	W	H	B	L	(≥ 0 ppt)	(5 ppt)
Houma Navigation Channel								
Sept. 20-21, 1986	0.5	0.1	neg				40-50	45
				6.6	100	50-60		
Oct. 17-18, 1986	1.7	0.1	neg				20-30	25
Bayou Petit Caillou								
Oct. 19, 1986	0.5	0.1	neg				15-25	20
				3.0	50	50-60		
Calcasieu Ship Channel								
Nov. 15-16, 1986	0.1	0.3	neg				70-80	75
				12.5	200	60-100		
Mar. 2-3, 1987	6.7	0.1	neg				20-30	25

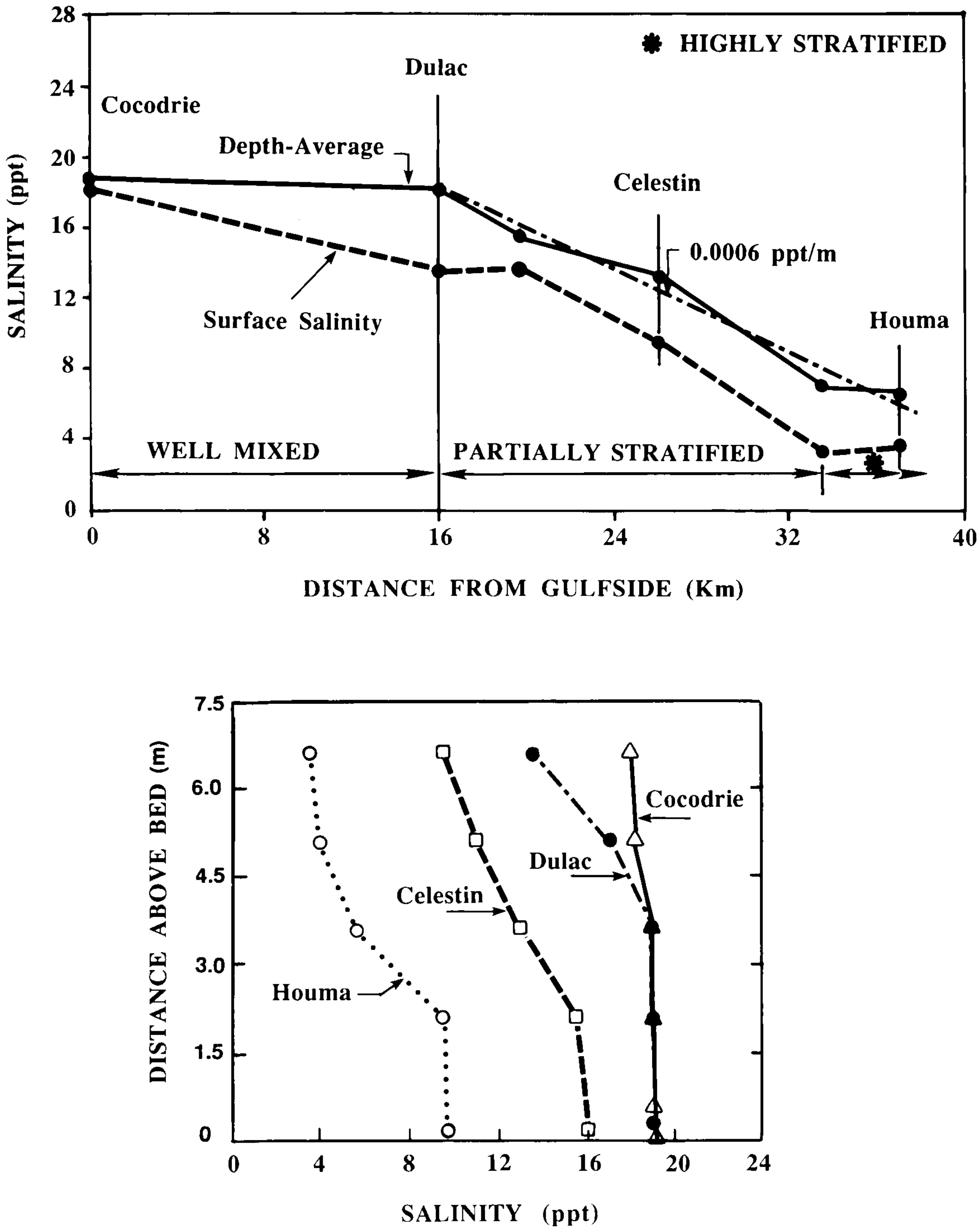


Figure 5-11. Salinity measurement along the Houma Navigation Channel on September 20-21, 1986; (a) longitudinal salinity gradient; (b) vertical salinity profiles.

Numerical Model of Saltwater Intrusion

The type of salinity distribution in Louisiana coastal channels ranges from well-mixed to partially stratified to highly stratified. The existence of this range stimulated us to develop our own numerical model to compute the velocity and salinity fields in channels that directly and indirectly support OCS activities.

The five basic governing equations (Appendix C) together with the specified boundary conditions (Appendix C) control the motion of water and salt in the channel. Hence, the momentum equation, the continuity equation, and the salinity conservation equation should be solved simultaneously because the water flow and salt transport are coupled together.

We developed a two-dimensional numerical model, which is a numerical scheme with variable grid size (Figure 5-12). The grid sizes Δx_i and Δy_j were chosen so that better resolutions at the channel bottom, the free surface, and the section of interest within the flow domain could be obtained.

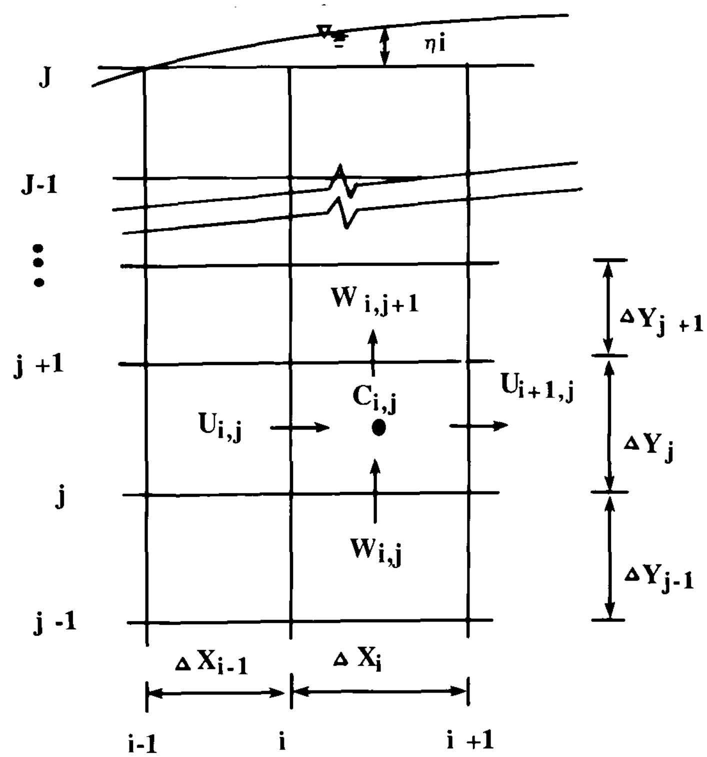


Figure 5-12. The two-dimensional numerical model grid system.

The subscripts, i and j, indicate the number of grids in the x and y direction, respectively. The horizontal velocity, u, is specified at the center of each vertical segment; the vertical velocity, w, is specified at the center of each horizontal segment; and, the salinity concentration, C, in terms of water density, ρ, as defined in Appendix C, Eqn. C.5, is specified at the center of each segment.

Computer Simulations of Salinity Distribution

The numerical model comprises a two-dimensional, laterally-averaged, and semi-implicit hydrodynamic model coupled with a salt flux transport model. In the model, the coefficients of eddy viscosity and diffusion are taken as functions of the water depth and the Richardson Number (see Appendix C for details); hence, they are varied with space and time domains and have to be determined by model calibration with field data.

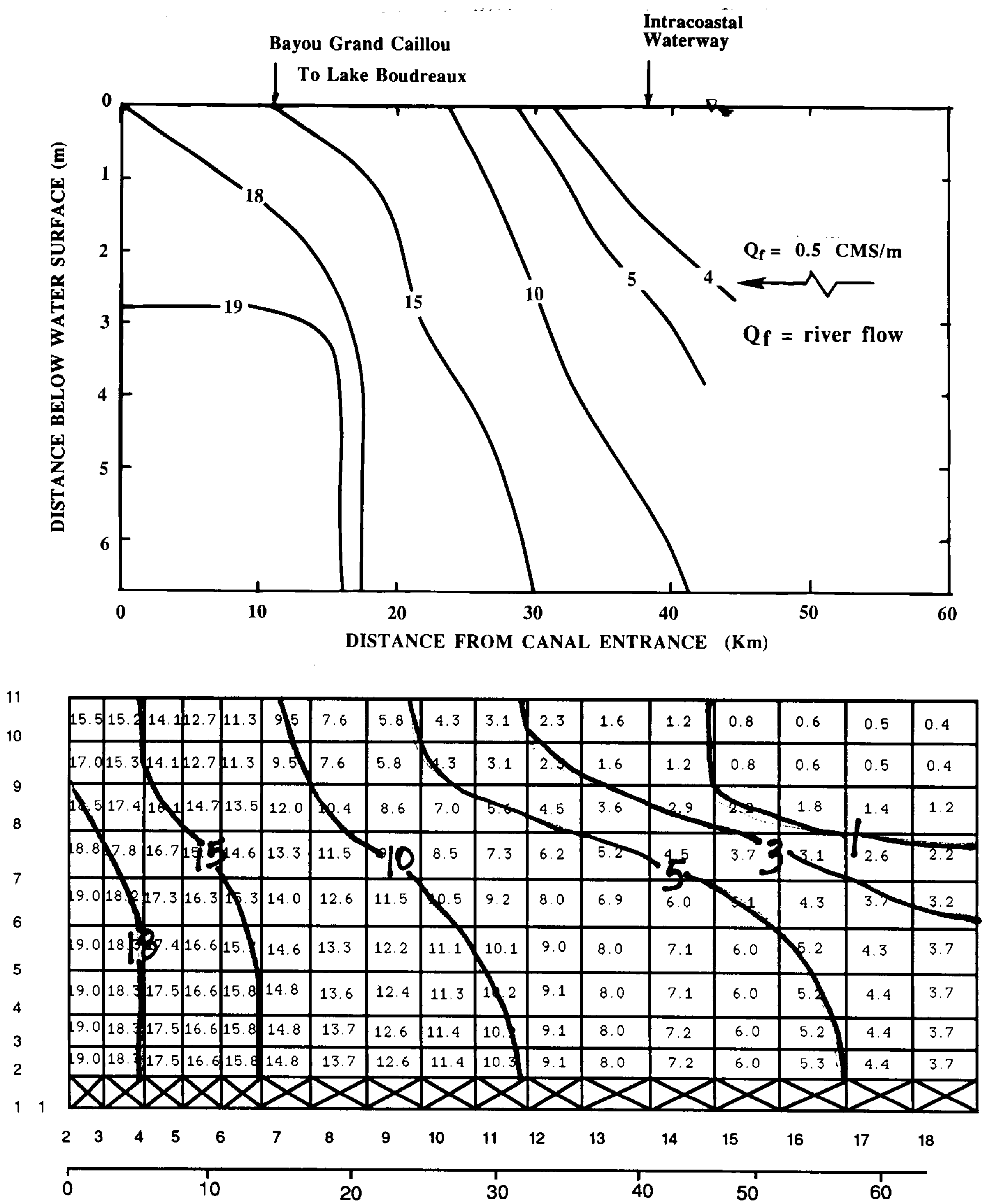
Model Calibration

The process of model calibration involves: (1) selecting an appropriate grid size in both x and y directions; (2) choosing a time step interval that fulfills the stability criteria; and, (3) adjusting the internal parameters so that computed results agree with field measurements. For each selected channel, two data sets are required; one is used for model calibration. The second independent data set, using the calibrated parameters, verifies the model.

For the Houma Navigation Channel, the first field survey (September 20-21, 1986) was used as basic input data for model calibration. Table 5-2 summarizes the values of grid size, time step interval, tidal period and amplitude, and river discharge used in sequential calibration runs. Figure 5-13 shows calibrated results of salinity distribution in the channel.

Table 5-2. Model calibration for Houma Navigation Channel (field data from September 20-21, 1986).

Run Number	1	2	3	4	5	6
Title	Conti 1	Case 1	Case 1	Case 1	Case 1	Case 1
No. of Total Time Step						
nott	2500.0	4500.0	4500.0	4500.0	4500.0	2500.0
Number of X-Grid Point						
nx	18.0	21.0	26.0	26.0	26.0	26.0
Number of Y-Grid Point						
nz	13.0	12.0	12.0	12.0	12.0	12.0
Tidal Period (sec)						
period	89,280.0	89,280.0	89,280.0	89,280.0	89,280.0	89,280.0
Time Step Interval (sec)						
deltt	186.0	186.0	186.0	186.0	186.0	180.0
Wind Velocity (m/sec)						
wind-speed	0.0	0.0	0.0	0.0	0.0	0.0
Tidal Amplitude (m)						
amplitude	0.1	0.1	0.1	0.1	0.1	0.1
River Discharge (cms/m)						
fresh-q	0.5	0.5	0.5	0.5	0.5	0.5
Kx-Diffusion=C0*Ny(m2/sec)						
coeffi0 x e6	0.02	0.2	0.5	1.0	2.0	1.8
Nx-Viscosity (m2/sec)						
coeffi1 x e4	1.0	1.0	1.0	1.0	1.0	1.0
Von-Karman Constant						
coeffi2	0.4	0.4	0.4	0.4	0.4	0.4
Ny-Viscosity (dimensionless)						
a1	0.3	0.3	5.0	5.0	5.0	5.0
q1	-0.5	-0.5	-0.5	-1.0	-1.0	-1.0
Ky-Diffusion (dimensionless)						
a2	10.0	30.0	30.0	30.0	30.0	30.0
q2	- 5.0	- 5.0	- 5.0	- 5.0	- 5.0	- 5.0



Time (minutes) = 13950.0
Maximum Velocity = 14.13 cm/s

Figure 5-13. Model calibration for salinity distribution in the Houma Navigation Channel: (a) field measurements on September 20-21, 1986; (b) calibrated results from numerical model.

Only one sampling trip was conducted for Bayou Petit Caillou on October 19, 1986. The bayou dimension, freshwater discharge, and tidal and wind conditions are listed in Table 5-3. This data set was used for model calibration for the upstream reach of the bayou (15 km from the channel entrance, north of Boudreaux canal; Figure 5-3) because a fairly uniform salinity is maintained at the downstream reach of the bayou. Table 5-3 also summarizes the process of model calibration by varying different parameters sequentially. Figure 5-14 shows the calibrated results of salinity distribution in the bayou where a general agreement between field measurements and results of model calibration is obtained.

Table 5-3. Model calibration for Bayou Petit Caillou (field data from October 18, 1986).

Run Number	1	2	3	4
Title	Caillou	Caillou	Caillou	Bayou
No. of Total Time Step				
nott	5000.0	5000.0	5000.0	2500.0
Number of X-Grid Point				
nx	11.0	16.0	16.0	16.0
Number of Y-Grid Point				
nz	10.0	9.0	9.0	9.0
Tidal Period (sec)				
period	89,280.0	89,280.0	89,280.0	89,280.0
Time Step Interval (sec)				
deltt	93.0	93.0	93.0	90.0
Wind Velocity (m/sec)				
wind-speed	0.0	0.0	0.0	0.0
Tidal Amplitude (m)				
amplitude	0.1	0.1	0.1	0.1
River Discharge (cms/m)				
fresh-q	0.5	0.5	0.5	0.5
Kx-Diffusion=C0*Ny(m2/sec)				
coeffi0 x e6	0.2	0.3	0.2	0.2
Nx-Viscosity (m2/sec)				
coeffi1 x e4	1.0	1.0	1.0	1.0
Von-Karman Constant				
coeffi2	0.2	0.3	0.4	0.4
Ny-Viscosity (dimensionless)				
a1	0.3	0.3	0.3	0.3
q1	- 1.0	- 1.0	- 1.0	- 1.0
Ky-Diffusion (dimensionless)				
a2	2.0	15.0	20.0	20.0
q2	- 4.0	- 4.0	- 4.0	- 4.0

Similarly, for Calcasieu Ship Channel, the first sampling trip (November 15-16, 1986) was used for model calibration. The channel dimensions and the meteorological conditions during the sampling period are listed in Table 5-4. Figure 5-15 presents the results of model calibration and field measurements of salinity distribution in Calcasieu Ship Channel, where a close resemblance was obtained.

In Tables 5-2, 5-3, and 5-4, the sequential adjustment of internal parameters of eddy viscosity and diffusion coefficients plays an important role in model calibration for achieving a close agreement of model results with field measurements. Increasing α_1, α_2 and decreasing Φ_1, Φ_2, in the functional forms of N_y and K_y, dramatically increases vertical stratification.

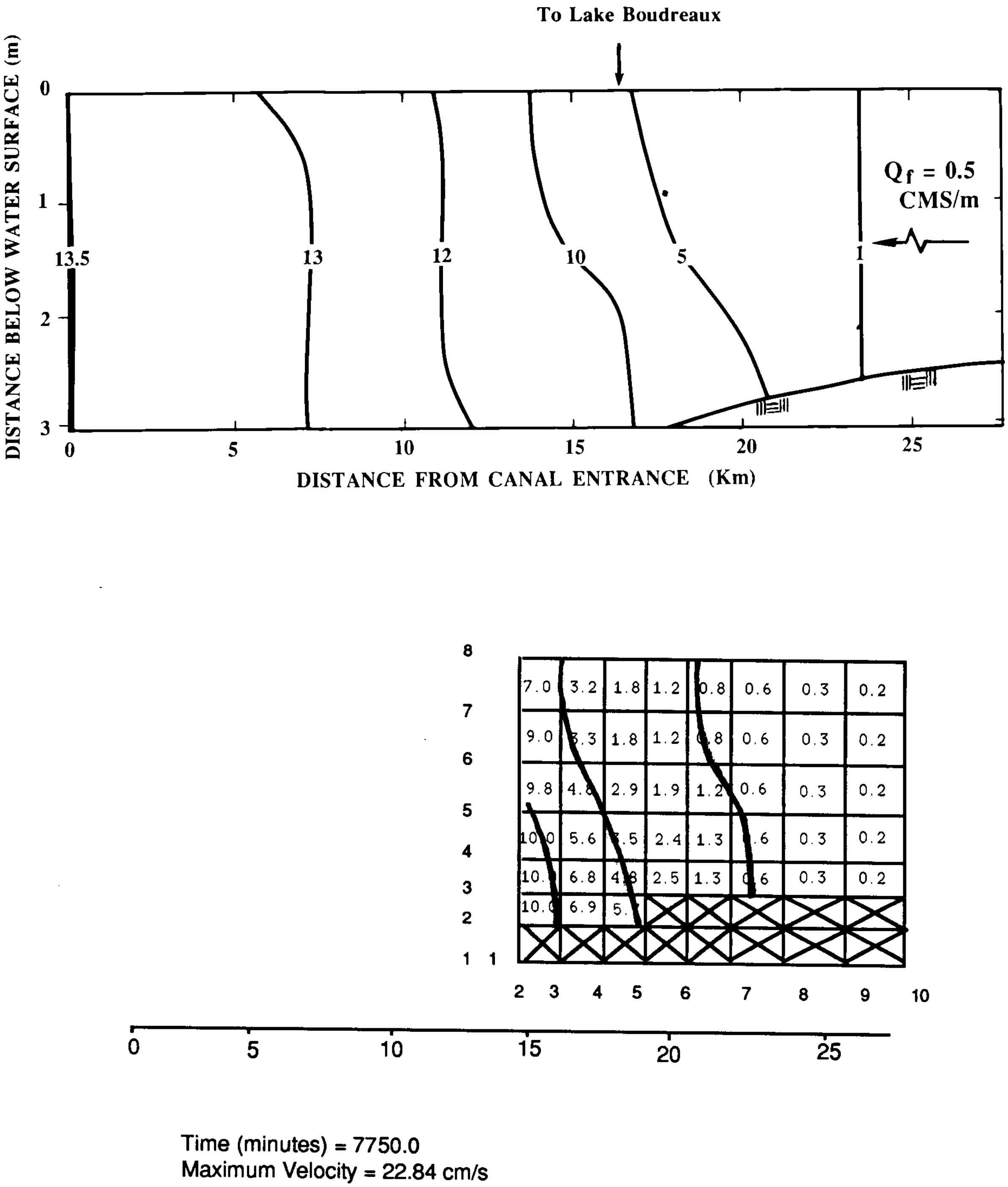


Figure 5-14. Model calibration for salinity distribution in Bayou Petit Caillou: (a) field measurements on October 19, 1986; (b) calibrated results from the numerical model.

Table 5-4. Model Calibration for Calcasieu Ship Channel (field data set on November 15-16, 1986).

Run Number	1	2	3	4
Title	Case 1	Case 1	Case 1	Case 1
No. of Total Time Step				
nott	2500.0	4500.0	4500.0	2500.0
Number of X-Grid Point				
nx	26.0	26.0	26.0	26.0
Number of Y-Grid Point				
nz	16.0	16.0	16.0	16.0
Tidal Period (sec)				
period	89,280.0	89,280.0	89,280.0	89,280.0
Time Step Interval (sec)				
deltt	186.0	186.0	186.0	180.0
Wind Velocity (m/sec)				
wind-speed	0.0	0.0	0.0	0.0
Tidal Amplitude (m)				
amplitude	0.3	0.3	0.3	0.3
River Discharge (cms/m)				
fresh-q	0.1	0.1	0.1	0.1
Kx-Diffusion=C0*Ny(m2/sec)				
coeffi0 x e6	0.06	0.06	0.6	0.6
Nx-Viscosity (m^2/sec)				
coeffi1 x e^4	1.0	1.0	1.0	1.0
Von-Karman Constant				
coeffi2	1.0	1.0	0.4	0.4
Ny-Viscosity (dimensionless)				
a1	0.3	0.3	2.0	2.0
q1	- 1.0	- 1.0	- 1.0	- 1.0
Ky-Diffusion (dimensionless)				
a2	15.0	15.0	30.0	30.0
q2	- 4.0	- 4.0	- 5.0	- 5.0

Model Verification

During model calibration, sequential adjustments were made to the physical coefficients to allow the calibrated results to closely resemble field measurements. An independent data set was then used for model verification using the calibrated parameters. The second field survey (October 17-18, 1986) for the Houma Navigation Channel site was used for verification. Figure 5-16 presents the results of salinity distribution in the channel and compares the field measurements and the verified results. A general agreement was obtained.

For the Calcasieu Ship Channel, the second field trip (March 2-3, 1987) was used for model verification. From the experience of previous simulation runs, we found that a simulation period of five tidal cycles was adequate for the tidal regime, velocity field, and salinity distribution to become established. Therefore, the simulated results from the fifth cycle were used for comparative purposes.

Figure 5-17 depicts the verified results of salinity distribution in Calcasieu Ship Channel. The verified results of salinity distribution in the channel, caused by the

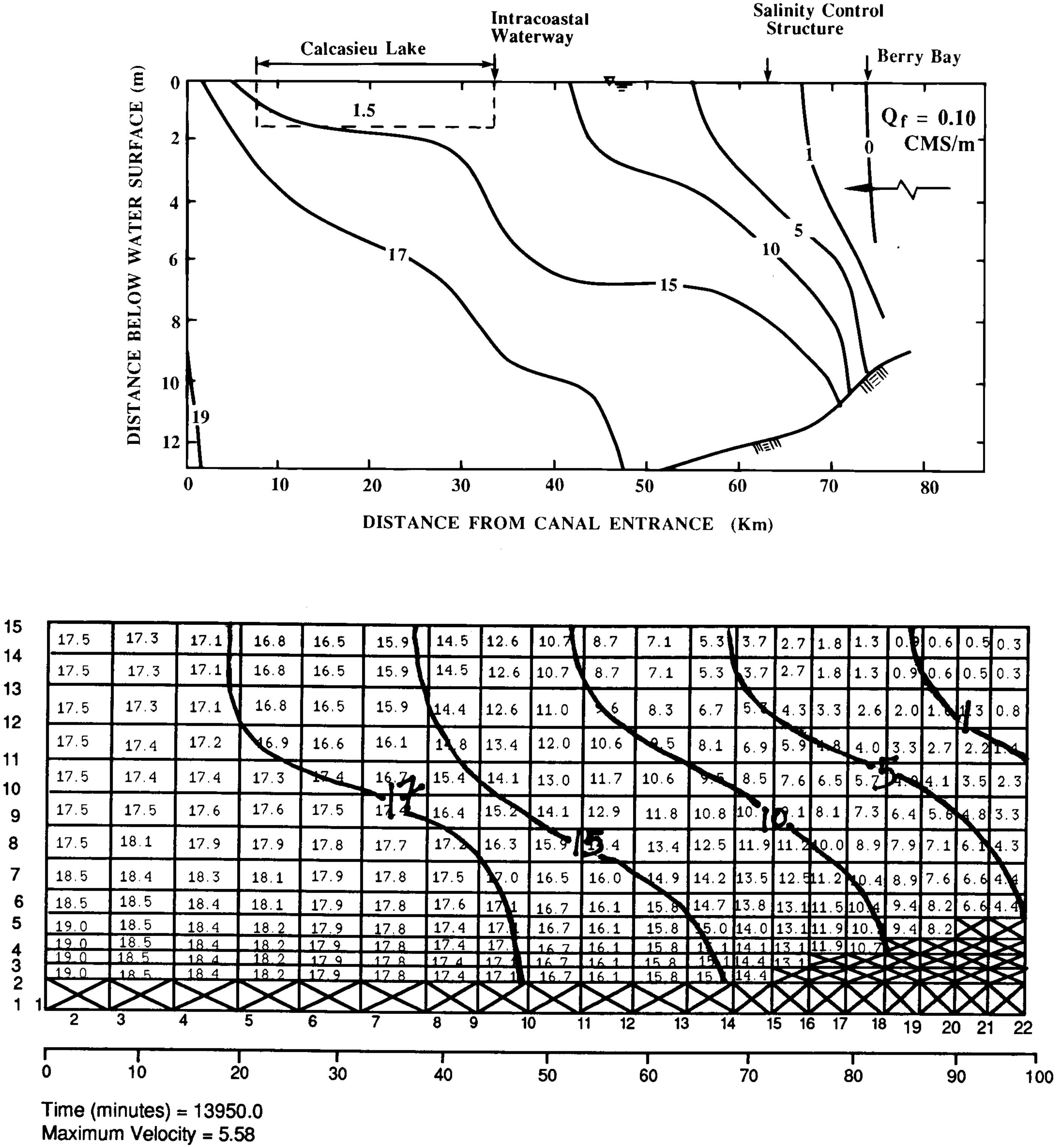
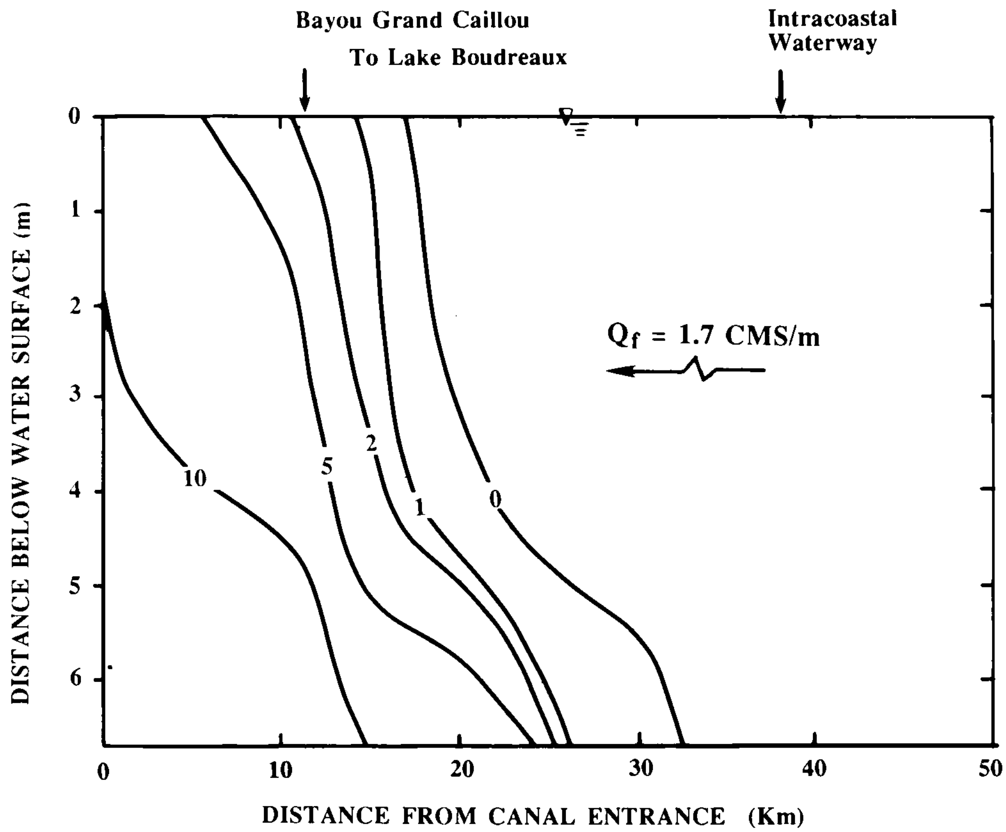


Figure 5-15. Model calibration for salinity distribution in the Calcasieu Ship Channel: (a) field measurements on November 15-16, 1986; (b) calibrated results from numerical model.



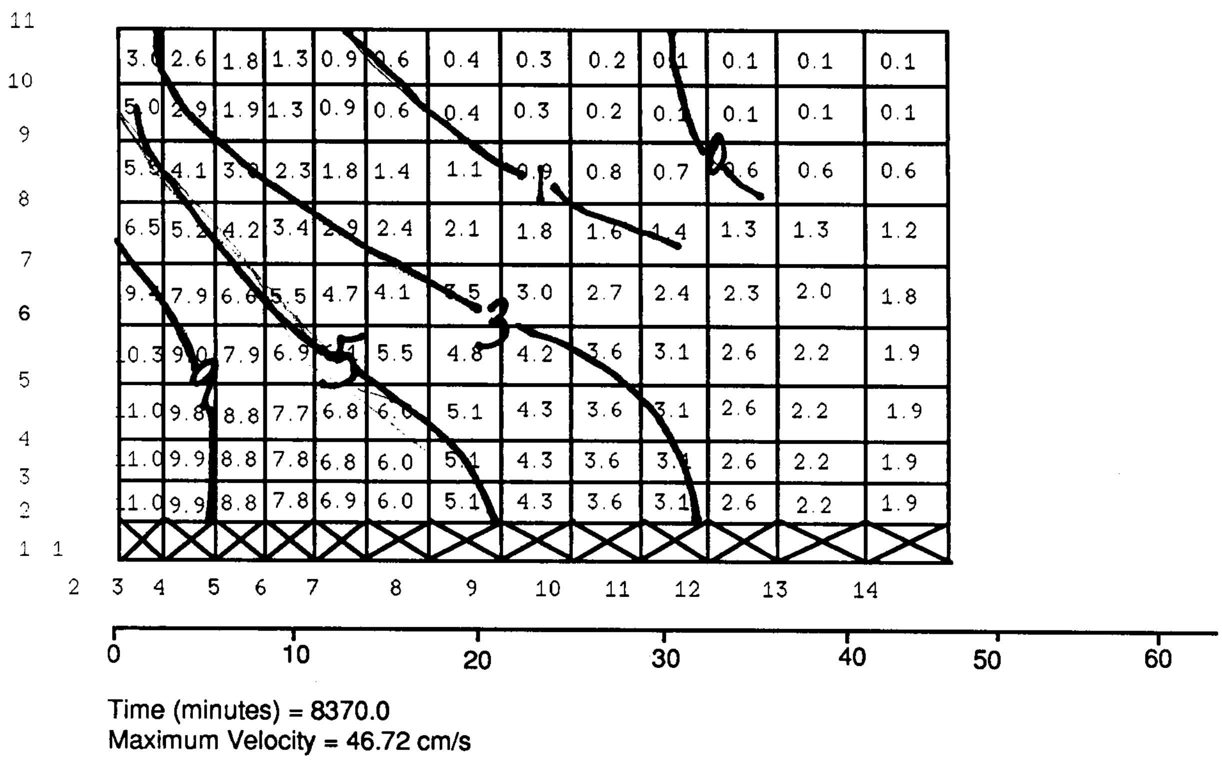
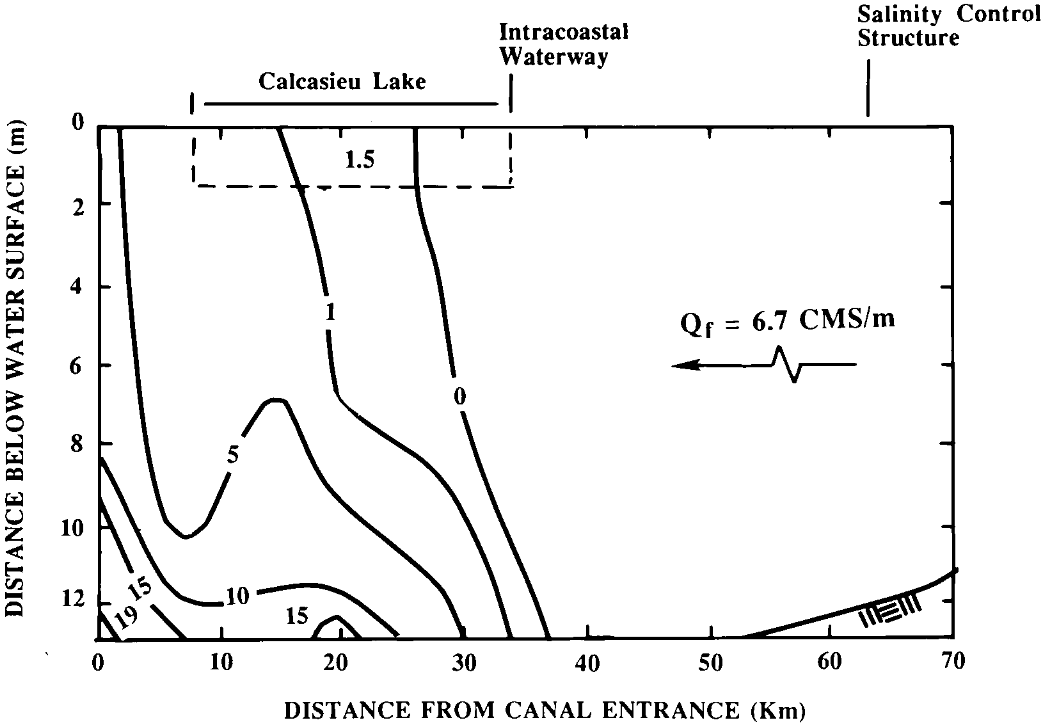


Figure 5-16. Model verfication for salinity distribution in the Houma Navigation Channel: (a) field measurements on October 17-18, 1986; (b) verified results from numerical model.



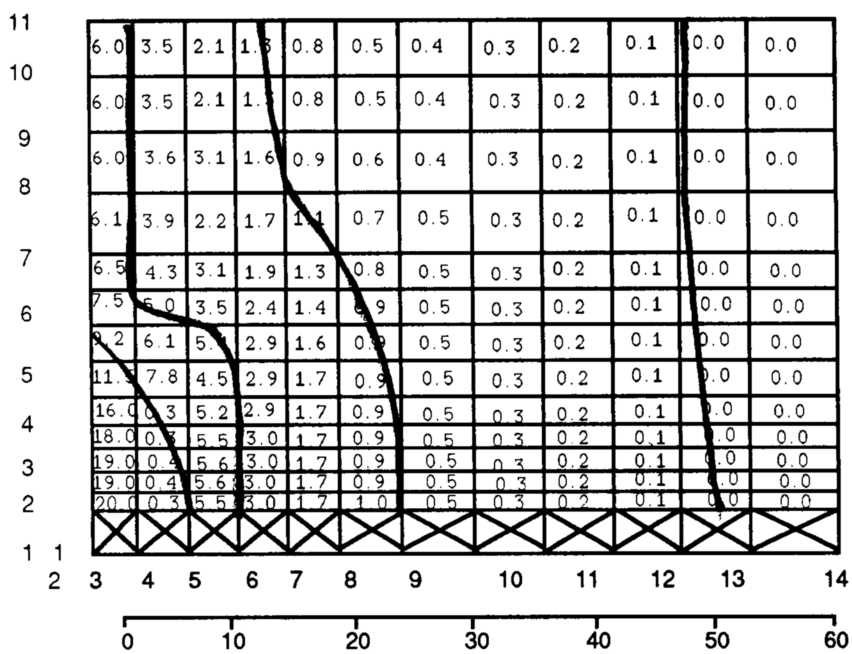


Figure 5-17. Model verification for salinity distribution in the Calcasieu Ship Channel: (a) field measurements on March 2-3, 1987; (b) verified results from numerical model.

interaction of tidal amplitude, freshwater discharge, and density gradient caused by variations in salinity were marginally satisfactory.

Only one field trip was conducted for the site of Bayou Petit Caillou site; therefore, the model has not been field verified for this site.

Model Simulation

The numerical model, once field calibrated and verified, can be used: (1) to better understand the dynamics of saltwater intrusion; (2) to estimate the length of salt water front upon the changes in channel depth, river inflow, tidal amplitude, and wind velocity; and, (3) to provide a partial answer to three questions:

(1) Is there a difference in salinity distribution in various types of channels, i.e., man-made channels versus natural bayous with various dimensions (depth, width, and length)?

(2) What are the magnitude and relative importance of the physical forcing functions, river inflow, tidal amplitude, and local wind, driving the salinity distribution in the channel?

(3) Does deepening the channel lead to migration further upstream of the saltwater front?

These questions and their answers can be simulated by the numerical model. However, in the process of computer simulation, a balance between the numerical procedure (grid size, time step, and simulation period) and physical dimension (channel depth, width, and length) needs to be well planned.

The three coastal channels in this study were used as pilot sites to demonstrate the capability of computer simulation (Wang, 1984; 1987). It should be noted that both the Houma Navigation Channel and the Calcasieu Ship Channel have been field calibrated and verified, while the Bayou Petit Caillou site has only been field calibrated.

The answer to the first question can be derived directly from our field measurements. The separate field surveys conducted for the three sites can be used for comparative purposes. During these sampling periods, river inflows were Bayou Petit Caillou, 0.5; Houma Navigation Channel, 0.5; and Calcasieu Ship Channel, 0.1 cms/m (or 25, 50, and 20 m^3/sec) ; their tidal amplitudes were 0.1, 0.1, and 0.3 meters, respectively; and, the local winds were relatively calm.

Salinity distributions are shown in Figure 5-18 for the above described physical and environmental conditions. The saltwater fronts, the 5 ppt locus, are discerned to be 20, 45, and 75 km from their respective channel entrance (Gulf side). These results (Figures 5-18a, 5-18b, and 5-18c indicate that there is a difference in the lengths of saltwater front and the patterns of salinity distribution for various types of channels.

To answer the second question, simulations were conducted to account for the effects of wind on salinity distribution in comparison with the no-wind condition encountered during the field sampling trip. The salinity in channels is transported in two ways (Prandle, 1986). In coastal Louisiana, freshwater, arising from the upstream drainage basin, flows in a one-way path through the channel to the Gulf. This path is superimposed by a two-way exchange of water which occurs as the rise and fall of tidal level and the reverse direction of local wind.

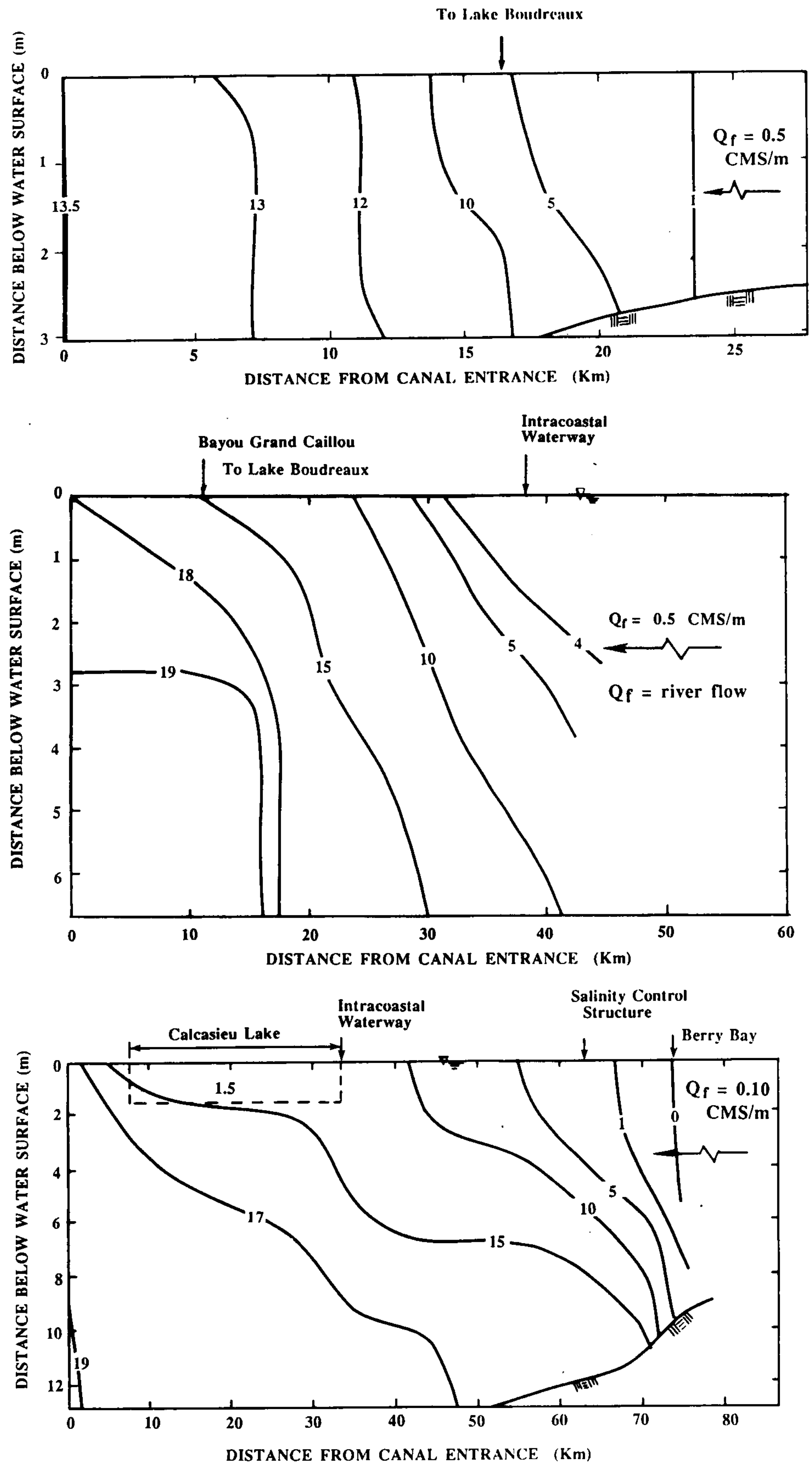


Figure 5-18. Salinity distribution in three selected channels: (a) Bayou Petit Caillou (h = 3.0 m, w = 50 m); (b) Houma Navigation Channel (h = 6.6 m, w = 100 m); (c) Calcasieu Ship Channel (h = 12.5 m, w = 200 m).

A series of computer simulations were conducted to assess the effects of wind on salinity distribution in the Houma Navigation Channel. A wind speed of 3 m/sec from the north is superimposed onto the river discharge and the tidal amplitude as measured in the field on September 20-21, 1986. The longitudinal wind stress is taken as positive toward the Gulf side (south). In a subsequent simulation, southerly wind with the same speed, 3 m/sec, is added to the same river and tidal forcings. Figure 5-19 shows the results of simulations with wind and no wind conditions. The simulated results suggest that local wind provides a more pronounced upstream channel flow in the upper water column in response to the pressure gradient set up by the wind, and that the wind direction exerts some differences in the distribution.

Similarly, Bayou Petit Caillou was tested for the effect of wind on the patterns of salinity distribution. A local wind of 3 m/sec from the north and south is added to the field data obtained on October 19, 1986, while all other variables are kept constant. The simulated results are displayed in Figure 5-20.

The issue related to deepening the shipping channel to allow larger and wider vessels to navigate the river has been raised. Computer simulations can offer partial solutions to this issue. A series of computer runs were conducted to estimate the extent of varying the channel depth and its influence on the patterns of salinity distribution in the channel.

The depth of Houma Navigation channel is increased from the original 6.6 to 13.2 m (twice the present depth), while all other variables remain the same as field measurements. Figure 5-21 demonstrates the effect of channel depth on salinity distribution in the channel. The simulated results indicate that deepening the channel has significantly changed the patterns of salinity distribution and that the saltwater front has intruded further upstream. The 5 ppt isohaline shifts from 45 km to 80 km.

Similarly, the channel depth of Bayou Petit Caillou is increased from the original 3 m to 6 m to test the effect of increasing channel depth on the salinity distribution in the channel. All other variables remain the same as field data. Figure 5-22 shows the results of simulation. It is interesting to note that the 5 ppt isohaline migrates inland from 5 km to 8 km (20 km to 23 km from the channel entrance), and the 1 ppt isohaline intrudes inland from 18 to 28 km (33 km to 43 km from the channel entrance).

Summary of Computer Simulations

The results of simulation, substantiated by field measurement, yield the following conclusions:

(1) River inflow, tidal exchange, and surface wind stress all play an interactive role on salinity distribution in channels on the Louisiana coast.

(2) The relative importance of each physical process varies with channel locations, types, and dimensions to such an extent that generalizations are inappropriate.

(3) Under similar physical forcing functions, the saltwater front intrudes further inland for larger and deeper channels than the smaller and shallower channels (Figure 5-18).

(4) The influence of wind stress on salinity distribution is much stronger in shallow channels than in deeper channels (Figure 5-19 and Figure 5-20).

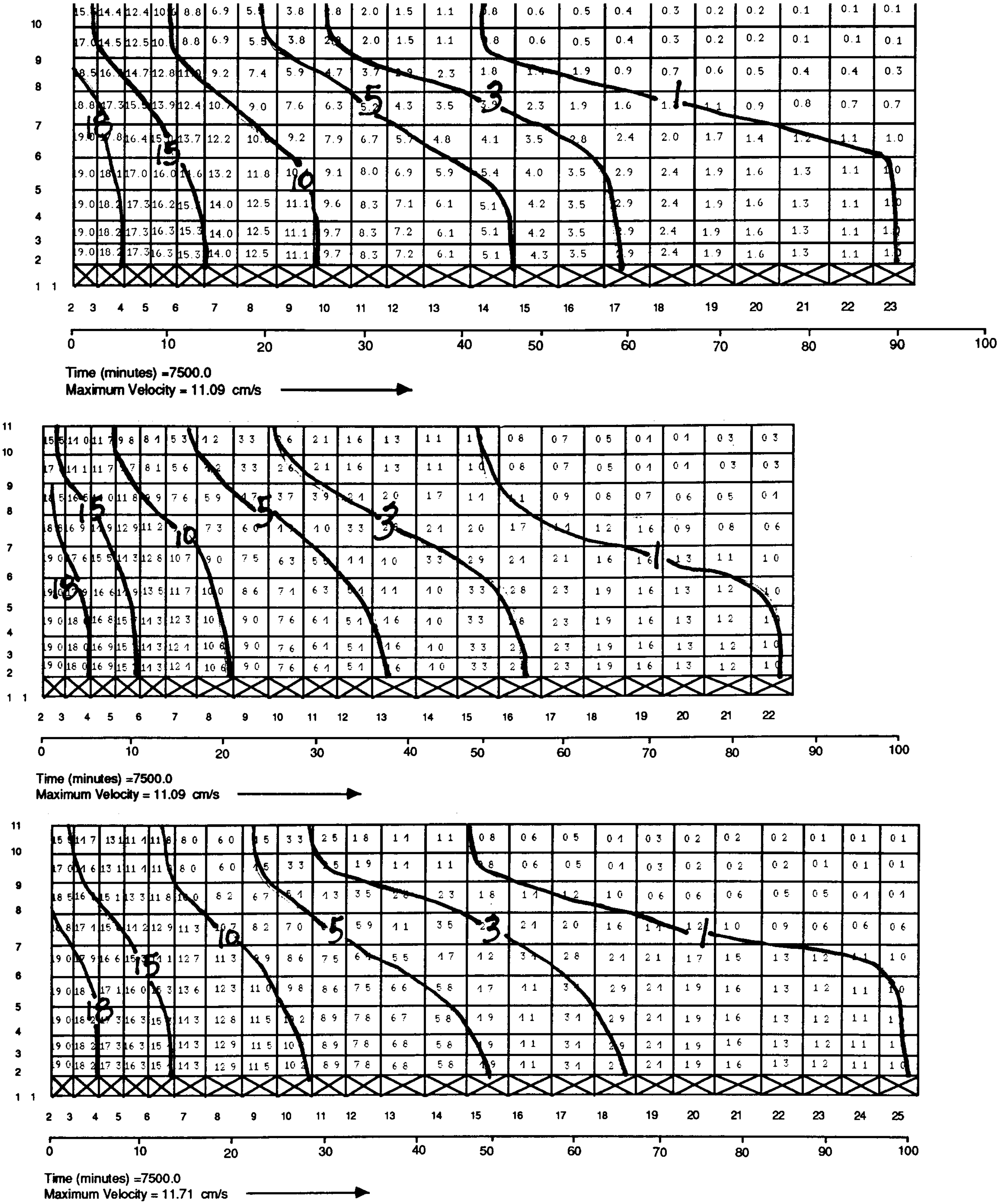


Figure 5-19. Effects of surface wind on salinity distribution in the Houma Navigation Channel; (a) no wind; (b) northerly wind = +m/sec; (c) southerly wind = -3 m/sec.

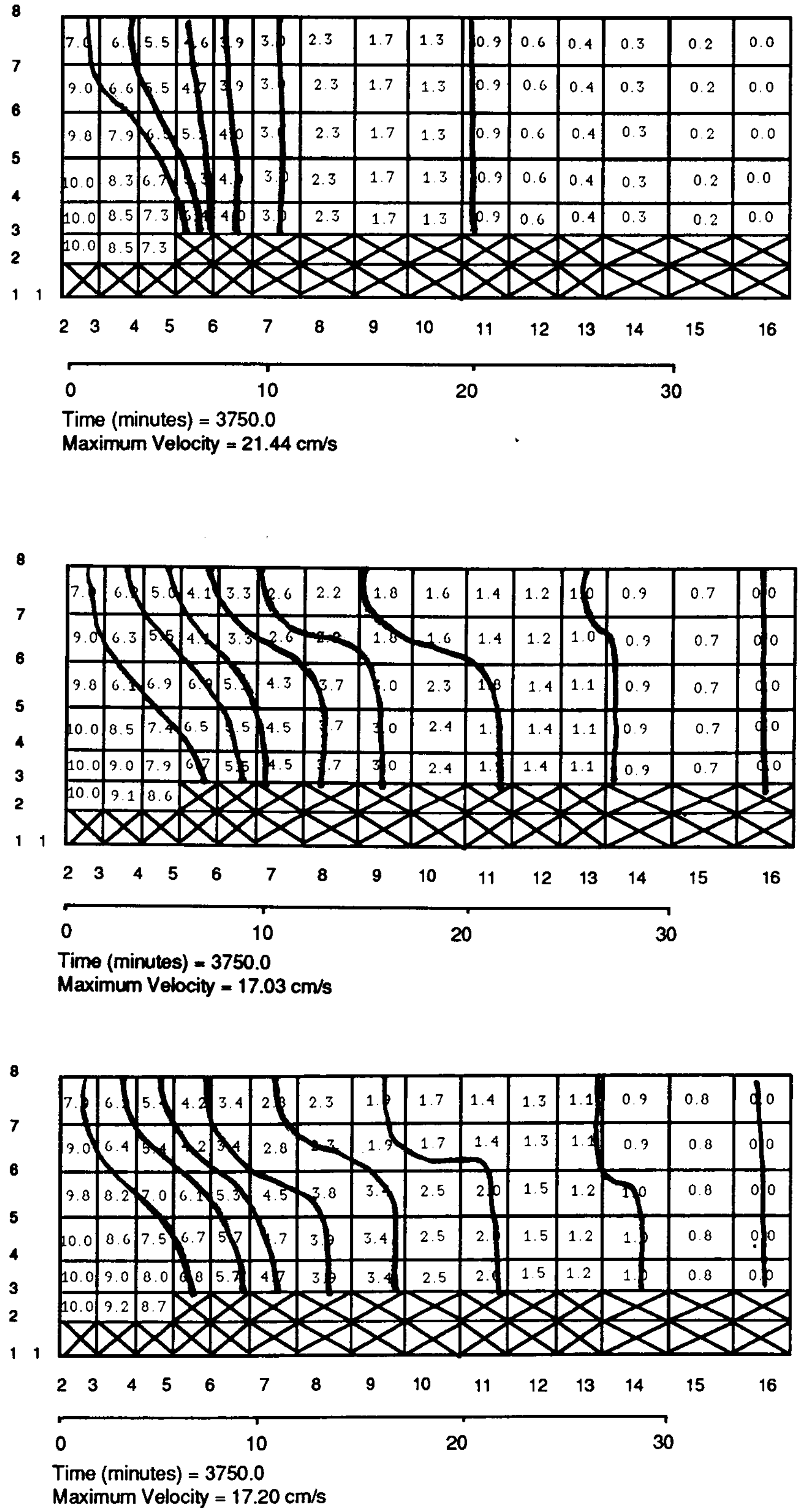
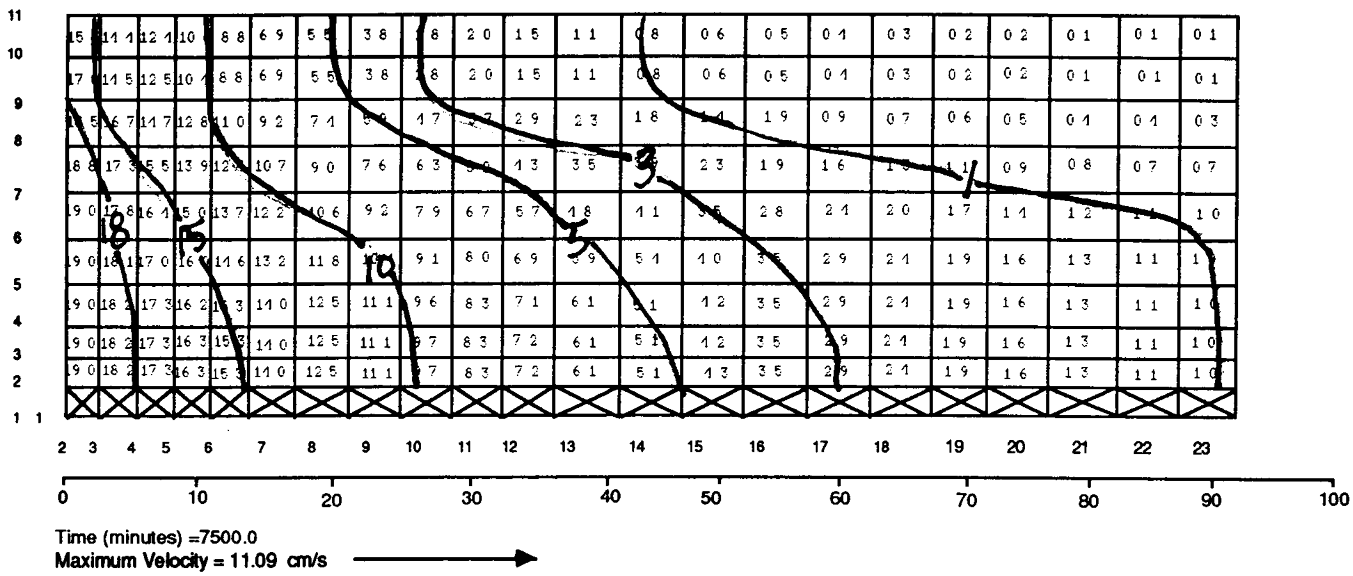


Figure 5-20. Effects of surface wind on salinity distribution in Bayou Petit Caillou: (a) no wind; (b) northerly wind = +3 m/sec; (c) southerly wind = -3 m/sec).



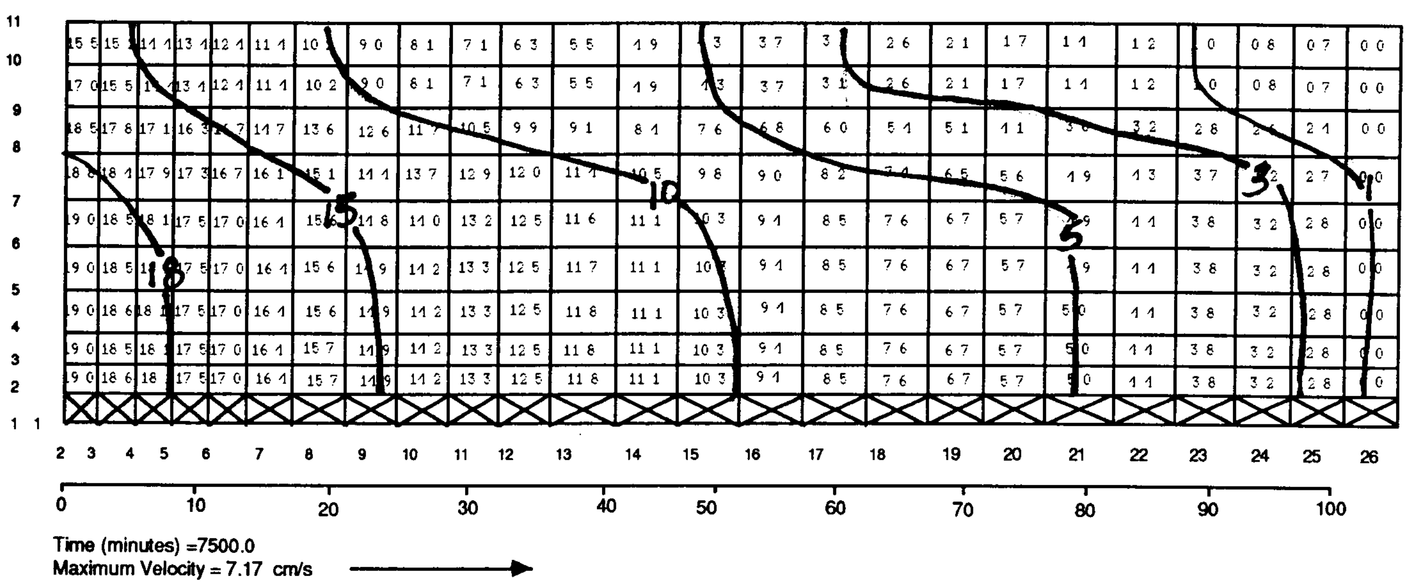


Figure 5-21. Effects of channel depth on salinity distribution in Houma Navigation Channel: (a) channel depth (h = 6.6. m); (b) channel depth (h = 13.2) meters.

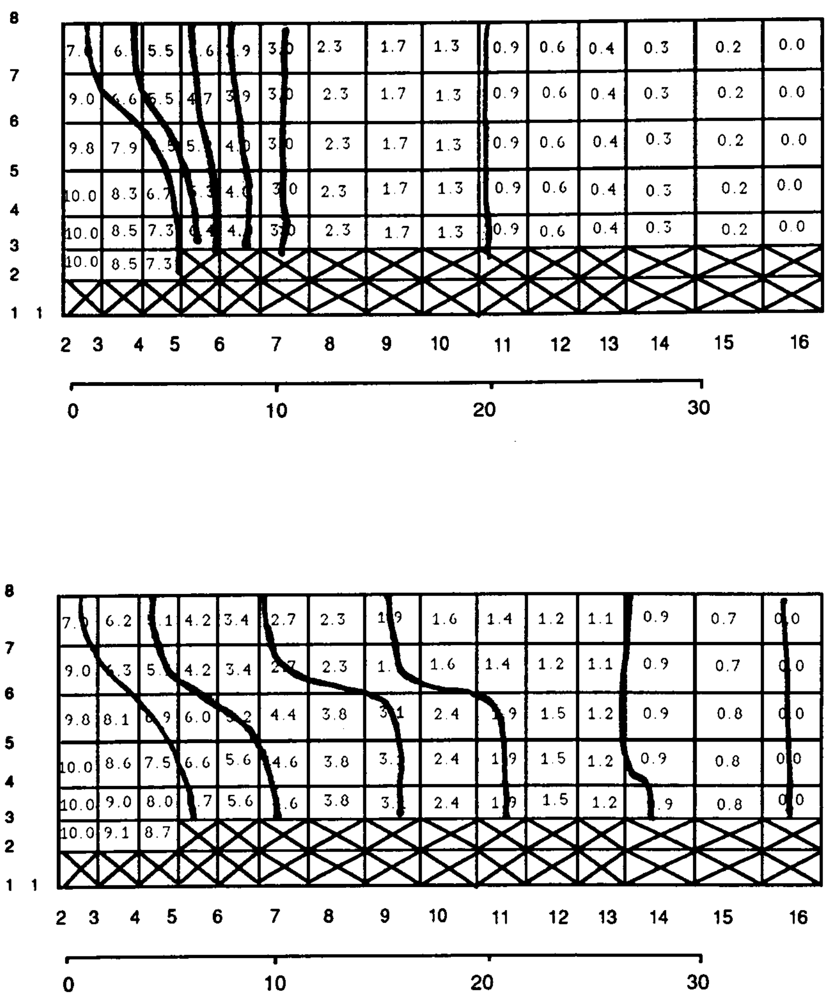


Figure 5-22. Effects of channel depth on salinity distribution in Bayou Petit Caillou: (a) channel depth (h = 3.0 m); (b) channel depth (h = 6.0 m).

(5) To a large extent, deepening the channel changes the patterns of salinity distribution and the degree of saltwater intrusion. By increasing the Houma Navigation Channel depth to 13.2 m, the 5 ppt isohaline migrates from 45 km to 80 km (Figure 5-21). By doubling the Bayou Petit Caillou depth to 6 m, the 5 ppt isohaline intrudes from 20 km to 23 km from the Gulf side (Figure 5-22), compared with the Houma Navigation Channel's current depth of 6.5 m, the 5 ppt locus is about 45 km (Figure 5-22a); and,

(6) These simulated results reveal the nonlinear and complicated nature of saltwater intrusion problems in coastal Louisiana.

The results of field measurements and computer simulations presented in this study are directed to the low to moderate flow periods from late fall to early spring in coastal Louisiana. During this period, the tides are small, amplitudes vary from 0.1 to 0.3 m. The tidal-induced currents are on the order of 1 to 2 cm/sec and are much less than channel velocities that are on the order of 10 to 30 cm/sec. The freshwater inflows, though, are in the range of 20 to 50 m^3/sec, and are important indicators of the presence of the gravitational circulation.

The numerical model developed in this task can be used as a management tool to predict quantitatively the flow stratification and salinity intrusion in coastal channels. However, many research aspects need to be done. In particular, the relative importance of each physical force in the degree of saltwater intrusion in channels needs to be evaluated. Also, the sensitivity of functional forms used for estimating the eddy viscosity and diffusion coefficients should be carried out. These efforts will enhance the models predictive capabilities as management tools which are a major subject for further research.

Chapter 6

LONG–TERM SALINITY TRENDS IN LOUISIANA ESTUARIES

by

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Classical estuarine circulation theory (Dyer, 1973), laboratory experiments, and theoretical models (Keulegan, 1966) all suggest saltwater intrusion into an estuary is enhanced by deepening the channel. Indeed, the modeling and field effort described in Chapter 5 supports these conclusions. Local lore attributes marsh degradation and land loss to enhanced saltwater intrusion caused by deepening of existing channels and dredging of canals through the marsh. This explanation for coastal land loss has been reinforced by numerous recent articles in the newspapers of southern Louisiana. The goal of our study was to analyze existing records of salinity from the Louisiana coastal zone to determine whether or not they support the notion that significant long-term trends in the estuarine salinity regime have occurred.

These data were analyzed to determine the long-term (30 to 40 years), seasonally adjusted trends. We were primarily concerned with the salinity regime as it relates to both natural and man-induced factors. Thus, the task had two major objectives: to determine the secular trends(if any) in the salinity regime and to identify the factors that may be controlling these changes. We were particularly interested to see if the existing salinity records indicated the occurrence of changes of sufficient magnitude to impact the vegetation.

To meet these objectives, we addressed the following questions:

(1) Has there been a statistically significant trend in the mean salinities at stations within the coastal zone?

(2) Has there been a statistically significant trend in the variance of the mean salinities at these stations?

(3) Has there been a statistically significant trend in the maximum salinities at these stations?

If the answer to any of the above is yes, two more questions arise:

(4) What is the magnitude of the trend?

(5) Do these salinity trends appear to be the result of natural variability?

Two historical data sets exist from the estuarine waters of south Louisiana. These have been collected by the Louisiana Department of Wildlife and Fisheries (LDWF) and by the United States Army Corps of Engineers (COE). The sampling sites for the records chosen are listed in Table 6-1. The collection of these data sets was not designed with the present study in mind. Station locations, sampling rates, and sampling durations are in no way optimal for these purposes. Nevertheless, the data form a geophysical data set whose length is rarely encountered. They offer the potential for gleaning information where only inferences could otherwise be made.

Table 6-1. List of stations used in the analysis. The major water body, the station number and location description (5 and 3 digit numbers refer to COE and LDWF data stations, respectively) are indicated. Summary statistics (mean, standard deviation and number of observations) for the period of record are also presented.

Major Water Body	Salinity Stations	Mean ppt	SD ppt	N Days
Lake Pontchartrain	102 - Chef Menteur	3.89	2.73	2157
	118 - The Rigolets	6.32	2.80	954
	85683 - North Shore	4.01	2.41	976
	85650 - Little Woods	3.95	2.35	10830
	85700 - The Rigolets	4.84	3.57	6378
	85750 - Chef Menteur	5.38	2.98	8189
Lake Borgne	117 - Grand Pass	16.25	5.85	856
Breton Sound	221 - Bay Gardene	13.61	5.07	2023
	251 - Long Bay	11.29	5.72	991
	252 - California Bay	17.14	5.85	806
	253 - Sable Island	19.29	6.35	788
	76042 - GIWW Paris Rd.	9.90	4.66	242[a]
	85820 - MRGO @ Navig. Light 101	15.28	6.97	275[a]
Bird Foot Delta	01500 - The Jump	0.42	1.25	1408
	01420 - Port Sulphur	0.17	0.38	14862
Barataria Bay	315 - Marine Lab @ Grand Terre	20.90	5.71	7664
	317 - St. Mary's Point	12.90	6.36	2984
	82203 - Larose	0.56	1.19	7951
	82750 - Barataria	1.93	1.58	168[a]
	82300 - Galliano	1.72	3.17	6527
	82350 - Leeville	15.50	5.45	7621
Terrebonne Bay	416 - Cocodrie	9.44	5.49	3370
	76403 - Bayou Terrebonne @ Bourg	0.62	1.60	5854
	76320 - GIWW @ Houma	0.34	1.04	10426
	76323 - Grand Caillou @ Dulac	1.20	2.79	11117
	76343 - Houma Nav. Canal @ Crozier	0.55	1.76	5883
Terrebonne Marshes	518 - Caillou lake Camp	10.76	5.14	2763
	03780 - Atchafalaya R. @ Morgan City	0.07	0.05	6134
	52800 - Bayou.Boeuf @ Amelia	0.14	0.11	1135[a]
	64800 - Bayou Teche @ Patterson	0.11	0.09	1467[a]
Atchafalaya–Vermilion Bays	619 - Cypremort Point.	3.83	2.41	2046
	620 - Southwest Pass	6.07	4.07	701
	03720 - Wax Lake Outlet	0.06	0.04	5561
	64450 - Char. Drainage Canal @ Baldwin	0.24	0.56	9772
	64380 - Bayou Teche @ Charenton	0.17	0.25	7970
	88600 - Atchafalaya R @ Eugene Island	4.93	7.16	3119
	88850 - Cypremort Point.	4.90	3.46	7027
Calcasieu, Sabine, White Lakes	701 - Rockefeller S.	13.55	6.83	1490
	702 - Rockefeller N.	11.74	6.79	1283
	719 - Cameron	15.89	5.86	2939
	76720 - GIWW Vermillion Lock East	1.73	2.28	3288
	76800 - GIWW Vermillion Lock West	1.32	1.96	5874
	76690 - Schooner Bay	1.33	1.04	647[a]
	70675 - Mermentau River	1.35	2.86	9367

[a] Stations with weekly instead of daily samples

The LDWF data set has been obtained principally from the open water of the estuaries. Over the record length, the data have been collected using three different methodologies, based on conductivity measurements. All three methodologies allow estimation of a daily mean salinity. Using the Kolmogorov-Smirnov test (Siegel, 1956) for comparison, the probability distributions of the daily mean data from the earliest technique appear to be significantly different from those collected using the two later techniques. On the other hand, we could find no statistically significant difference between the monthly mean salinities or the monthly variances about the monthly means of the data collected using the different techniques. It is not clear whether this results from reduced degrees of freedom in the monthly averaged data sets or error cancellation. Nevertheless, estimates of trend based on monthly mean data were made using the complete data set (referred to as data set L1). Trends of monthly maximum salinities and persistence estimates based on daily mean data were made using only the data collected with the latter two techniques (referred to as data set L2). The availability of daily means from this data set makes it extremely amenable to analysis for trends. Unfortunately, the records are relatively short, even when the earliest data are included, and natural variability may mask any anthropogenically induced trend. Some of the most recent data have not yet been processed and quality controlled by the LDWF, and, thus, were not available to us. As longer records become available, more powerful tests may be run. Finally, it should be noted that the data are principally collected from near-surface waters. No effort is made to estimate the vertical stratification of the water column.

The COE data set (referred to as data set C) is collected from the navigable waterways of south Louisiana. Many of the measurement sites are fresh for extended periods of the year and only exhibit measurable salinity during selected periods, i.e., the data contain spikes. The salinity data are sampled once a day, nominally at 0800 hours, thus introducing tidal aliasing into the data sets.* Occasionally, data are collected at more than one depth. When this occurs, we averaged the measurements. Examination of the samples from multiple depths suggests that the water columns are generally not highly stratified at these sites. The complete data set is very large. We analyzed the data from the longest time-series available (up to 49 years) to obtain the greatest reliability and to reduce the influence of climatic perturbations.

Time-lines for the data sets used in the subsequent analyses are in Figure 6-1. Since the data sets were sampled at different time intervals, the LDWF data were averaged to produce daily values. A given day of data was assumed to be missing if less than 18 hours of measurements from that day was available. Monthly values were produced from both data sets by averaging daily values. A given month of data was assumed to be missing if less than 20 days of data from that month was available. We chose to compare monthly data rather than daily data because, while data set L2 involved hourly values and, thus, could have been analyzed to remove the tidal signal, data set C contained a single sample per day which aliased the tidal signal into the data. The impact of this aliasing effect was reduced by monthly averaging. Examples of the data available for analysis are presented in Figures 6-2 and 6-3. Data plots of the monthly mean salinities and the variance about the monthly mean, for all stations, are presented in Appendix D. Occasionally two stations, one maintained by the COE and one by the LDWF, were operating in close proximity to each other for a year or longer. When such a situation occurred, the two data sets of daily values were compared. The two data sets were highly correlated. Considering the spatial

* When a tidal signal is samples less than twice each cycle, it appears to have a longer period than the tidal period. Thus variance associated with the tide seems to occur at at longer time scale. For example, if the tidal signal is sampled only at the crest of the tide, an apparent mean is added to the measurements. This effect, resulting from undersampling the process, is called tidal aliasing.

variabilities known to occur in coastal waters, the results were interpreted to indicate that the two data sets were comparable.

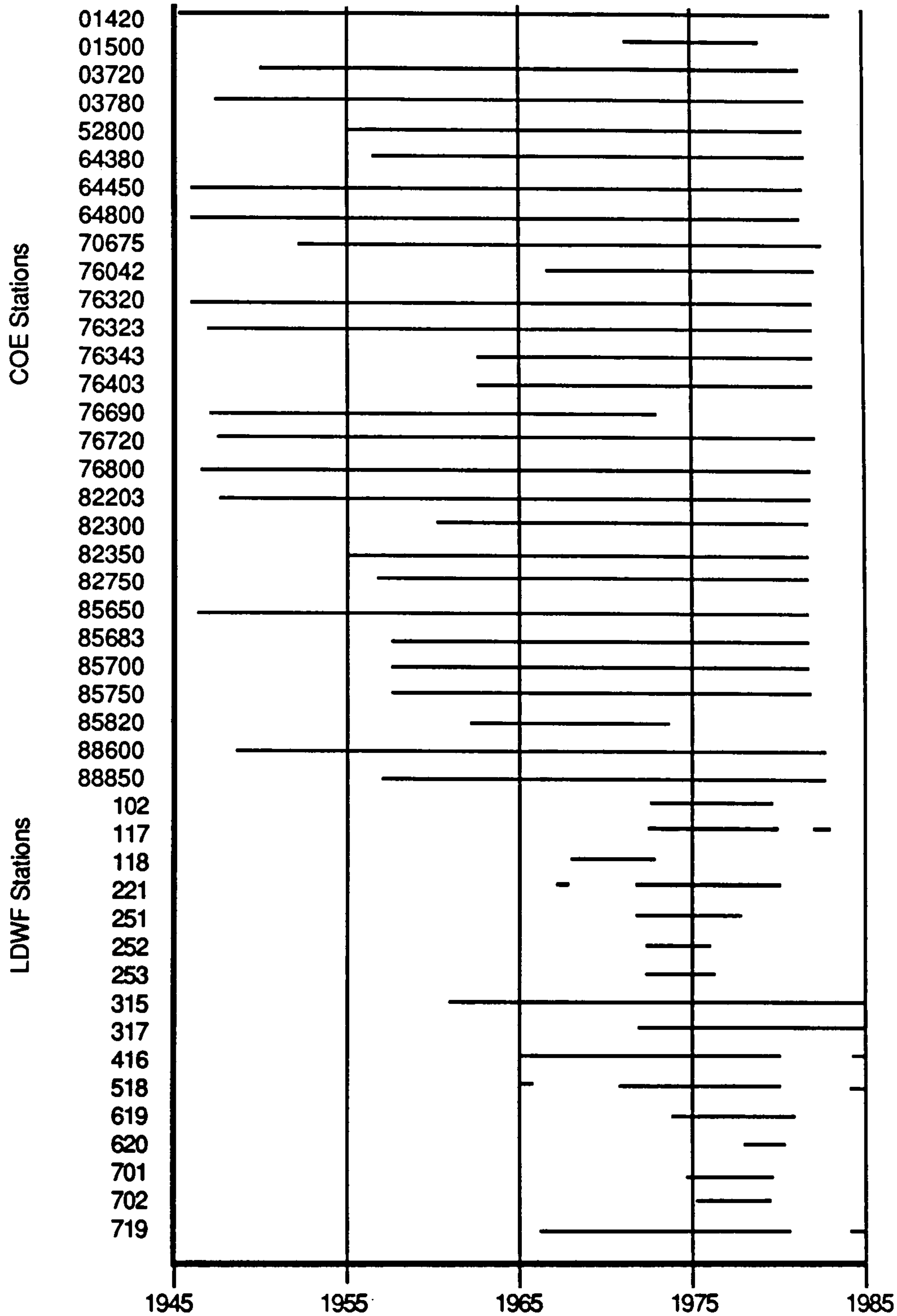

Figure 6-1. Time line showing the time period of data availability for the salinity stations used in the analysis.

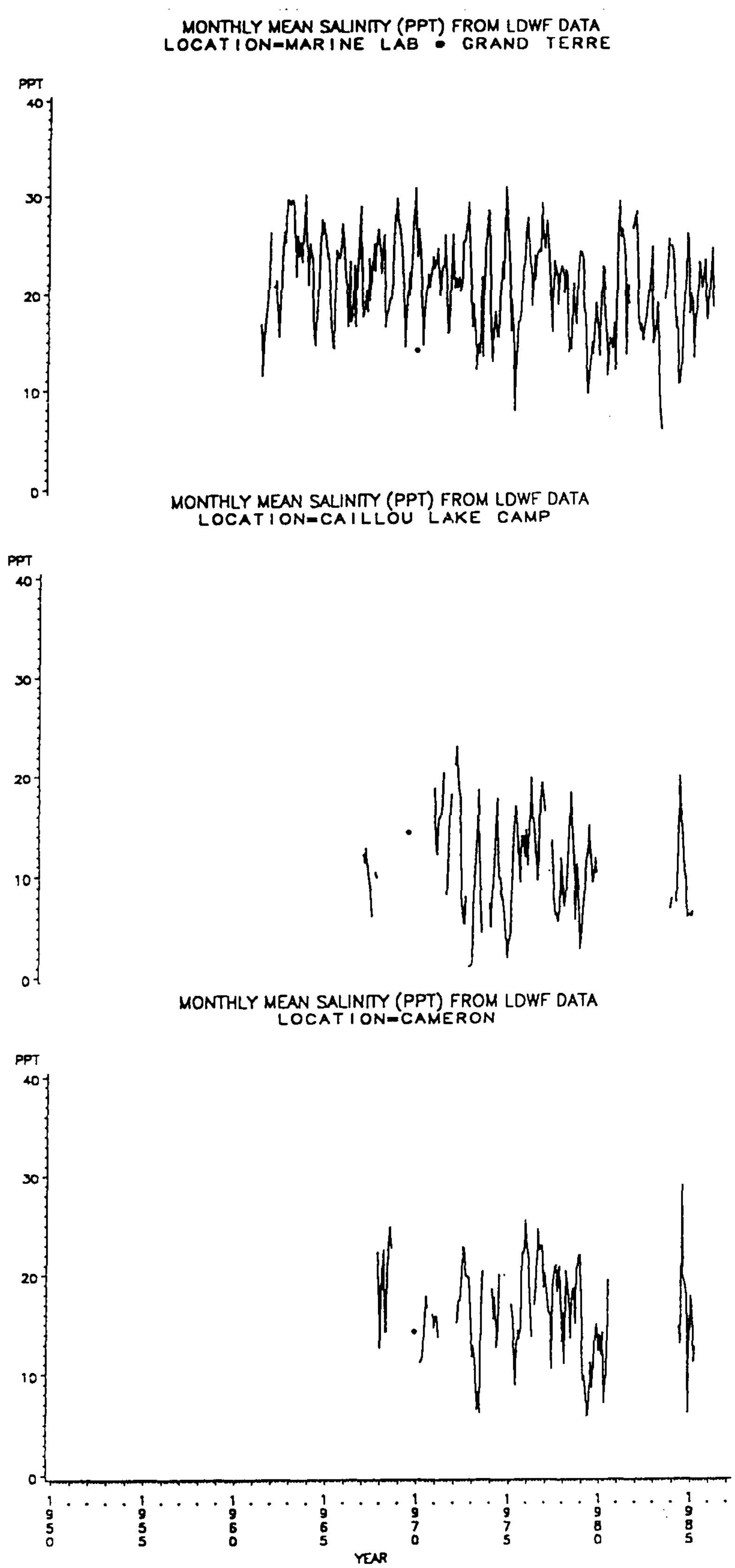

Figure 6-2. Examples of monthly mean data plots from LDWF salinity stations at (top to bottom) the marine lab at Grand Terre, the Caillou Lake Camp,and Cameron.

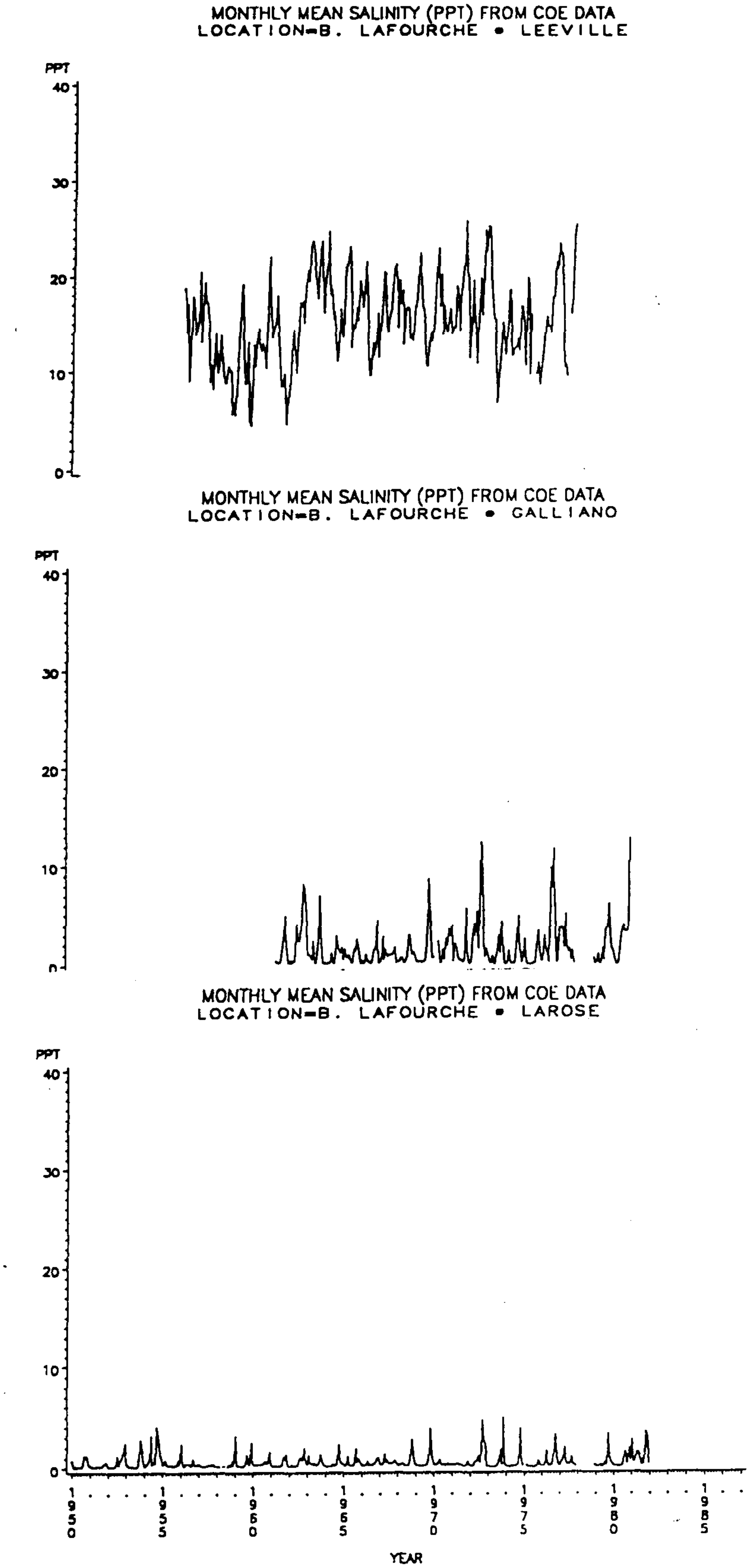

Figure 6-3. Examples of monthly mean data plots from COE salinity stations at (top to bottom) Bayou Lafourche at Leeville, Bayou Lafourche at Galliano, and Bayou Lafourche at Larose.

Analysis Procedures

We began an investigation of secular trends by fitting a linear model to the data sets with time and an annual harmonic (to remove the seasonal serial correlation effects) as the independent variables in the model. We then tested the slope parameter of the model for statistical significance (Neter and Wasserman, 1974). In this case and in all subsequent tests, we used an alpha level of 90% as representative of statistical significance. When a trend was significant at this minimum level, the actual alpha level resulting from the test is presented. The procedures involved are well understood and numerous computer codes exist to perform the analysis. We selected a procedure supported by Statistical Analysis System called GLM (general linear models; SAS, 1985a, b). The statistical interpretation of the results involves certain assumptions about the data. The residuals, after the model has been subtracted from the data, are assumed to be normally distributed with homogeneous variance. We tested the residuals for normality, and the results generally indicated that the residuals were non-normally distributed. The test procedures used in GLM are, however, rather robust with respect to the assumption of normally-distributed residuals (B. Moser, LSU Department of Experimental Statistics, personal communication). Therefore, rather than reject the analyses out-of-hand, we visually compared the probability distribution function of the residuals to that of a normal distribution and also visually inspected the raw residuals. If the two distributions did not widely diverge and neither the distributions nor the raw residuals indicated an undue influence of outliers, we accepted the test results. If, though, there was indication of a secular trend in the variance of the residuals or outliers were prevalent, the results of the test were considered suspect. This procedure was run on the monthly mean salinity time-series, on the time-series of the monthly variance about the monthly mean, and on the monthly standard deviations in the hope that the square root transformation might bring the data more nearly in line with the assumptions of the model. Finally, we ran the same analyses on the maximum daily mean salinities observed during a given month. Since this last test involved only daily values, the analysis was run on data sets L2 and C, but not on L1. The results were not terribly satisfying. Many of the test results had to be discarded because the data did not satisfy the assumptions of the model. Sorting the data by month before fitting a linear model, e.g., fitting a linear model to all January data from a given station followed by a separate fit to all February data, did not significantly improve the results or our understanding of the temporal variability of the system. Much of this difficulty arose because of the non-normal distribution of the data, especially data set C. This was particularly disconcerting as these latter data were from the upper reaches of the estuarine systems, where plants might be expected to be most sensitive to variations in the salinity regime.

We, therefore, began a search for a non-parametric test statistic which would answer the questions we had initially posed. We chose the seasonal Kendall-Tau (Hirsch et al., 1982). This procedure tests for the presence of a statistically significant, monotonic trend in the data. It is important to note that the trend need not be linear. A Monte Carlo analysis of this procedure has been presented by Hirsch et al. (1982). It should be noted that this test was initially designed to study data containing spikes similar to those in data set C. This test was applied to the monthly mean salinity data sets, the monthly variances about the monthly means, and the monthly maximum salinities (the monthly standard deviations give identical answers to those derived for the variances). This test does not estimate the magnitude of the trend, only that a trend exists and its sign. There is a non-parametric estimator of the linear portion of an existing trend, the seasonal Kendall slope estimator. Once again, this estimator has been extensively described and Monte Carlo studies of its performance characteristics presented by Hirsch et al (1982). This technique was used to estimate linear portions of the trends in the data sets mentioned above. Note that catastrophic, monotonic changes in salinity regimes will appear as a trend in a time series.

The amplitude of the linear portion of this trend will diminish as the record increases in length. Specific tests to identify and explain catastrophic changes *per se* were not performed on the available data sets.

Persistence statistics from the daily data were estimated for each station from data sets L2 and C, recognizing that the LDWF data represented daily averages, while the COE data represented daily samples. Salinity levels above which persistent events were measured were set at salinity increments of 5ppt. Because of the quality of the data sets, the possibility of long-term natural variability in the data, and the infrequent occurrence of long-duration, high-salinity events, we only estimated the persistence statistics for entire data records. No attempt was made to determine trends in the persistence statistics. Even with this restriction in the analysis, the resulting statistics are less reliable than anticipated because the time series of daily samples are often broken. When missing data points occur, it is not clear how to interpret the data. If breaks are short, linear interpolation might be acceptable, but when longer breaks occur, interpolation can no longer be used. We tried setting the missing data higher than any reasonable value to extend runs of high salinity, as well as setting missing values to zero to be extremely conservative and not overestimate the duration of a high salinity event. By comparing the two results, we begin to determine bounds on the extreme duration, high salinity events. Although the results of these analyses are interesting in their own right, they do not answer questions concerning secular trends, and their presentation is restricted to Appendix D.

Results

When the linear trend plus seasonal cycle was fit to the monthly mean salinities from data set L1 using GLM, the trend term appeared to be significant in five cases (see Table 6-2). In all cases, the trend was negative, i.e., salinity appeared to be decreasing with time. The total salinity change predicted over the record length ranged from 2 to 6 ppt. One result, though, was discounted because the record at that station was very short, less than five years in duration. Two others were discounted because the residuals from the fit exhibited outliers. Thus, only the predicted trends at Grand Terre and Cocodrie (stations 315 and 416) appeared to be reliable. A similar analysis of the variances about the monthly means detected only four stations with significant trends (Table 6-2). Two results were rejected because the records were less than five years in duration and one because of the strong influence of outliers. A square root transformation reduced the influence of these outliers and resulted in two stations showing reliable trends in the standard deviation of salinity about the monthly mean: the trend was positive at Cameron (station 719) and negative at Grand Terre (station 315).

Finally, when the monthly maxima from data set L2 were fit with the model, four stations showed statistically significant trends (Table 6-2). Two of these records were less than five years in duration. A third was affected by the presence of outliers. Only the station at Grand Terre (station 315) showed a believable mean trend in the monthly maxima, and it was negative. When the same model was fit to the records from data set C, 14 stations showed statistically significant trends in the mean salinities (Table 6-3). One was rejected because of the small number of available data points. Ten were rejected because of the number of outliers in the data set. None of the trends in salinity variance were accepted because of the presence of outliers in the records (Table 6-3). While 15 stations suggest significant trends in the monthly salinity maxima (Table 6-4), only the Leeville station (station 82350) results appear reliable. All other stations exhibited outliers in the residuals or were of short duration.

Table 6-2. Results of the seasonally adjusted GLM on monthly means and variances from data set L1 maxima from data set L1. Stations locations are shown on Figure 6-4.

Station	Years of Record		Alpha	Trend	Change during Period of Record
		Means:		ppt/yr	ppt
117	3.0		0.004	-2.748	-8.2[a]
221	12.3		0.032	-0.384	-4.7[b]
315	26.3		0.000	-0.228	-6.2
416	17.8		0.072	-0.216	-3.8
719	17.4		0.003	-0.336	-5.8[b]
		Variances:		ppt^2/yr	ppt^2
117	3.0		0.092	-3.948	-11.8[a]
252	3.0		0.094	-14.028	-42.1[a,b]
317	12.6		0.013	-0.744	-9.4
719	17.4		0.005	+0.684	+11.9[b]
		Maxima:		ppt/yr	ppt
117	3.0		0.006	-3.036	-9.1[a]
221	12.3		0.076	-0.480	-5.9[b]
315	10.5		0.043	-0.276	-2.9
619	3.5		0.002	-1.280	-4.5

[a] Short record
[b] GLM inappropriate (outliers and/or non-normal residuals)

We anticipated that the seasonal Kendall-Tau test for trend would be more appropriate to the observed data distributions. The results of applying this test to the same data sets are presented in Tables 6-5 and 6-6 and Figures 6-4 through 6-9. It is evident that a much larger number of stations show up as exhibiting reliable and significant secular trends using this test. Data set L1 contains five stations at which the trend of the monthly means appears to be significant (Table 6-5, Figure 6-4). The station at Cocodrie, which was deemed to have a significant trend using the least squares fit, no longer appears significant. Two of the stations, which appear to show significant trends, are deemed unreliable because of their short record length. The remaining three all show a decreasing trend in the mean salinity. Two stations, St. Mary's Point (station 317) and Cocodrie (station 416), show significant decreasing trends in the variance (Table 6-5, Figure 6-5). The station at Cameron (station 719), which was identified as having a significant trend in variance using the least squares technique, no longer shows up as significant. The monthly maxima again show four stations with significant trends (Table 6-5, Figure 6-6). Three are the same as those found using the least squares model. The station at Bay Gardene (station 221) no longer appears significant, while that at Cameron (station 719) does. All trends are negative.

Table 6-3. Results of the seasonally adjusted GLM on the monthly means and variances for the COE data. Station locations are shown on Figure 6-7.

Station	Years of Record	Alpha	Trend	Change during Period of Record
	Means:		ppt/yr	ppt
01420	48.6	0.000	-0.006	-0.2[b]
03720	17.0	0.002	+0.001	+0.0[b]
03780	18.0	0.000	-0.002	-0.0[b]
64380	24.4	0.000	-0.010	-0.2[b]
64450	29.0	0.000	-0.018	-0.5[b]
70675	31.6	0.023	-0.034	-1.1[b]
76320	30.8	0.000	+0.010	+0.3[b]
76323	33.5	0.076	+0.017	+0.6[b]
76800	19.6	0.000	-0.075	-1.5[b]
82203	22.9	0.082	+0.012	+0.3[b]
82350	21.8	0.000	+0.192	+4.2
85650	31.9	0.000	+0.081	+2.6
85683	3.8	0.000	+1.761	+6.7[a]
85750	24.0	0.035	+0.043	+1.0
	Variances:		ppt^2/yr	ppt^2
01420	48.6	0.000	-0.001	-0.0[b]
03720	17.0	0.027	+0.000	+0.0[b]
03780	18.0	0.063	-0.000	-0.0[b]
64380	24.4	0.020	-0.003	-0.1[b]
64450	29.0	0.000	-0.026	-0.7[b]
76320	30.8	0.000	+0.058	+1.8[b]
76720	29.1	0.022	+0.032	+0.9[b]
82203	22.9	0.017	+0.074	+1.7[a]
82300	19.7	0.033	+0.239	+4.7[b]
82350	21.8	0.014	+0.229	+5.0[b]
85650	31.9	0.094	+0.006	+0.2[b]
85700	24.2	0.088	+0.068	+1.6[b]
88850	24.3	0.060	-0.056	-1.4[b]

a Short record
b GLM inappropriate (outliers and/or non-normal residuals)

Table 6-4. Results of the seasonally adjusted GLM on the monthly maxima for the COE data. Station locations are shown on Figure 6-7.

Station	Years of Record	Alpha	Trend	Change during Period of Record
			ppt/yr	ppt
01420	48.6	0.000	-0.011	-0.5[b]
03720	17.0	0.000	+0.005	+0.1[b]
03780	18.0	0.008	-0.005	-0.1[b]
64380	24.4	0.000	-0.028	-0.7[b]
64450	29.0	0.000	-0.055	-1.6[b]
76320	30.8	0.000	+0.078	+2.4[b]
76720	29.1	0.041	+0.054	+1.6[b]
76800	19.6	0.006	-0.084	-1.6[b]
82203	22.9	0.001	+0.078	+1.8[b]
82350	21.8	0.000	+0.207	+4.5
85650	31.9	0.000	+0.094	+3.0[b]
85683	3.8	0.000	+2.325	+8.8[a,b]
85700	24.2	0.029	-0.083	-2.0[b]
85750	24.0	0.006	+0.073	+1.7[b]
88850	24.3	0.069	-0.076	-1.8[b]

[a] Short record
[b] GLM inappropriate (outliers and/or non-normal residuals)

Table 6-5. Results of the Kendall-Tau test on monthly means, variances and maxima for the LDWF data. Data includes chart data for means and variances only. These results are also shown on Figures 6-4, 6-5, and 6-6.

Station	Years of Record		Alpha	Trend	Change during Period of Record
		Means:		ppt/yr	ppt
221	12.3		0.010	-0.344	-4.2
253	4.8		0.095	+0.870	+4.2[a]
315	26.3		0.000	-0.242	-6.4
702	4.7		0.097	-0.848	-4.0[a]
719	17.4		0.006	-0.449	-7.8
		Variances:		ppt^2/yr	ppt^2
317	12.6		0.020	-0.541	-6.8
416	17.8		0.063	-0.285	-5.1
		Maxima:		ppt/yr	ppt
117	3.0		0.044	-3.702	-11.1[a]
315	10.5		0.005	-0.333	-3.5
619	3.5		0.030	-1.427	-5.0[a]
719	6.4		0.003	-1.063	-6.8

[a] Short record

Table 6-6. Results of the Kendall-Tau test on monthly means and variances for the COE monthly data. These results are also shown on Figures 6-7 and 6-8.

Station		Alpha	Trend	Change during Period of Record
	Means:		ppt/yr	ppt
03720		0.012	+0.000	+0.0
03780		0.000	-0.000	+0.0
64380		0.000	-0.009	-0.2
64450		0.000	-0.008	-0.3
70675		0.000	-0.010	-0.3
76320		0.000	+0.002	+0.1
76343		0.000	-0.012	-0.2
76403		0.003	-0.009	-0.2
76720		0.025	+0.039	+1.3
76800		0.000	-0.024	-0.5
82203		0.093	-0.003	-0.1
82350		0.000	+0.156	+3.4
85650		0.000	+0.086	+2.8
85683		0.002	+1.992	+7.8[a]
85700		0.061	+0.039	+0.9
88600		0.068	+0.131	+1.2
88850		0.012	-0.067	-1.6
	Variances:		ppt^2/yr	ppt^2
01420		0.000	-0.00	-0.00
03720		0.000	0.00	+0.00
64380		0.000	-0.0002	-0.01
64450		0.000	-0.0002	-0.01
70675		0.067	-0.0001	-0.00
76320		0.000	0.0000	+0.00
76343		0.002	-0.0000	-0.00
76403		0.056	-0.0002	-0.00
76720		0.000	0.0230	+0.75
76800		0.022	-0.0018	-0.04
82300		0.021	0.0167	+0.33
82350		0.024	0.1314	+2.88
85650		0.000	0.0057	+0.18
85750		0.000	0.0170	+0.41
88850		0.007	-0.0148	-0.36

[a] Short record

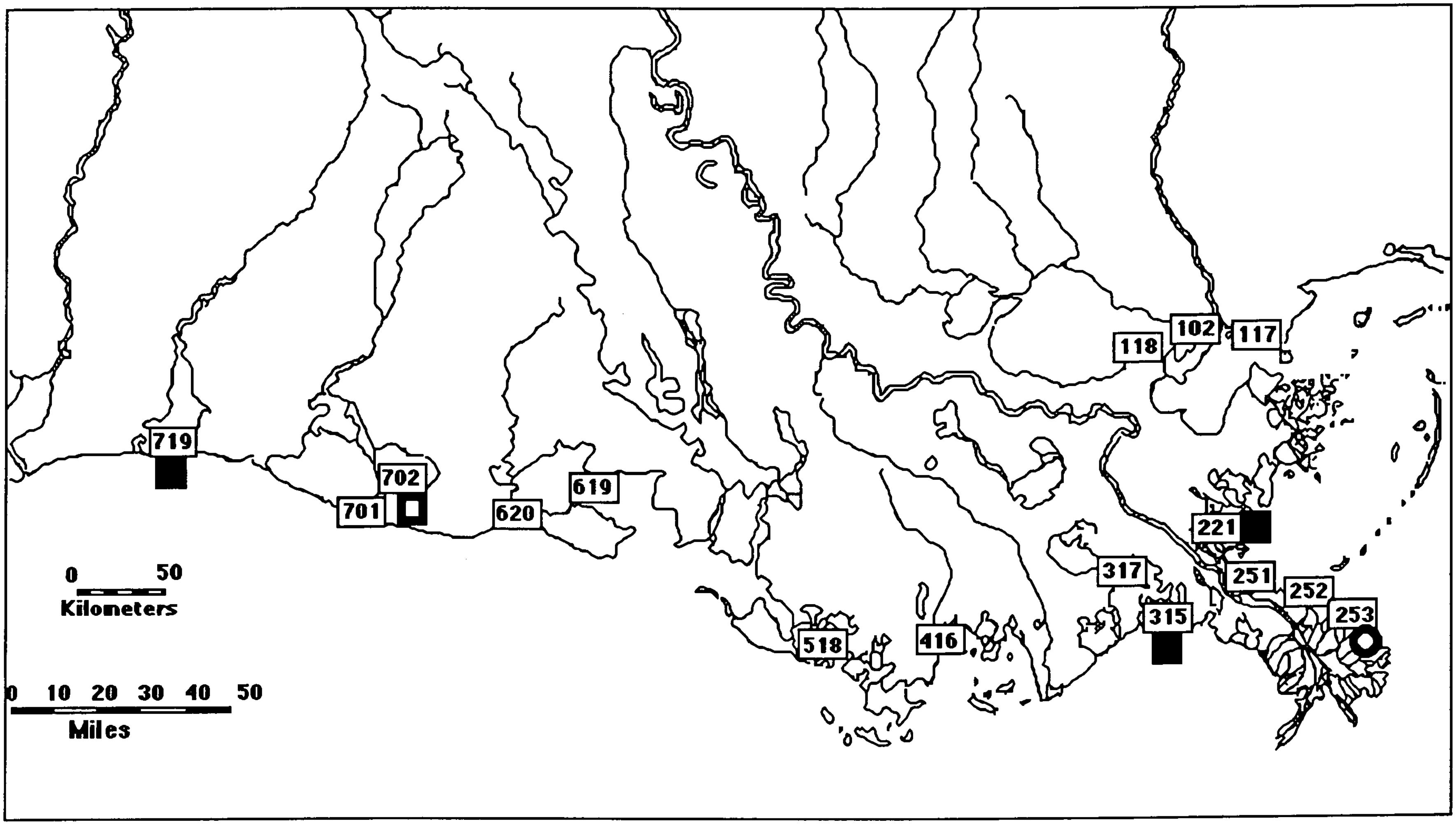

Figure 6-4. Map of the Louisiana coastal zone showing results of the Kendall-Tau test on the LDWF mean monthly salinities. Squares indicate statistically significant negative trends (solid = 95% level, open = 90% level) and circles indicate statistically significant positive trends (solid = 95% level, open = 90% level).

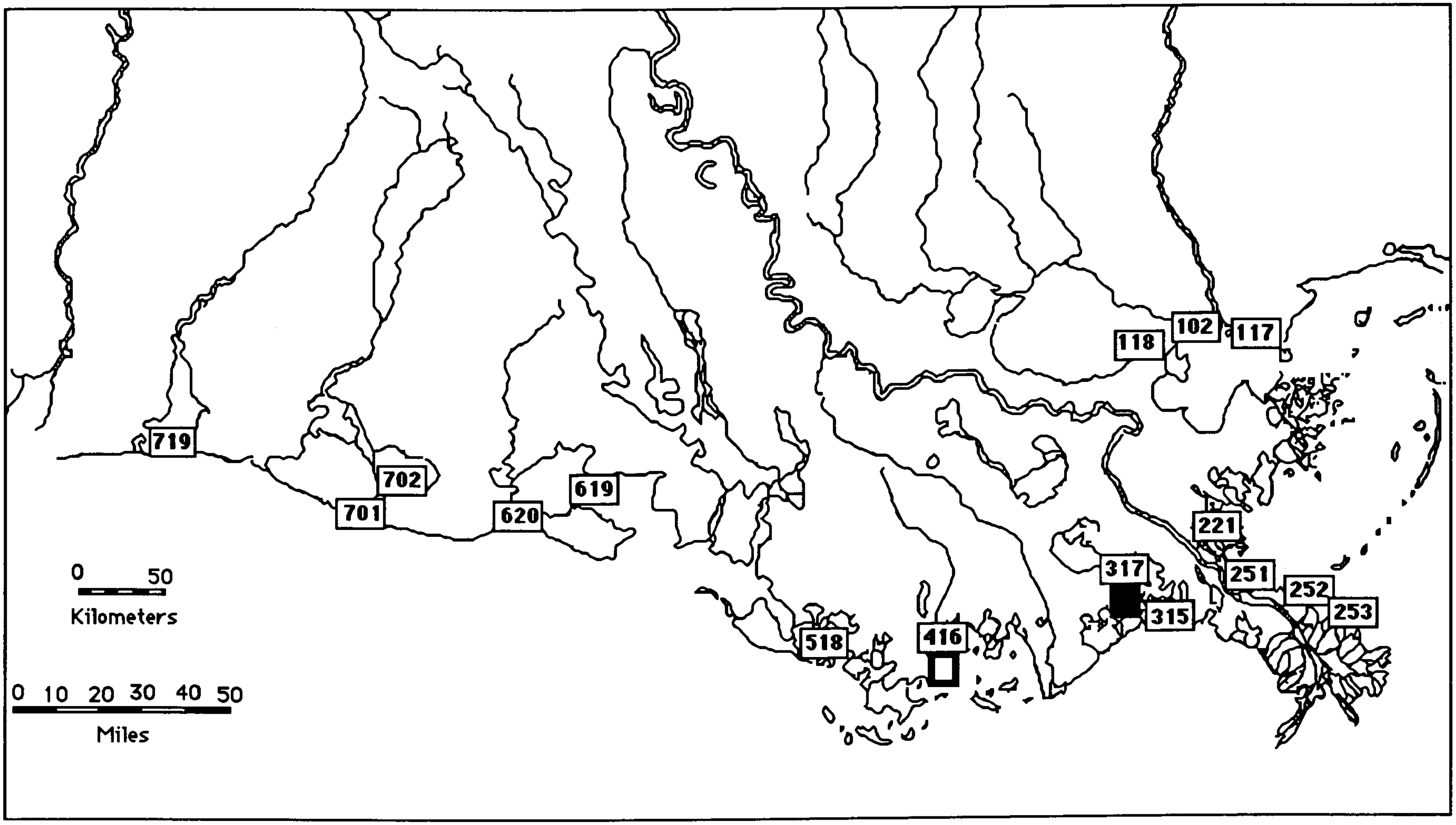

Figure 6-5. Map of the Louisiana coastal zone showing results of the Kendall-Tau test on the variance of the monthly mean salinities from LDWF data. Squares indicate statistically significant negative trends (solid = 95% level, open = 90% level) and circles indicate statistically significant positive trends (solid = 95% level, open = 90% level).

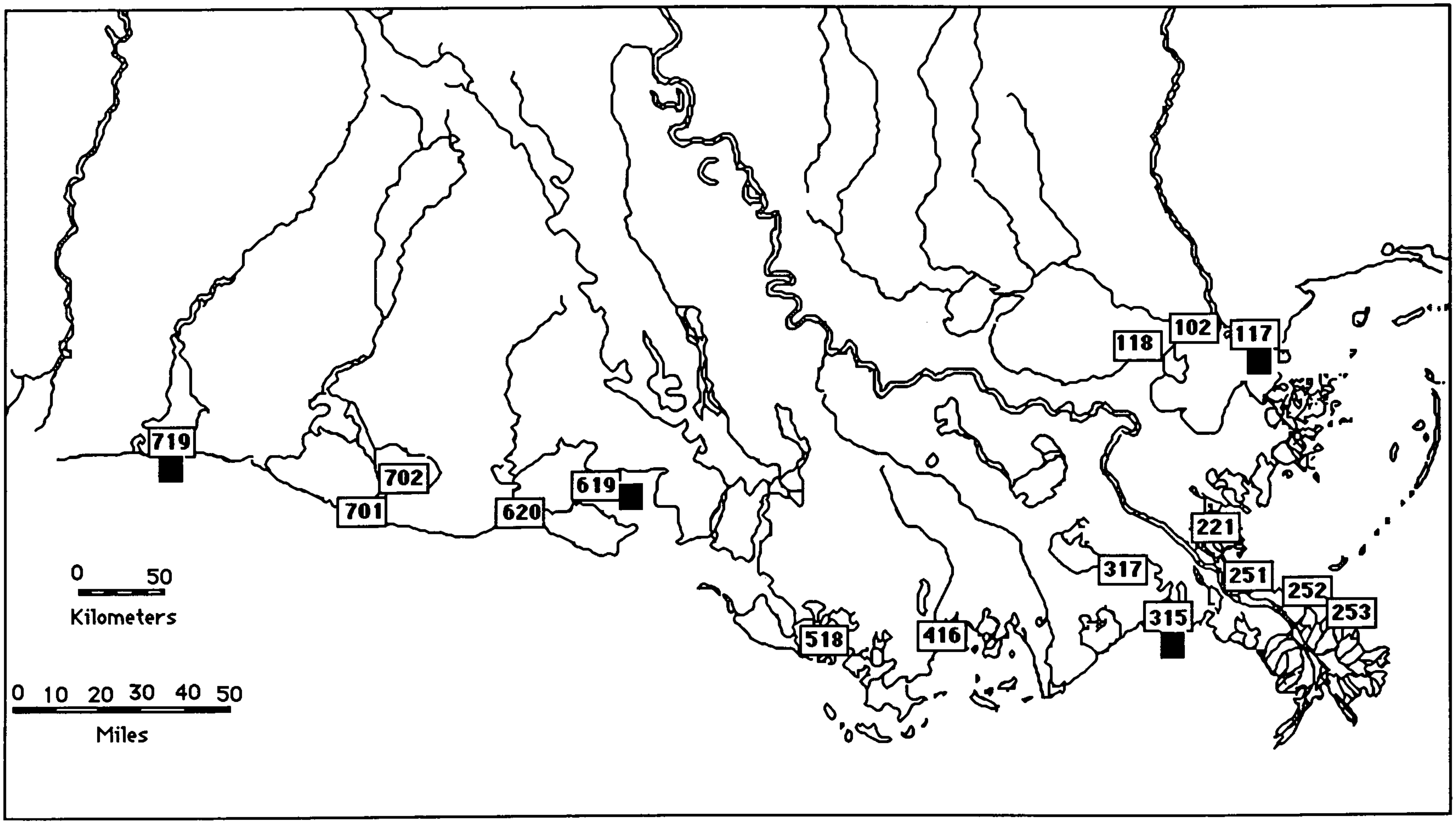

Figure 6-6. Map of the Louisiana coastal zone showing results of the Kendall-Tau test on the LDWF monthly maxima. Squares indicate statistically significant negative trends (solid = 95% level, open = 90% level) and circles indicate statistically significant positive trends (solid = 95% level, open = 90% level).

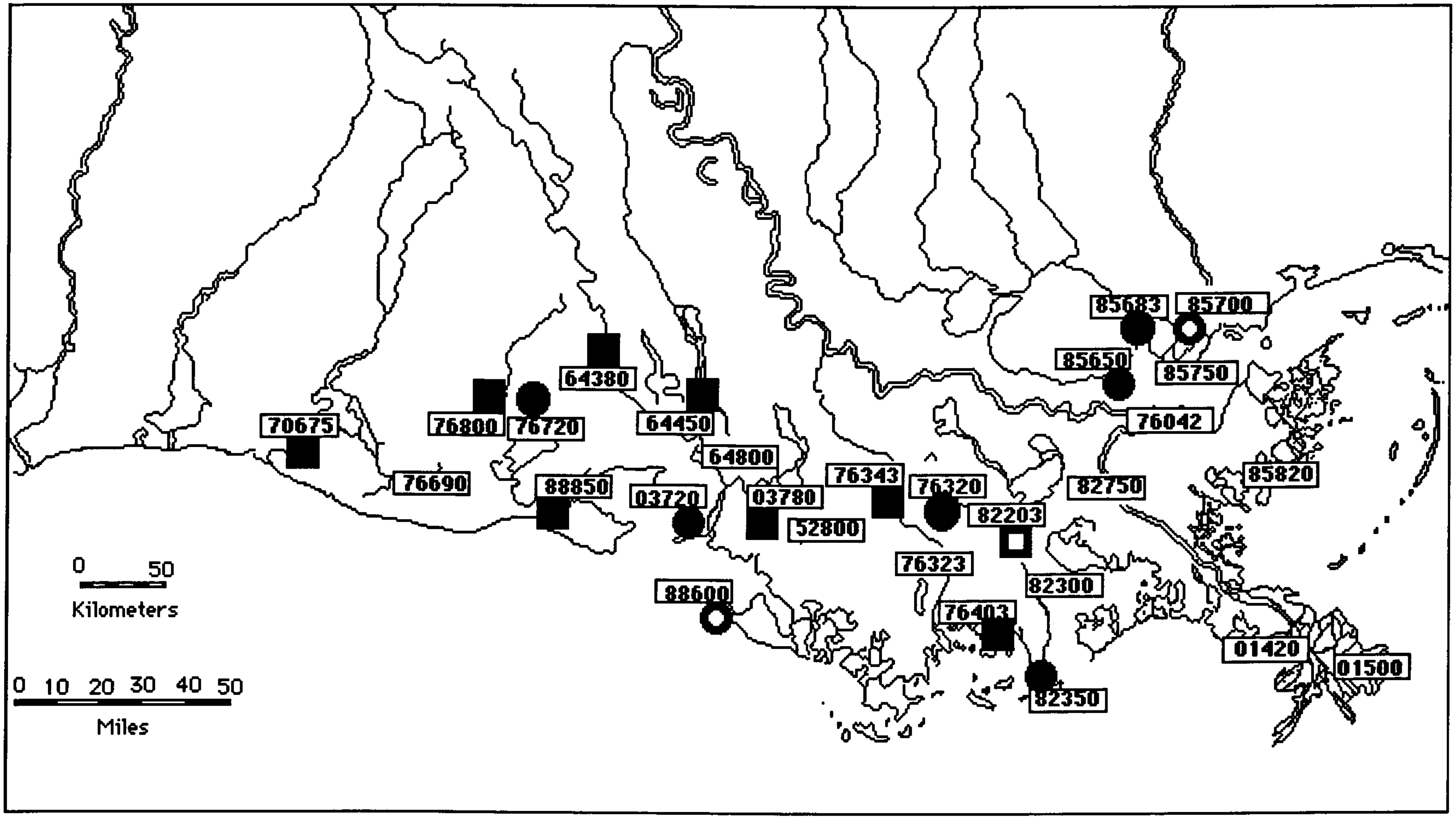

Figure 6-7. Map of the Louisiana coastal zone showing results of the Kendall-Tau test on the COE mean monthly salinities. Squares indicate statistically significant negative trends (solid = 95% level, open = 90% level) and circles indicate statistically significant positive trends (solid = 95% level, open = 90% level).

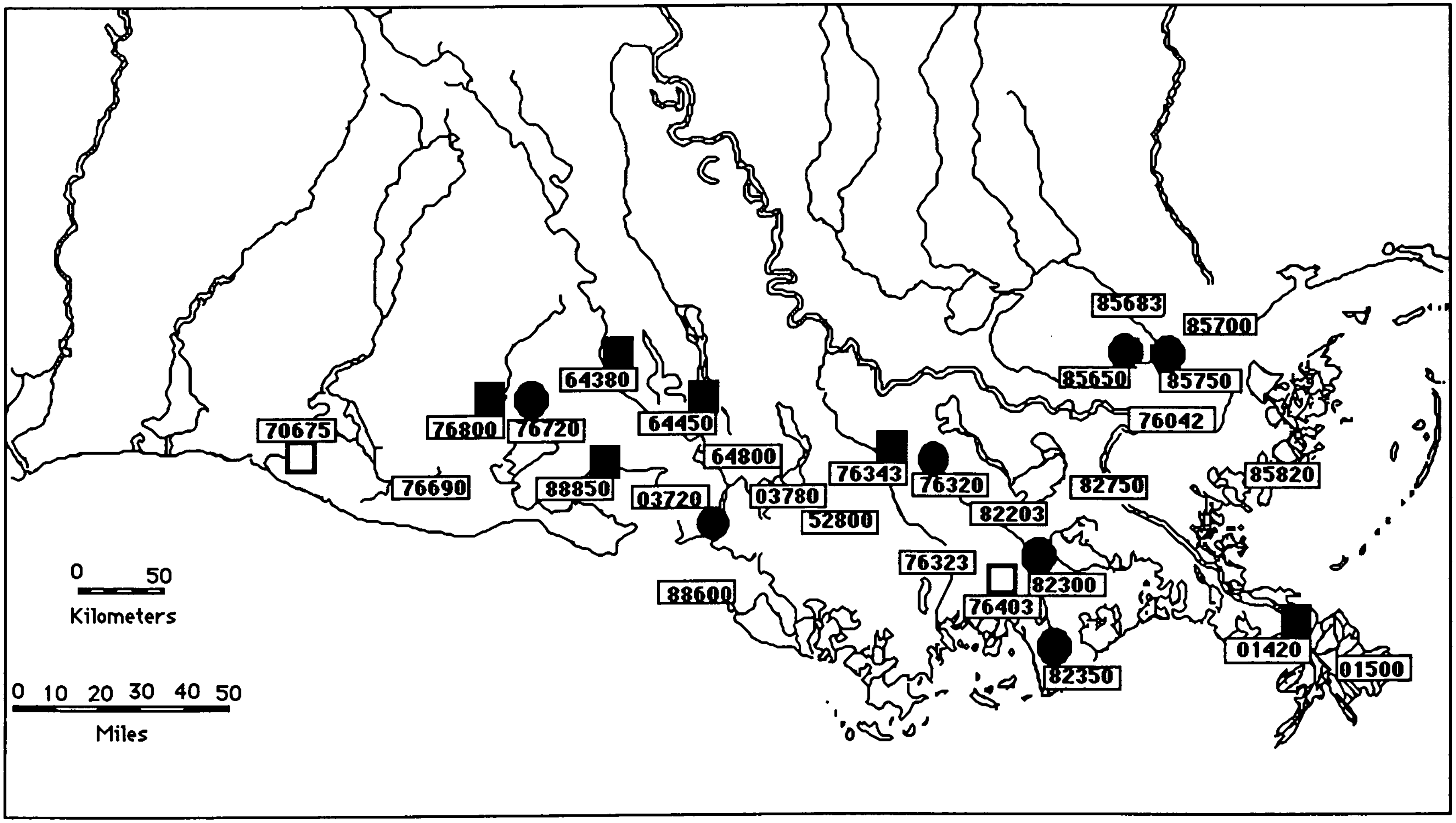

Figure 6-8. Map of the Louisiana coastal zone showing results of the Kendall-Tau test on the variance of the monthly mean salinities from COE data. Squares indicate statistically significant negative trends (solid = 95% level, open = 90% level) and circles indicate statistically significant positive trends (solid = 95% level, open = 90% level).

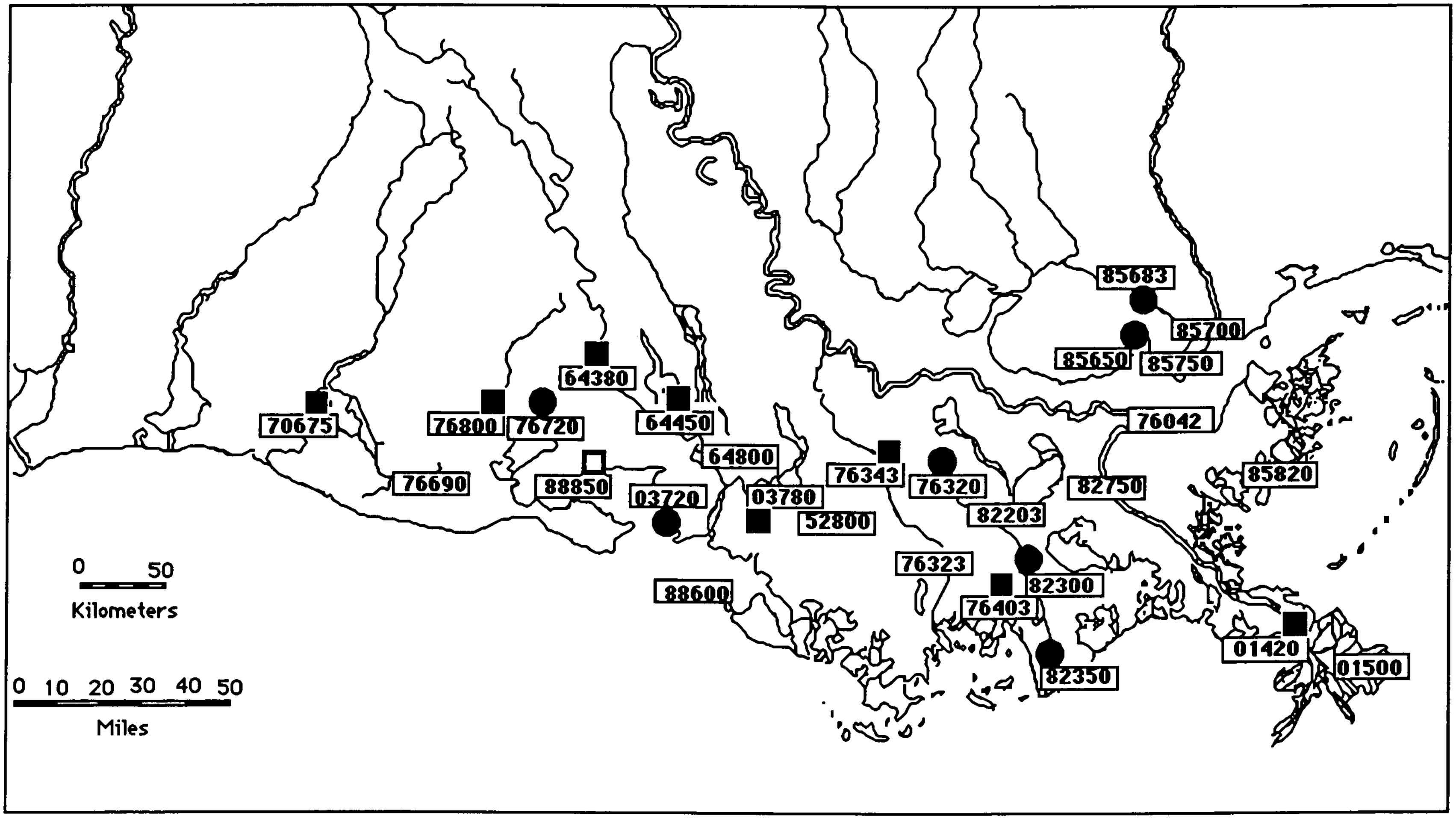

Figure 6-9. Map of the Louisiana coastal zone showing results of the Kendall-Tau test on the COE monthly maxima. Squares indicate statistically significant negative trends (solid = 95% level, open = 90% level) and circles indicate statistically significant positive trends (solid = 95% level, open = 90% level).

When the seasonal Kendall Tau test is applied to the monthly mean salinities from data set C, 17 of the 22 stations show statistically significant trends (Table 6-6, Figure 6-7). Nine stations exhibit decreasing salinity and eight exhibit increasing salinity. Fifteen stations show significant trends in the monthly variance about the monthly mean (Table 6-6, Figure 6-8). Seven of these are positive and eight are negative. Seventeen stations show significant trends in monthly maximum salinities; eight are positive trends and nine are negative (Table 6-7, Figure 6-9). No consistent spatial pattern of positive, negative or zero trend is apparent.

Table 6-7. Results of the Kendall-Tau test on the monthly maxima of the COE data. These results are also presented in Figure 6-9.

Station	Years of Record	Alpha	Trend	Change during Period of Record
			ppt/yr	ppt
01420	48.6	0.044	-0.000	-0.0
03720	17.0	0.000	+0.002	+0.0
03780	18.0	0.003	-0.001	-0.0
64380	24.4	0.000	-0.018	-0.4
64450	29.0	0.000	-0.014	-0.4
70675	31.6	0.018	-0.010	-0.3
76320	30.8	0.000	+0.006	+0.2
76343	20.3	0.000	-0.015	-0.3
76403	19.1	0.015	-0.020	-0.4
76720	29.1	0.000	+0.107	+3.1
76800	19.6	0.003	-0.040	-0.8
82300	19.7	0.017	+0.100	+2.0
82350	21.8	0.000	+0.171	+3.7
85650	31.9	0.000	+0.101	+3.2
85683	3.8	0.046	+2.212	+8.4[a]
85750	24.0	0.003	+0.076	+1.8
88850	24.3	0.021	-0.085	-2.1

[a] Short record

The total change over the record length as predicted by the linear portion of the trends was estimated (Tables 6-2 through 6-7). This information will be used below in discussing stations that exhibit both statistically and biologically significant trends.

Discussion

While statistically significant trends in many aspects of the long-term salinity records are present, the observed trends do not exhibit a consistent pattern across the state of Louisiana nor are the predicted changes, over the record of observation, generally of a magnitude which would appear to be detrimental to the marsh plants (see Chapter 8). We will focus attention on those stations where large changes are predicted. First, we will discuss possible criticisms resulting from the quality of the data. Long-term records of climate, river runoff, and relative sea level rise all show strong variability at time-scales on the order of a decade. Our longest salinity records are only beginning to approach lengths sufficient to allow us to estimate weak trends hidden within this natural variability. For example, many of the records in data set L1 extend from the early sixties to the late seventies when Mississippi River discharge was increasing. This would, presumably, result in lowered coastal salinities. An inverse relation of coastal salinity to river discharge is clearly evident in the data from 1975 to 1979 (Figure 6-10). The COE has had some success in hindcasting seasonal mean salinities assuming runoff, water level, and time to be independent variables (T. Drake, COE, personal communication). We avoided this approach as our interests were less in hindcasting and more in understanding processes and potential effects on plant life. Without a theoretically sound model, we were hesitant to relate salinity directly to runoff. Because of gaps in the data records, e.g., Figures 6-2 and 6-3, we could not even reliably extract that portion of the salinity signal that was coherent with the runoff records.

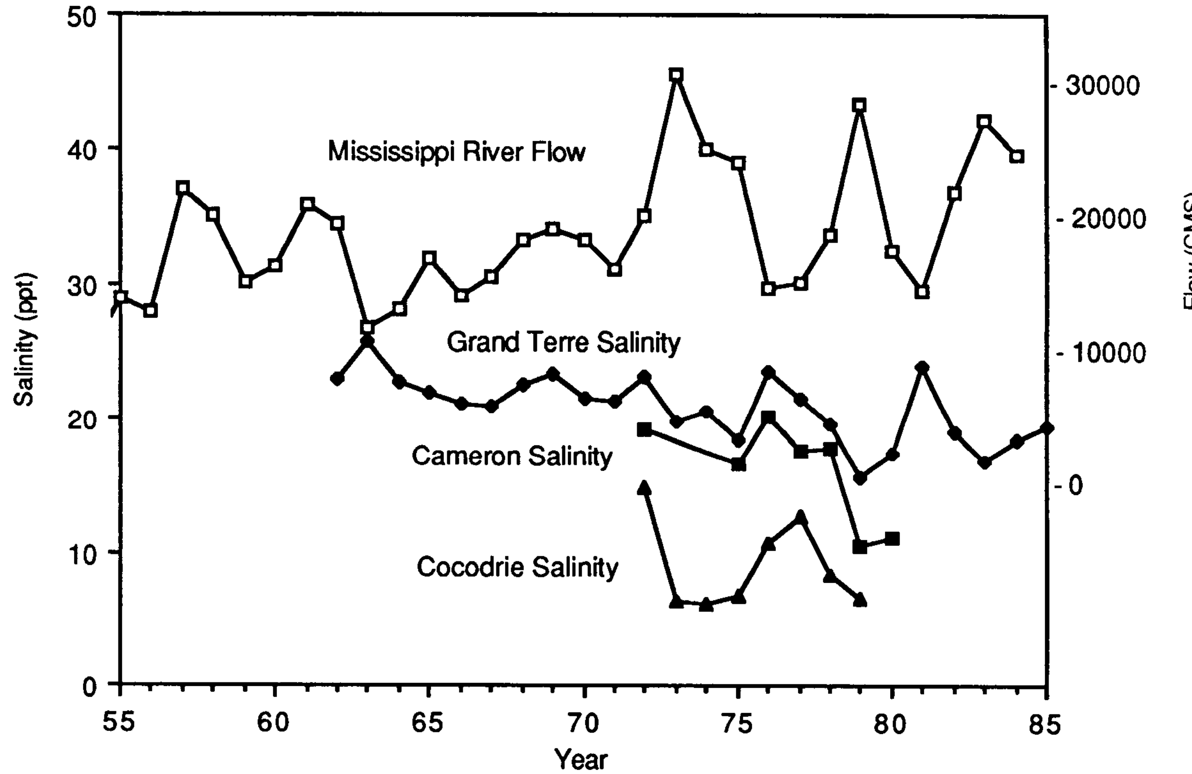

Figure 6-10. Time series plots of the combined annual mean flow of the Mississippi and Atchafalaya Rivers and plots of mean annual salinity from selected LDWF sampling stations. River flow is in thousands of cubic meters per second (CMS) with a 15,000 CMS offset.

Conventional lore suggests that a long-term increase in salinity exists within the estuaries of southern Louisiana and that this increase is responsible for the death of marsh plants and, subsequently, increased land loss. Yet many of our station records, particularly those from data set L1, show a decrease in mean salinity. As mentioned above, this could be caused by increased river discharge. One must also recall that these are near-surface measurements. If upland runoff is increasing or the local water depth is increasing, then the deeper salinities could increase, while the surface salinities decrease (Bowden and Hamilton, 1975; Pritchard, 1967). Nevertheless, it is precisely these surface salinities that are most likely to affect marsh plant health, either through overbank flooding or groundwater flow within the root zone, and these salinities often appear to exhibit a decreasing trend.

Consideration of coastal vegetation changes between 1948 (O'Neill, 1949) and 1978 (Chabreck and Linscombe, 1978) suggests two regions of major change in marsh vegetation type near sites where we have long-term data sets. An area in the vicinity of lower Bayou Lafourche has changed from brackish to salt marsh, while a region between the Mississippi River Delta and Lake Pontchartrain has gone from salt to brackish vegetation. Our data from lower Bayou Lafourche show an increase in mean salinity, salinity variance, and maximum salinities. The lower bayou was dredged to a depth of 6 m (20 ft) in 1968 and is maintained at a minimum depth of 2.7 m (9 ft). No evidence of intervention in 1968, though, is apparent in the salinity records. The salinity data we analyzed from the region that changed from salt to brackish marsh is only a 12-year record. It does show, though, the anticipated decreasing salinity trend (Figure 6-4). The records from two other regions also show large changes in mean salinity. A number of stations in Lake Pontchartrain exhibit increasing trends in both mean salinity and salinity variance. The Mississippi River Gulf Outlet (MRGO) was opened in 1964. Sikora and Kjerfve (1985) analyzed salinity records from the lake since 1964 and found a salinity increase, which they attributed to the completion of MRGO. While our analysis of a longer data set corroborates their conclusion of a secular increase, the records are not conclusive as to the cause. Finally, the stations east and west of the Vermilion Locks on the Intracoastal Waterway show secular trends of opposite sign in the salinity distributions. The most striking change is in the salinity maxima, with those east of the locks increasing and those west of the locks decreasing.

The most extensive extant analysis of long-term salinity records from the Louisiana coastal zone is that by Byrne et al. (1976). They analyzed data from within a single drainage basin, the Barataria Bay watershed. Their analysis suggests that salinities throughout the system, prior to 1962 when the Barataria waterway dredging began, were lower than those subsequent to that date. Byrne et al. treated their data somewhat differently than we did, but they were equally aware of the data quality problem caused by missing data, sample site location, sampling frequency, and record length, as well as external natural variability. It is interesting to note that, while one of their stations was not available to us, by using somewhat longer records and different data quality-control criteria, we conclude that a negative trend in mean salinity is presently occurring at the mouth of Barataria Bay, and no mean salinity trend occurs in the upper reaches of the Bay.

Conclusions

Analyses of existing salinity time series show the following:

(1) Statistically significant trends in mean salinity, salinity variance, and maximum salinity exist.

(2) These trends are of both sign with no apparent spatial pattern.

(3) Linear estimates of the trend magnitudes suggest that, over the period of available record, the resultant changes in mean salinity, salinity variance, and maximum salinity have generally been small.

(4) The magnitude of the predicted changes in mean salinity are generally small enough as to be non-detrimental to the adjacent marsh plants. In a few locations trends in the salinity regime are large and major changes in marsh vegetation have been observed in the area.

(5) Natural variability in these systems is high and may hide weak trends. Nevertheless, extensive coastal degradation and land loss has been documented over time periods of the same order as our data records. If this were caused by a trend in the salinity regime, the present analysis should have identified the trend.

Chapter 7

MEASUREMENTS OF SALTWATER MOVEMENT IN A MARSH SYSTEM

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Salt water from the continental shelf has the potential for moving up a channel that is deepening through natural subsidence or dredging (Chapter 5). Increased salt content in the interstitial waters of the marsh impacts the growth rate of marsh plants. Therefore, it is necessary to understand the mechanisms that allow estuarine salinities to propagate into the marsh proper. Many processes influence the salinity of interstitial waters. These include overbank flooding followed by vertical percolation, rainfall, evopotranspiration, and groundwater flow. At the suggestion of the Science Review Board, a field study was designed to determine the importance of lateral groundwater flow on the penetration of salt from the estuary into the adjacent marsh.

There are very few field studies on the mechanisms of saltwater migration in the marshes. Lindberg and Harriss (1973) measured interstitial salinities in regularly flooded salt marshes along the northeastern Gulf Coast of Florida. In their study area, pore water salinities near the marsh surface were controlled by a vertical density gradient of time-varying sign induced at the sediment-water interface, while salinities at depth (> 20 cm) were controlled by groundwater movement . Hemond and Fifield (1982) investigated subsurface flow in salt marsh peats near Falmouth, Massachusetts, through the use of a numerical model. In general, their data indicated that tidal influence is slight, with evapotranspiration the dominant process that controls the upward flux of groundwater in the interior marsh (50 m inland). They also concluded that peat permeability is one of the dominant factors controlling subsurface flow in the marshes.

The simplest model of the estuary-marsh system is based on Darcy's Law. This is a linear model in which salinity variations in the estuary are attenuated and delayed as they propagate into the marsh. The amplitude and phase response of the system may be spatially variable and depend on the physical properties of the fluid and the geotechnical properties of the marsh sediment. By measuring the time history of salinity and pressure at various points throughout the system, one should be able to determine the characteristics of the linear system, in particular, its transfer function, using standard frequency domain time series analysis (Mason and Zimmerman, 1960).

Field sites were chosen within the brackish to intermediate marsh of south-central Louisiana. Within this region, monthly mean water level exhibits a seasonal low during the winter (Marmer, 1954) at the same time that storms tend to force saline Gulf waters up into the estuaries (Chuang and Wiseman, 1983). We wished to initiate field experiments during this time period to minimize the effects of overbank flooding and to maximize the salinity signal available for analysis. Difficulties with purchasing and testing the equipment delayed the start of the program. An initial drift in the data from the measurement system itself under field conditions further hampered the study, as will be described below. Some data were salvaged from the early field work, but the most reliable data were collected from May through June during a time when runoff and water level were increasing, and salinity perturbations in the estuary adjacent to the marsh were diminishing in amplitude and frequency.

Field Methods

Field data was collected during two types of experiments: (1) intensive three-day surveys of marsh salinity and (2) time series data collection. The goal of the initial, intensive surveys was to determine the distance inland from a canal or bayou that salinity variations may be reasonably expected to propagate. The results from these studies were used to determine the placement of six recording water level and salinity gages that were subsequently deployed along a transect perpendicular to the marsh edge for the time series data collection.

Figure 7-1 presents a map of the Lake DeCade area (in eastern Terrebonne marshes) with the study sites indicated. Intensive, three-day studies were conducted prior to deployment of the recording gages. During these studies, sampling wells were placed along three parallel transects extending inland from the bayou or canal. Wells were placed at 0, 1, 3, 5, 10, 15, 20, 25, 35, and 45 m inland along the transect. Salinity samples were collected from the wells every three to six hours throughout the three-day study period. In addition to the samples collected in the wells, samples were also collected from the adjacent bayou. During each sampling period, samples were taken from the wells using a vacuum pump and brought back to the boat where they were analyzed, using a "Yellow Springs Instrument" conductivity meter. Laboratory calibration of the meter indicated that the instrument is reliable to about 0.25 ppt.

Vertical salinity profiles were measured using sampling pipes made from 1.3-cm (1/2") diameter PVC plumbing pipe. The pipe was cut to the desired length, a PVC point was cemented on the end, and a series of small holes was drilled in the pipe about 10 cm above the point. The pipes were then inserted into the marsh until the holes were at the desired depth for sampling and allowed to stay in place for about 30 minutes. The pipes were then withdrawn from the marsh, and the water that had collected in the pipe was withdrawn, placed in small vials, and returned to the lab for salinity determination. In addition, samples were also drawn up from several depths in the sampling wells used for the recording instruments (see below). Drawdown experiments were also conducted to estimate *in-situ* lateral marsh hydraulic conductivity, and cores were collected to determine vertical hydraulic conductivity in the laboratory.

Two time series experiments (RB1 and RB2) were conducted in the marshes along Raccourci Bayou, a natural system. The other two time series experiments were conducted in the marshes along Superior Canal (SC) and Raccourci Canal (RC). Recording water level and conductivity gages were placed along a transect extending from the water's edge to 75 m inland. The gages were housed in wells that extended from 0.75 m above the marsh surface to a depth of 1.0 m below the marsh surface. The wells were constructed of 0.13 m (5 in) diameter PVC pipe and had inlet holes beginning with a depth of 15 cm below the marsh surface and extending to the bottom of the well, which was open. This construction was designed to ensure free flow around the sensor package (43 cm long) and to preclude surface waters from entering the wells, thus sampling the water from within the root zone. It is not clear, however, if the seal between the marsh and the well was compromised, thus allowing surface waters to flow down the sides of the well and reach the sensor.

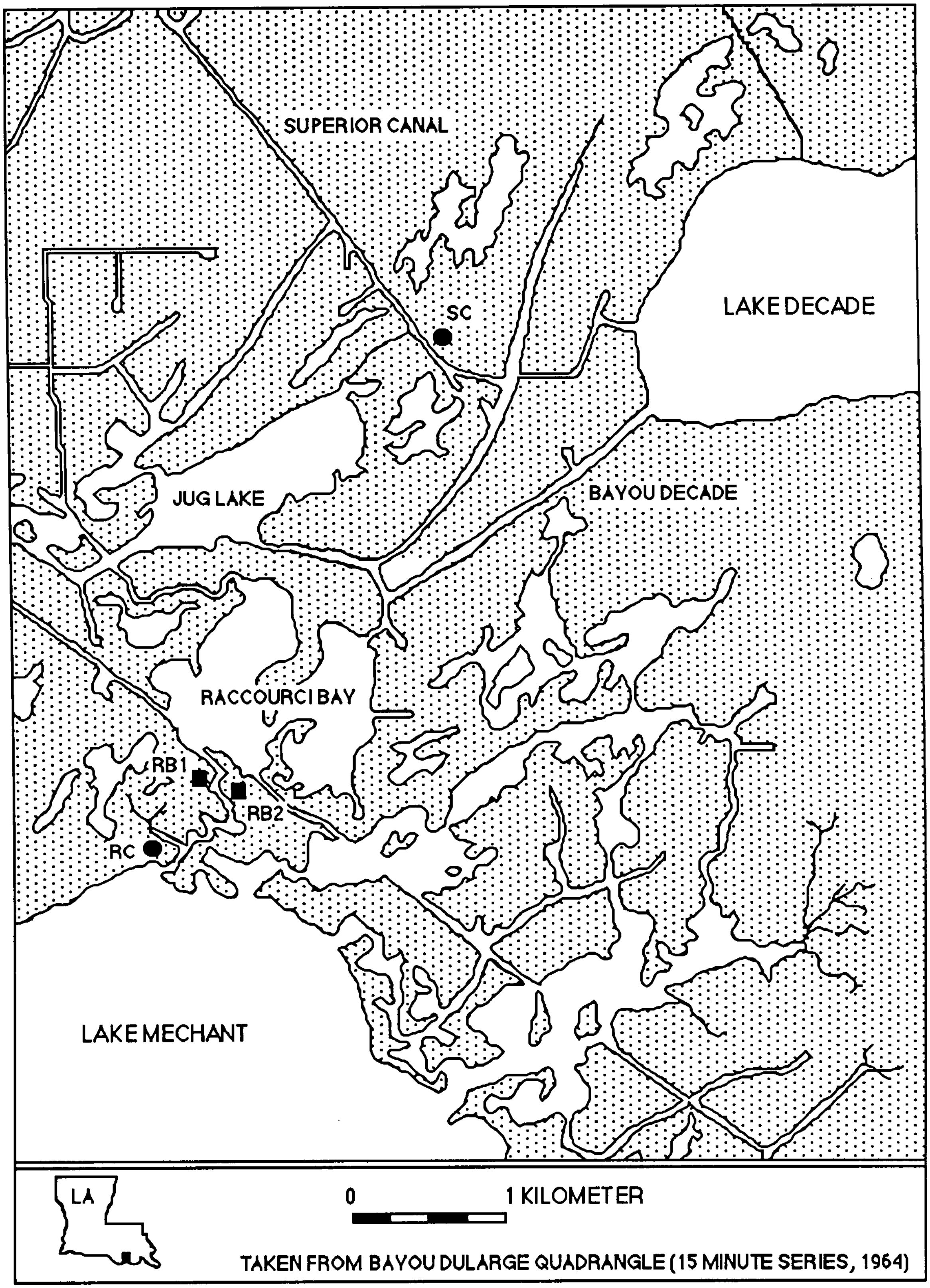


Figure 7-1. Map of the Lake DeCade area showing the sample sites. SP = Superior Canal, RC = Raccourci Canal, RB1 = Raccourci Bayou 1 and RB2 = Raccourci Bayou 2.

This design was changed after the first two deployments (RB1 and SC), based on discussions with the Science Review Board. For the last two deployments (RB2 and RC), the wells were modified by placing a cap on the bottom of the well, to preclude the vertical movement of saline water from below the root zone into the wells.

The gages used, ENDECO® TYPE 1152 Density Compensating Water Level Recorders, measure water level with a temperature and atmospheric pressure compensated strain gage transducer, conductivity with an inductance cell, and temperature with a thermistor. The pressure, temperature, and conductivity data are recorded on removable solid state memory modules. Upon receiving the instruments, the calibration of all six were checked in-house. The results of the calibrations are summarized in Table 7-1. All of the meters performed to specifications, yielding reliable data over the entire range of salinities from 0 to 20 ppt.

Table 7-1. Summary of calibration data for the instruments used during the study.

A. Salinity:

Gage	Intercept (ppt)	Slope	R^2
014	0.10	0.94	0.9989
015	0.10	0.98	0.9987
016	0.11	0.98	0.9988
017	0.11	0.98	0.9985
018	0.07	0.99	0.9988
019	0.06	1.00	0.9989

B. Temperature

Gage	Intercept (C°)	Slope	R^2
014	0.03	0.97	0.9964
015	0.07	0.95	0.9889
016	0.02	0.96	0.9897
017	0.05	0.96	0.9890
018	0.26	0.95	0.9918
019	0.01	0.96	0.9951

A severe positive bias was detected in the salinity data from the first two deployments (RB1 and SC). Upon retrieving the gages, we noticed the meters from the inland marsh sites were coated with a black film, a conducting coating that presumably developed from a reaction between the sulfides in the marsh and the anti-fouling paint used on the gages. For further deployments, all of the paint was stripped from the instruments. Before this was done, however, the gages were re-calibrated with the coating intact, to develop a regression equation that would allow us to remove the bias from the field data. Using this technique, a small portion of the data from the second deployment was recouped, albeit with a somewhat reduced accuracy: ± 0.5 ppt compared with ± 0.2 ppt for a new gage. After the anti-fouling paint was removed, the gages were again re-calibrated. The calibrations for the gages without the paint were not statistically different from the calibrations made when the gages were purchased.

Table 7-2 lists the locations of the gages along the transect for each of the four experiments and a matrix of data recovery for each of the experiments. Sketch maps showing a plan view of each of the sites are presented in Figure 7-2. The maps show the relation between the gage locations and the major ponds within each experimental area. The maps also indicate the distances and direction from the inland end of the transect to the nearest open water body (lake or bayou). Marsh surface elevations along the transect line and the locations of the gages along that line are presented in Figure 7-3. The elevations are in centimeters relative to an arbitrary base level (the lowest point measured).

Table 7-2. A. Listing of gage locations for each of the four experiments. Indicated are the experiment number, location of the experiment, the dates of the experiment, and the location of the gages along the transect. Distances are meters from the water's edge for Experiment 1 (RB1) and Experiment 3 (RB2) and meters behind the spoil bank for Experiment 2 (SC) and Experiment 4 (RC).
B. Matrix of data recovery. Indicated, for each experiment, are the gages where reliable data was obtained. S refers to salinity, T refers to temperature, and P refers to pressure (water level). Data is available at the locations marked with X.

A.

		Gage Locations					
Experiment	Dates	014	015	016	017	018	019
1. RB1	01/23/87-03/11/87	Bayou	Berm	5M	10 M	35 M	75 M
2. SC	03/13/87-04/21/87	Bayou	Spoil	1 M	5 M	35 M	75 M
3. RB2	05/08/87-06/04/87	Bayou	Berm	5 M	10 M	35 M	75 M
4. RC	06/05/87-07/21/87	Bayou	Spoil	1 M	5 M	35 M	75M

B.

	014			015			016			017			018			019		
Experiment	S	T	P	S	T	P	S	T	P	S	T	P	S	T	P	S	T	P
1 - RB1	X	X	X	X	X	X	X	X	X		X	X		X	X		X	X
2 - SC	X	X	X	X	X	X	X	X	X		X	X		X	X	X	X	X
3 - RB2	X	X	X		X	X	X	X	X	X	X	X	X	X	X	X	X	X
4 - RC	X	X	X	X	X	X	X	X	X	X[a]	X[a]	X[a]	X	X	X			

[a] Only first half of record is good.

Results

Intensive Surveys

Figure 7-4 presents marsh salinity data from the three-day sampling trip at RB1 in January, 1987. Plots are presented for various times throughout the sampling period for each of two transects, which were separated by 20 m. The highest salinities occurred on the berm, then decreased with distance into the marsh, and finally increased again at the inland end of the transect. This increase at the end of the transect appears to be related to the proximity of ponds that may be serving as a second source of salt water to the system. The salinities in the marsh were generally higher than the salinities in the adjacent bayou. This general pattern occurred at both of the transects although there is lateral variability in the system. The data at SC are presented in Figure 7-5. Again, salinity increases on the spoil bank, with a decrease as one moves into the marsh, particularly on transect 1. As was the case with RB1, this location also exhibited marsh salinities higher than the bayou salinities at the time of sampling.

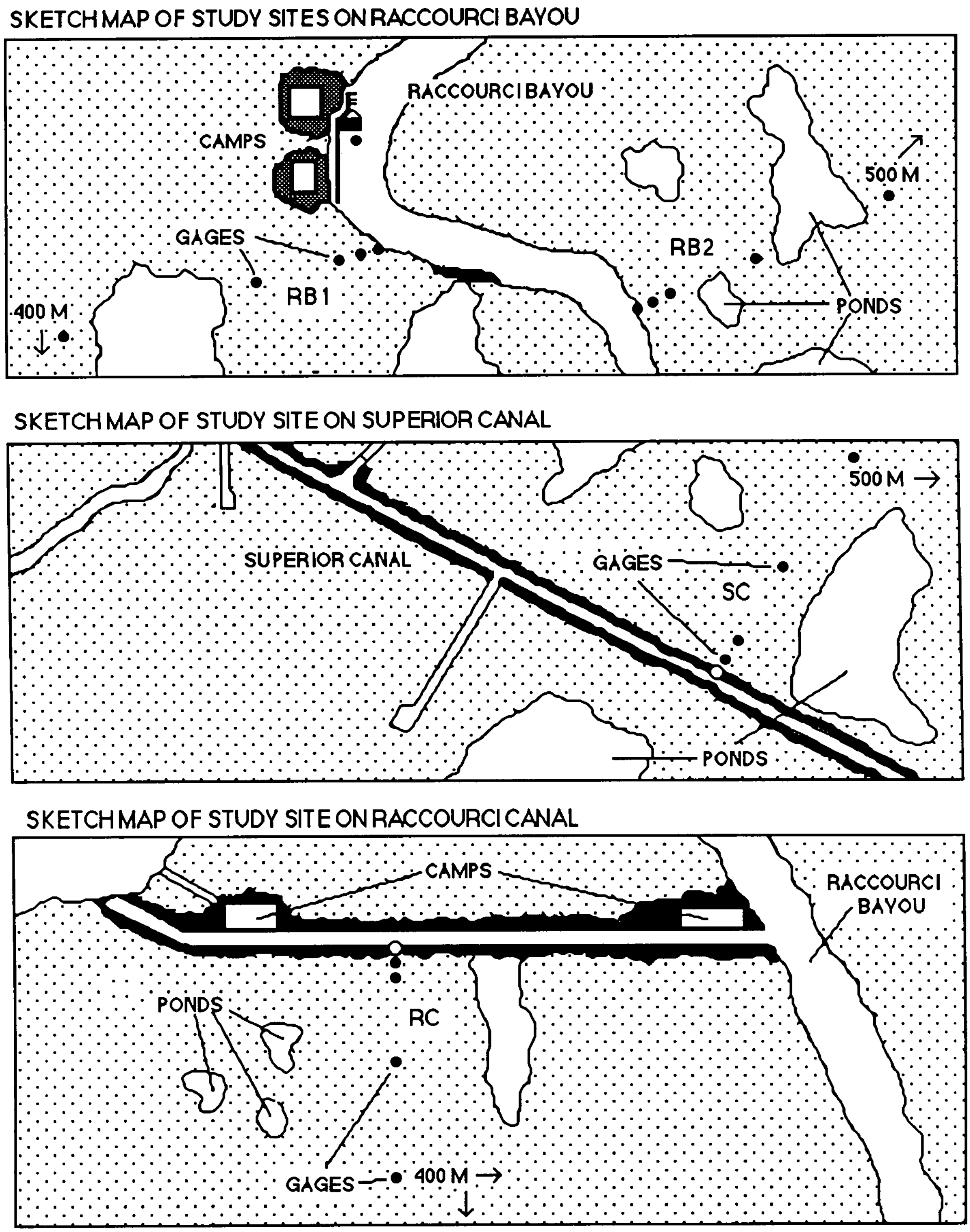


Figure 7-2. Sketch maps of the study sites showing the locations of the water level-salinity gages (circles). The locations of the major ponds in the marsh at each site are also shown. Numbers at the end of each transect indicate the approximate distance and direction to the nearest open water body. Black areas represent spoil banks.

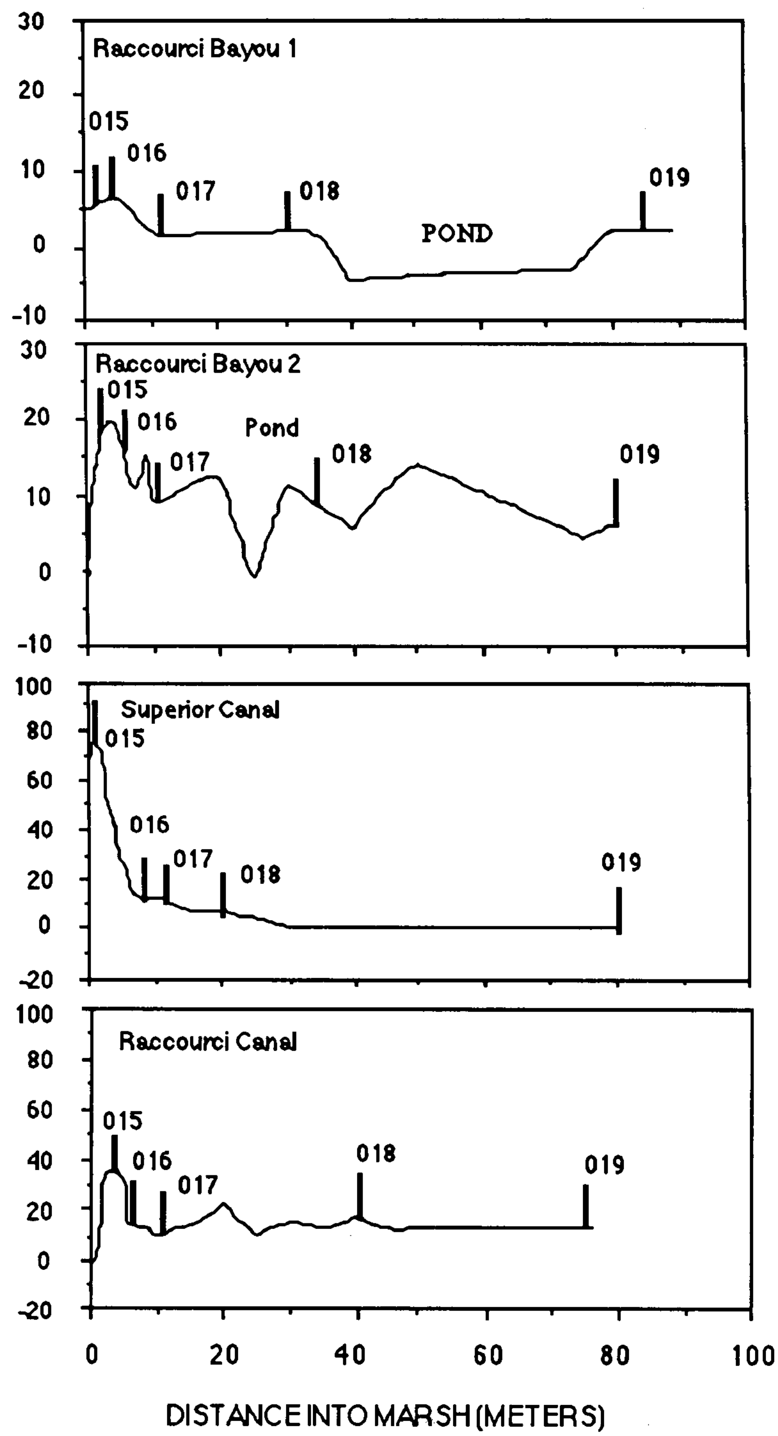
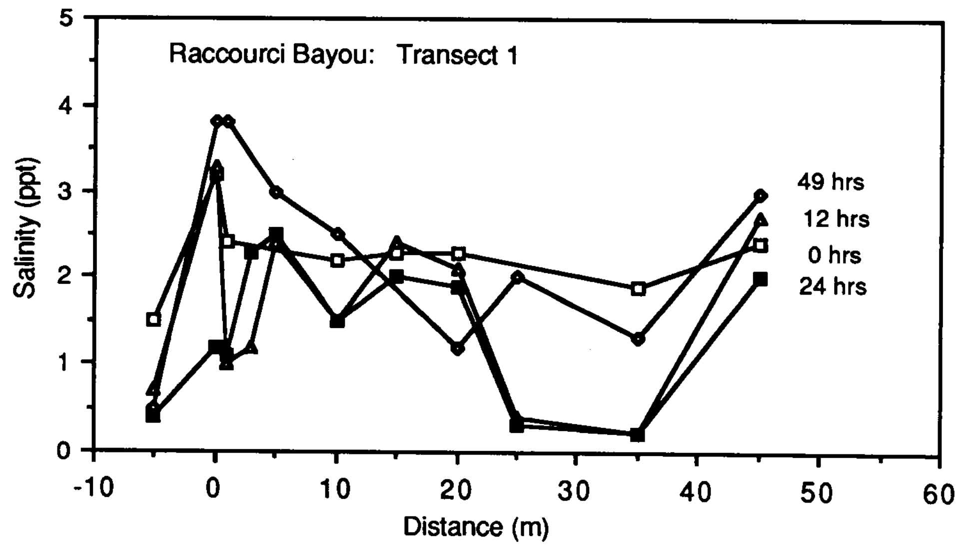


Figure 7-3. Marsh elevations along the water level-salinity gage transect for each site. Indicated are the relative marsh elevations (in centimeters) as a function of distance into the marsh (in meters). The black rectangles indicate the locations of the gages. Although not indicated on this figure, there was also a gage (014) in the bayou (or canal) at each site.



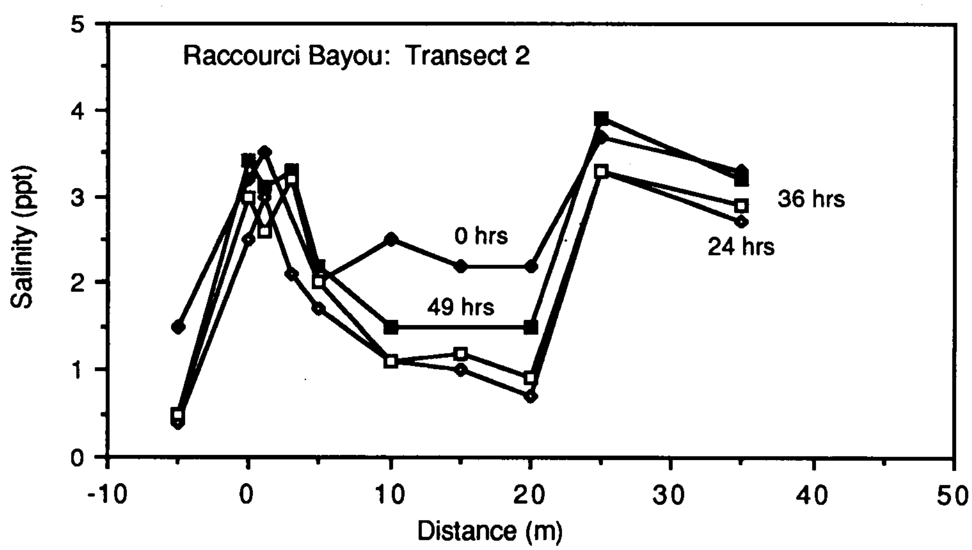
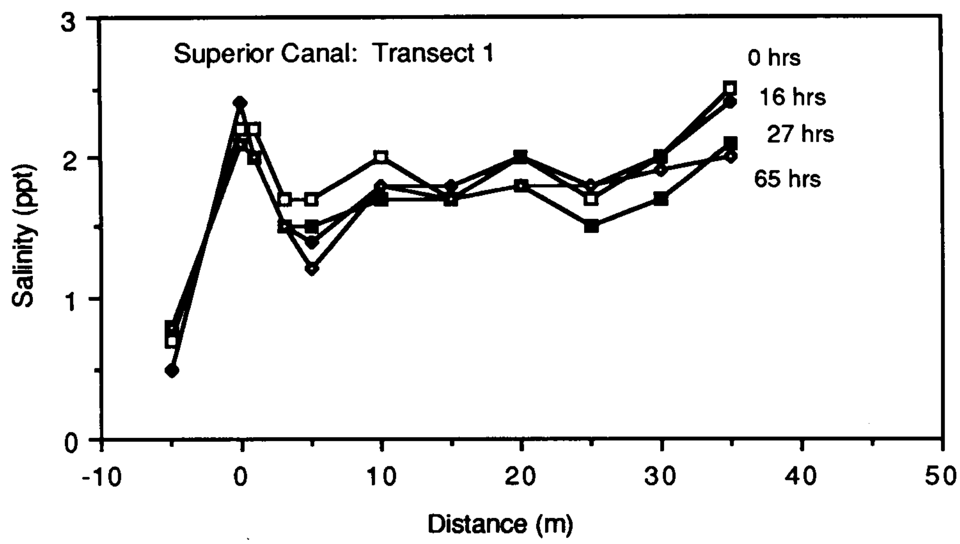


Figure 7-4. Marsh salinity (in PPT) as a function of distance into the marsh (in meters) for Raccourci Bayou 1 at two parallel transects from the January three-day sampling trip. The value for hours on each plot indicates the elapsed time since the start of the sampling trip.



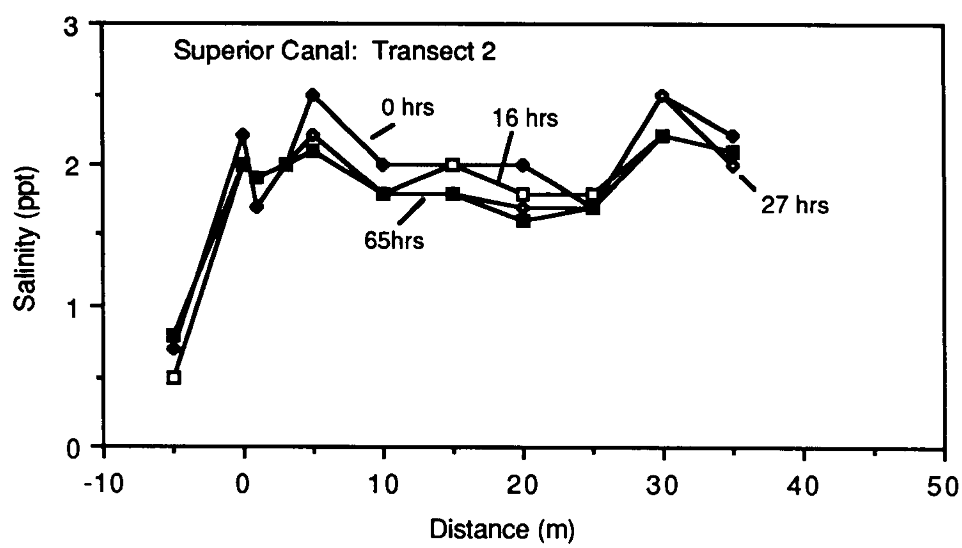


Figure 7-5. Marsh salinity (in PPT) as a function of distance into the marsh (in meters) for Superior Canal at two parallel transects from the March three-day sampling trip. The value for hours on each plot indicates the elapsed time since the start of the sampling trip.

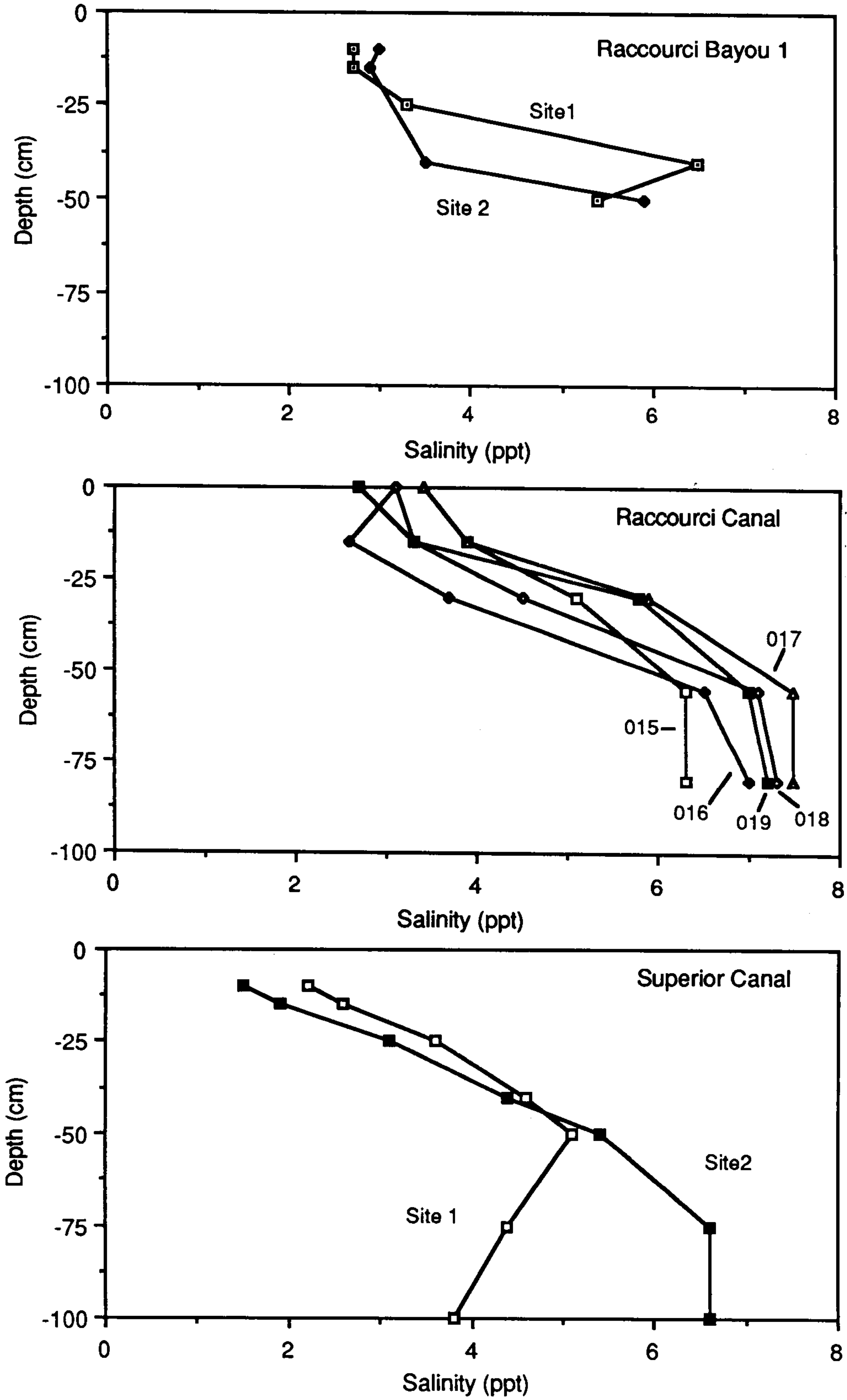


Figure 7-6. Vertical salinity profiles from (top to bottom) RB1, RC and SC. Samples from RB1 and SC were collected using salinity sampling pipes; salinities from RC are from samples collected from the wells used for the time series data collection. Sites 1 and 2 for RB1 and SC were replicates collected at 10 m inland.

Figure 7-6 presents vertical salinity profiles for the upper meter of the marsh at RB1 and SC. At each of the study areas, the samples were collected at two sites, about 15 m apart, 10 m into the marsh at each of the study areas. Vertical salinity profiles measured in the sampling wells from RC are also shown in this figure. The vertical salinity data are quite consistent at all of the locations and show increases of about 5.0 ppt over a depth of 50 cm. Salinities appear constant at depths greater than 50 cm, but this is an artifact of the design of the well itself since free connection with the marsh only extend downward for 50 cm. Note that the gradient in the wells (above 50 cm) is the same as the gradient measured with the sampling pipes, indicating that the wells reflect the actual salinity conditions within the marsh mat.

Geotechnical Properties

Field and laboratory measurements were used to estimate the hydraulic conductivity of the upper marsh mat, the major geotechnical parameter of interest for this study. To the accuracy of our measurements, hydraulic conductivity is directly proportional to permeability. Field studies consisted of drawdown experiments, outlined in Cedergren (1967), where water is pumped from a 1 m deep well which is monitored during refilling. The rate at which the well refills is used to estimate hydraulic conductivity. Laboratory analyses were conducted (by the LSU Department of Civil Engineering) on cores to determine vertical conductivities. Tests were run on the cores at salinities of 1.0 and 10.0 ppt to investigate the possible effects of salinity on the permeability.

The field determinations gave hydraulic conductivities of about 5.0×10^{-5} cm/sec and 9.0×10^{-4} cm/sec for the spoil bank and inland marsh, respectively. There was no noticeable difference between any of the inland marsh sites. The natural levee gave values that were only slightly lower than the spoil bank. It is not clear that these differences are significant because of the large degree of variability in the system. In general, we can conclude that the hydraulic conductivities are about an order of magnitude larger in the inland marsh than the spoil bank or natural levee. The laboratory analyses gave estimates of vertical conductivities ranging from 1.0×10^{-5} cm/sec to 7.0×10^{-5} cm/sec for the marsh substrate. There was no detectable influence of salinity on the hydraulic conductivity as has been noted in other soils (Goldenberg, et al., 1983).

Time Series Data

Fast Fourier transform techniques (Bendat and Piersol, 1986) were used to estimate energy spectra and coherence squared for signals from which the mean and linear trend had been removed. All the spectrum and coherence estimates were performed with 12 degrees of freedom. No smoothing across estimates was done so that neighboring estimates are independent. Because of the differing record lengths, this procedure provides estimates in bands which are not all centered at the same frequencies. We were interested in resolving the diurnal band, as the dominant tide is diurnal near the study sites, and have thus restricted our analyses to frequencies of 0.05 cycles per hour (20 hour periods) and lower. All the spectrum and coherence estimates are presented in Appendix E. Plots of the time series data of salinity and pressure (water level) used for these analyses are presented in Figures 7-7 through 7-10.

When considering the coherence estimates, signals were regarded as significantly coherent if their coherence level was statistically different from zero at the 95% level. Significant coherence in a single frequency band was generally ignored unless there was a physical justification for accepting it as real. Thus, an apparently significant but isolated

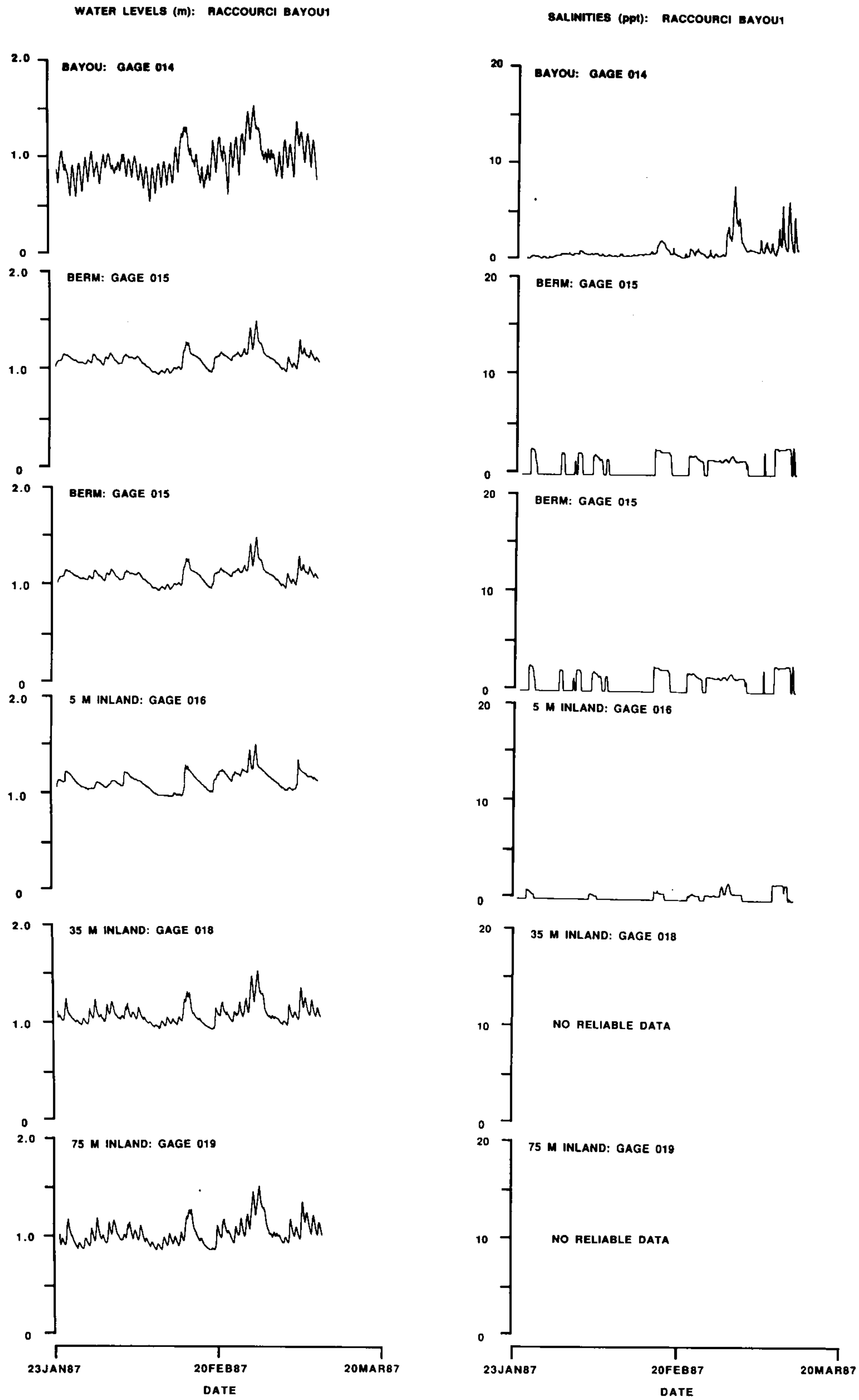


Figure 7-7. Time series plots of water levels, in meters (left) and salinities, in PPT (right), from the gage deployments at Raccourci Bayou 1. The figures are stacked as follows (top to bottom): gage 014 (bayou), gage 015 (berm), gage 016 (5 m inland), gage 017 (10 m inland), gage 018 (35 m inland) and gage 019 (75 m inland).

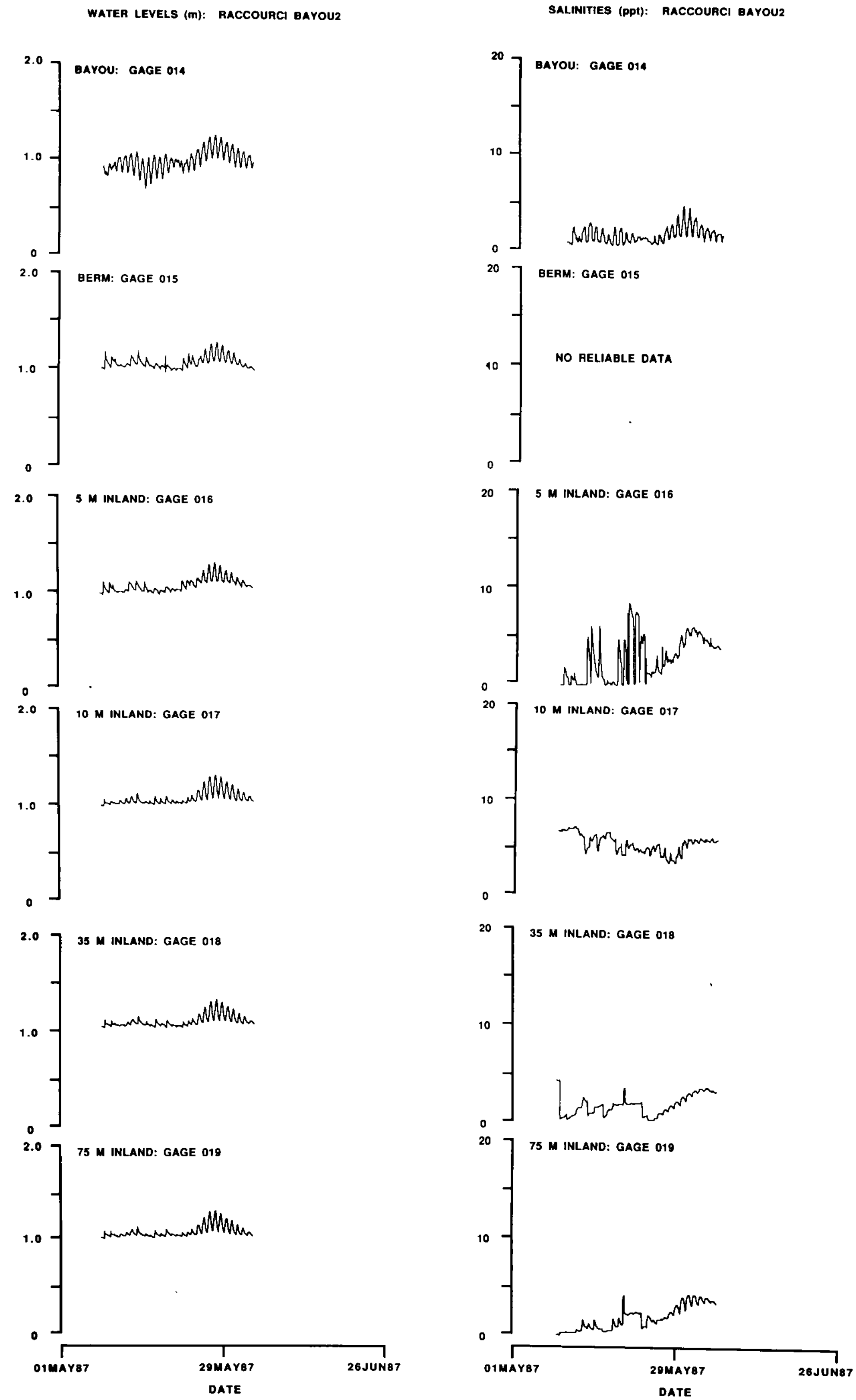


Figure 7-8. Time series plots of water levels, in meters (left) and salinities, in PPT (right) from the gage deployments at Raccourci Bayou 2. The figures are stacked as follows (top to bottom): gage 014 (bayou), gage 015 (berm), gage 016 (5 m inland), gage 017 (10 m inland), gage 018 (35 m inland) and gage 019 (75 m inland).

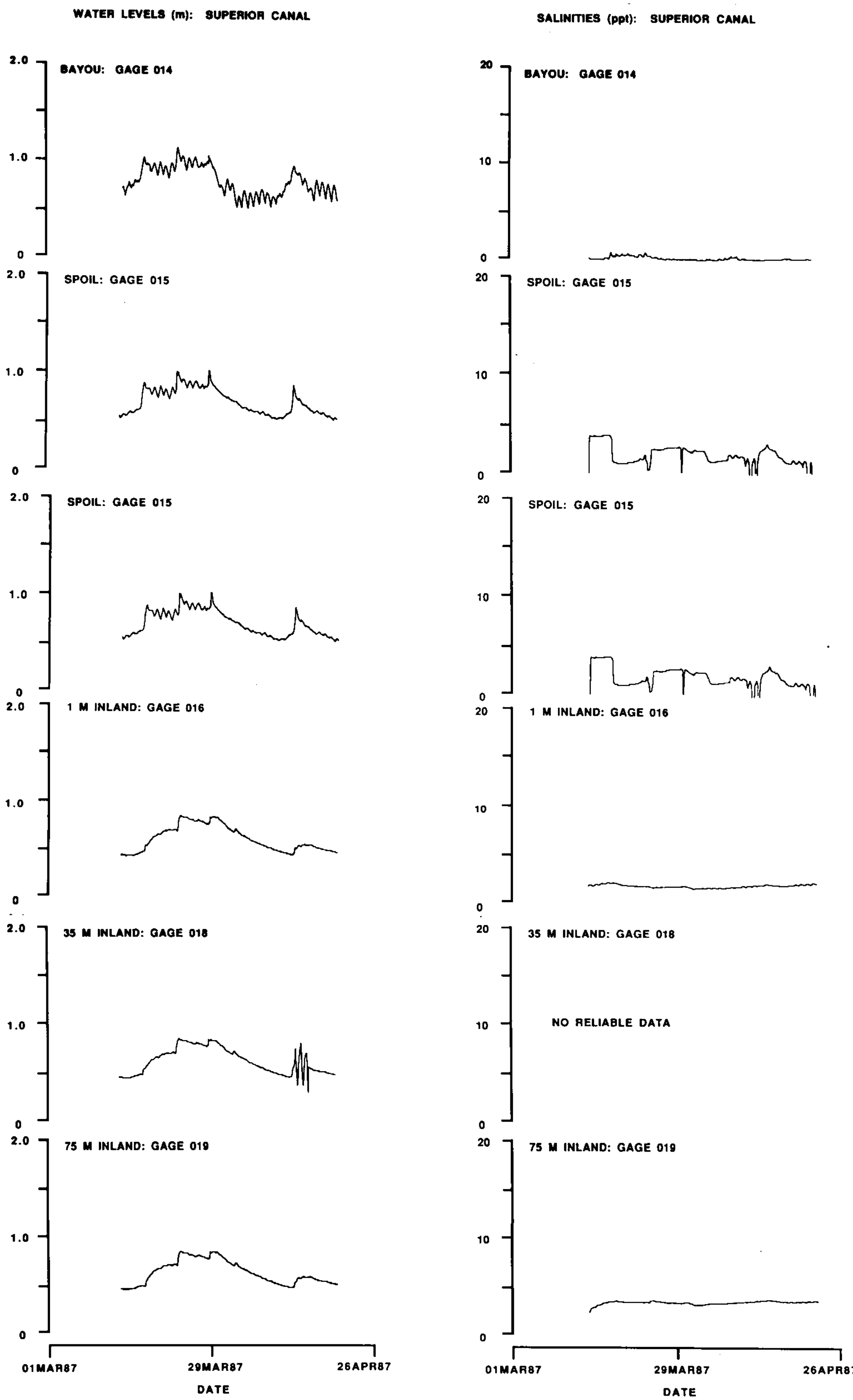


Figure 7-9. Time series plots of water levels, in meters (left) and salinities, in PPT (right) from the gage deployments at Superior Canal. The figures are stacked as follows (top to bottom): gage 014 (bayou), gage 015 (spoil bank), gage 016 (1 m behind spoil bank), gage 017 (5 m behind spoil bank), gage 018 (35 m behind spoil bank) and gage 019 (75 m behind spoil bank).

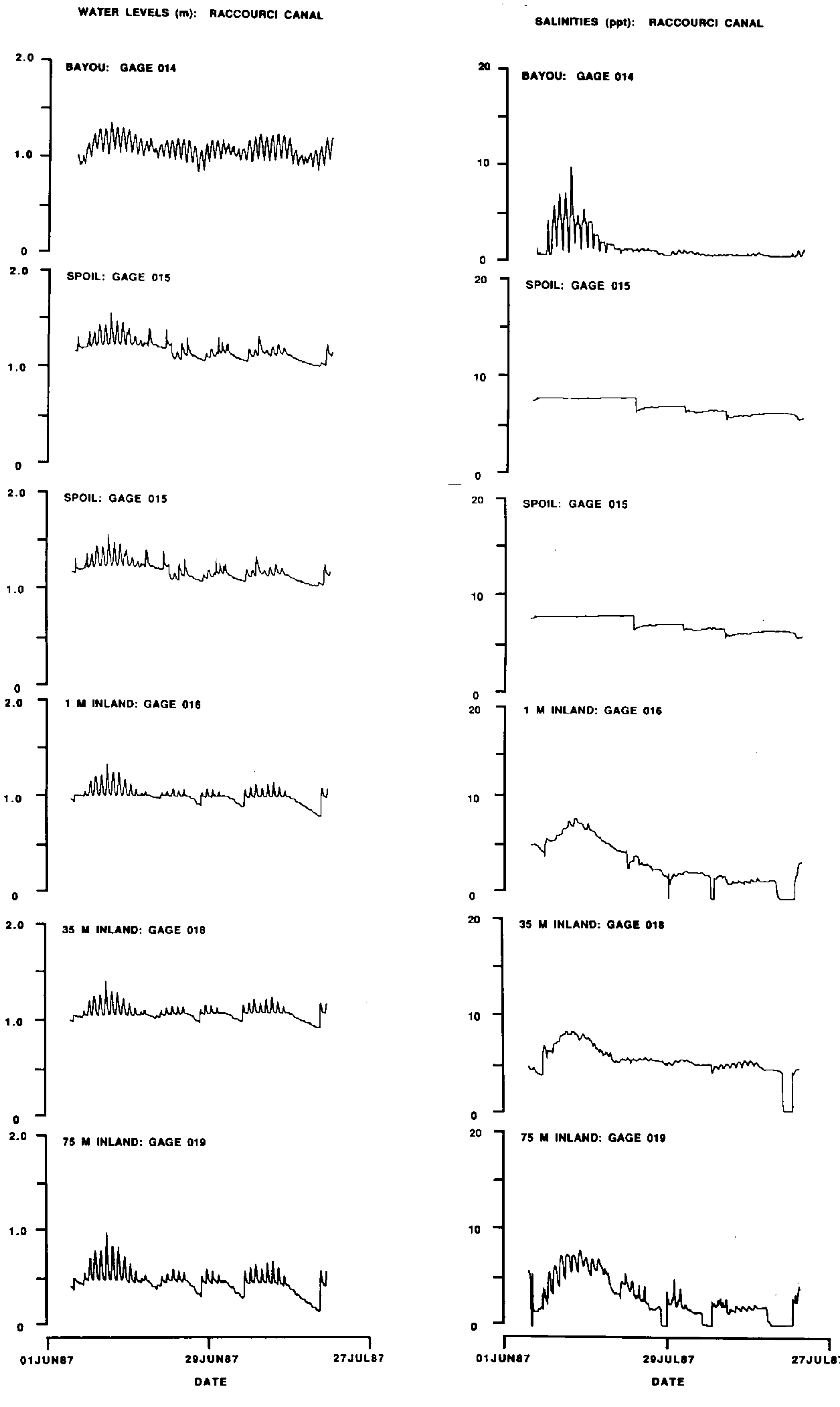


Figure 7-10. Time series plots of water levels, in meters (left) and salinities, in PPT (right) from the gage deployments at Raccourci Canal. The figures are stacked as follows (top to bottom): gage 014 (bayou), gage 015 (spoil bank), gage 016 (1 m behind spoil bank), gage 017 (5 m behind spoil bank), gage 018 (35 m behind spoil bank) and gage 019 (75 m behind spoil bank).

peak at 0.024 cycles/hour (cph), for example, would not be given much credence, while one at 0.040 cph, the diurnal band, would be accepted as real and forced by the astronomical tide. When two or more neighboring estimates were significant, the coherence within the band was accepted as real.

Field Experiment 1. The energy spectra for both the salinity and pressure signals at all stations generally increase as frequency decreases. Other than the low frequency peak, the dominant energy peak is at the diurnal frequency. This latter peak, though, is reduced significantly in the salinity signals from station 15 and absent at station 16. Bayou pressure is generally coherent with the pressure signal at the other stations at frequencies lower than 0.024 cph and at the diurnal frequency. Beyond the natural levee, coherence with the bayou increases with distance into the marsh. The pressure signals are highly coherent among themselves at the three stations in the marsh proper.

Low frequency water level variability normally has a large spatial scale. We expected the pressure signal to be strongly coherent at low frequencies, and this is the case between the bayou water level and the pressure signals within the marsh. Furthermore, there is direct transmission of the bayou signal through tidal creeks, i.e, natural breaks in the levee. In fact, the gages farthest back in the marsh showed the strongest coherence with the bayou signal. (This, though, still does not prove that the observed signal is uniquely caused by local forcing rather than from a distant region, such as the back of the marsh, as will be discussed below). What is more difficult to explain is the lack of coherence between the bayou and the two stations on the natural levee. The picture that begins to emerge is of a large-scale water level signal, coherent between the water bodies and the marsh, that only slowly penetrates the levee itself. The transmission path between the bayou and marsh is, presumably, topographic lows in the natural levee.

The salinity in the bayou is not clearly coherent with the two reconstructed records from the levee. Within the bayou, the salinity is coherent with the water level at frequencies below 0.024 cph and at the diurnal frequency as one would expect in a system dominated by advection. Finally, the coherence between the bayou water level and salinity at the two levee stations is significant only at the very lowest frequencies.

It should be remembered that the signals in the natural levee (or the spoil bank in subsequent experiments) are intermittent, falling to zero when water level drops below the sensor level. If the analyses are interpreted as describing continuous signals, then the apparent breaks in signal continuity impose a very special spectral window on the data. This spectral window is broad and greatly smears the frequency resolution.

Field Experiment 2. During the second deployment of the instruments, spectra of the recorded signals again rose toward low frequencies. All signals exhibited a weaker secondary peak near the diurnal band, except the pressure measured at station 19.

Bayou water level was coherent with the water level signal at all other stations at frequencies less than 0.017 cph. The pressure variations at stations 16, 17, and 19 were coherent across the band of interest. Station 18, though, was coherent with stations 16 and 17 only below 0.030 cph and with station 19 only below 0.023 cph. While the marsh again appears to respond uniformly to external forcing, this forcing is not clearly local. Furthermore, the behavior within the marsh at station 18 is anomalous.

The salinity records at stations 19 and 16 exhibit a broad band of coherence between 0.023 and 0.042 cph. The salinity signals at stations 15, 16, and 19 consistently exhibit a coherence peak with the bayou water level at the diurnal frequency.

Field Experiment 3. All recorded signals exhibited spectra with both a low-frequency peak and a diurnal peak. Pressure signals at stations 16, 17, 18, and 19, within the marsh, were coherent at all frequencies. Station 15 was also coherent with the marsh pressures, except in a narrow band near 0.014 cph. The bayou water level, though, was coherent with the marsh pressures only at specific frequency bands: 0.005 cph, 0.024 cph, and 0.042 cph. Bayou salinity and water level are also coherent at these same frequency bands, suggesting that these bands combine to create the most energetic events in the observed signals.

The coherence analysis between salinity gages showed no consistent pattern except that stations 16 and 19 were coherent below frequencies of 0.024 cph. Similarly, the coherence between bayou water level and marsh salinity was dominated by isolated spikes of significant coherence at 0.024 or 0.042 cph, the bands suspected of representing energetic events in the bayou.

Field Experiment 4. The spectra of salinity and pressure rose at low frequencies at all stations. A diurnal peak was also present at all stations except in the salinity record at station 15, although the peak was decidedly weak in the salinity records at stations 16 and 18. Pressure signals were generally coherent across the frequency band of interest with a peak in coherence at the diurnal frequency. The strongest coherence was between stations within the marsh proper, as before. Salinity records were generally incoherent between stations, except at the lowest frequencies. Only station pairs 16 and 18, and 16 and 19, within the marsh, showed broad bands of significant coherence. No pattern between the pairs was apparent.

Within the canal, salinity and pressure were coherent below 0.009 cph. The bayou pressure was totally incoherent with the salinity signal on the spoil bank, and the coherence with the salinity signals farther back in the marsh was weak and spotty. Figure 7-11 presents examples of the spectral density estimates and the coherence-squared estimates for this site.

Discussion

Our experimental design was based on the concept of a quasi-linear, single-input (the adjacent bayou) system (Mason and Zimmerman, 1960). We anticipated that the salinity variations within the marsh were driven coherently by processes occurring in the adjacent channel system. The time series analysis described above does not bear out these assumptions. Had the bayou salinities been the source for the marsh salinities and the two been related through the hypothesized linear system, bayou salinities should be coherent with salinities in the marsh. A progressively greater time lag between the two signals should be observed as one moves back in the marsh. The strength of the signal might also be damped with distance from the source. Similar relationships should also hold for the pressure signals. Finally, marsh salinities should be coherent with bayou pressures that are the energy source to pump the salt water into the marsh. Such relationships are not consistently observed. The weak or insignificant coherence between signals in the channel proper and those in the marsh suggest that the signals are extremely noisy, the system is highly non-linear or there are multiple inputs to the system. The quality of the instrument calibrations, at least for the last two deployments, precludes instrument noise as a serious problem. Small-scale inhomogeneities in the marsh composition may have affected the transfer of signals to the marsh from the adjacent channel, but we have no evidence to suggest that this occurred. Also, the possibility of a highly non-linear transfer function is not a valid explanation for the observed statistics. If the system were single input but non-linear, we would expect the signals to propagate unidirectionally. The high salinity event recorded in late May and early June during the third deployment of the instrumentation

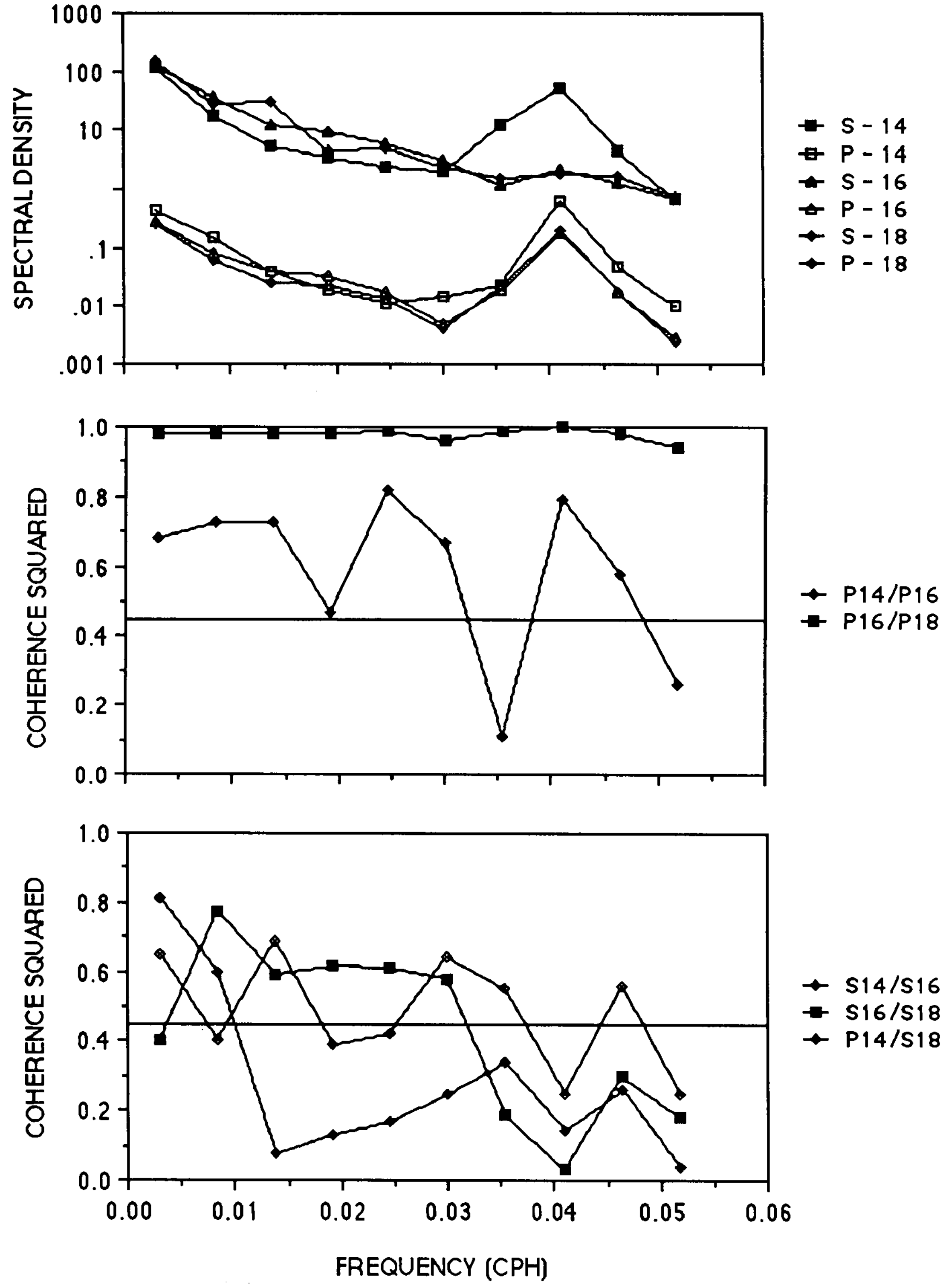


Figure 7-11. Examples of spectral density estimates and coherence-squared estimates from the experiment at RC. The horizontal line indicates the 95% significance level. In the legend, S indicates spectral density or coherence for salinity, P indicates spectral density or coherence for pressure, numbers refer to the particular gage (14 = bayou, 15 = berm, 16 = in between berm, 17 = 5 m behind berm, 18 = 35 m behind berm).

occurs later at station 18 than at station 19 or at the stations closer to the bayou. Similarly, tidal oscillations in the salinity signal at station 18 in early July, during the last deployment, are unrelated to any fluctuations of similar period at the other stations. It appears that multiple inputs and/or multiple pathways are forcing the system.

If the dominant signals were penetrating the marsh through the groundwater flow, we could expect that, unless the marsh soil permeability were a highly anisotropically distributed parameter, the nearest open water point would force the dominant response at a given point within the marsh. The response would be expected to decay with distance from the source in a reasonably homogeneous system.

Although hydraulic conductivities within the spoil banks and natural levees appear to be smaller than in the marsh proper, we have no reason to expect an extremely anisotropic distribution of hydraulic conductivity within the marsh soil. Drawdown experiments to estimate the hydraulic conductivity of the marsh soil resulted in numbers 5.0×10^{-5} cm/s for the spoil bank and natural levee and 9.0×10^{-4} cm/s for the marsh proper. We assume a pressure gradient on the order of 0.03 to be characteristic of normal conditions during a modest storm, tidal cycle or seasonal cycle in the absence of flooding. This represents a pressure head of approximately 30 cm across a levee or spoil bank 10 m wide. Using a characteristic value for the porosity of 0.75, we estimate the velocity of water flow within the marsh soil (not the discharge velocity) as, at most, 0.2×10^{-5} cm/s through the spoil bank or natural levee. A similar gradient within the marsh proper would result in a velocity of 3.6×10^{-5} cm/s. Clearly, on the time scales of our measurements, salt will not be advected great distances into the marsh through groundwater flow, although this may be an important process on much longer time scales.

The only other possible mechanism for extensive salt transfer to the marsh is overbank flooding. The large-scale topographic gradient of southern Louisiana is extremely small. This is the reason coastal flooding and the possibility of sea level rise is of such concern. Furthermore, the marsh surface, natural levee, and spoil banks for any region are not of uniform elevation. Thus, given a small area of marsh surrounded by bays, bayous, and canals, small-scale changes in the water level slopes may preferentially allow water to enter the marsh from different regions and flow through the marsh along channels created by the relative lows in the interior topography.

Variations in wind strength, direction, and spatial scale, as well as stream discharge, may generate small-scale water level gradients within the open water region of the coastal zone (bayous, bays, and canals). These are added to the larger scale, predictable gradients associated with tidal forcing. These water level changes of small spatial scale potentially are able to interact with the topographic variations of the marsh surface and its boundaries to allow water to enter the marsh and flow through it by a variety of different paths. This means that the source of overbank flooding that drives an observed salinity signal within the marsh may be locally or far-field driven (Figure 7-12)

We might suspect that the presence of spoil banks would alter the response of a marsh system to the mechanisms of saltwater influx that we propose. While the records we observed do not exhibit the extensive influence of any such effect, two points are important. The records we obtained were of relatively short duration. During the winter months, storm surges along this coast occur at 3 to 10 day intervals. During the summer this time scale lengthens up to three weeks. Our records covered only a few events which contribute a flooding signal at each site. Secondly, none of our marsh sites were significantly impounded.

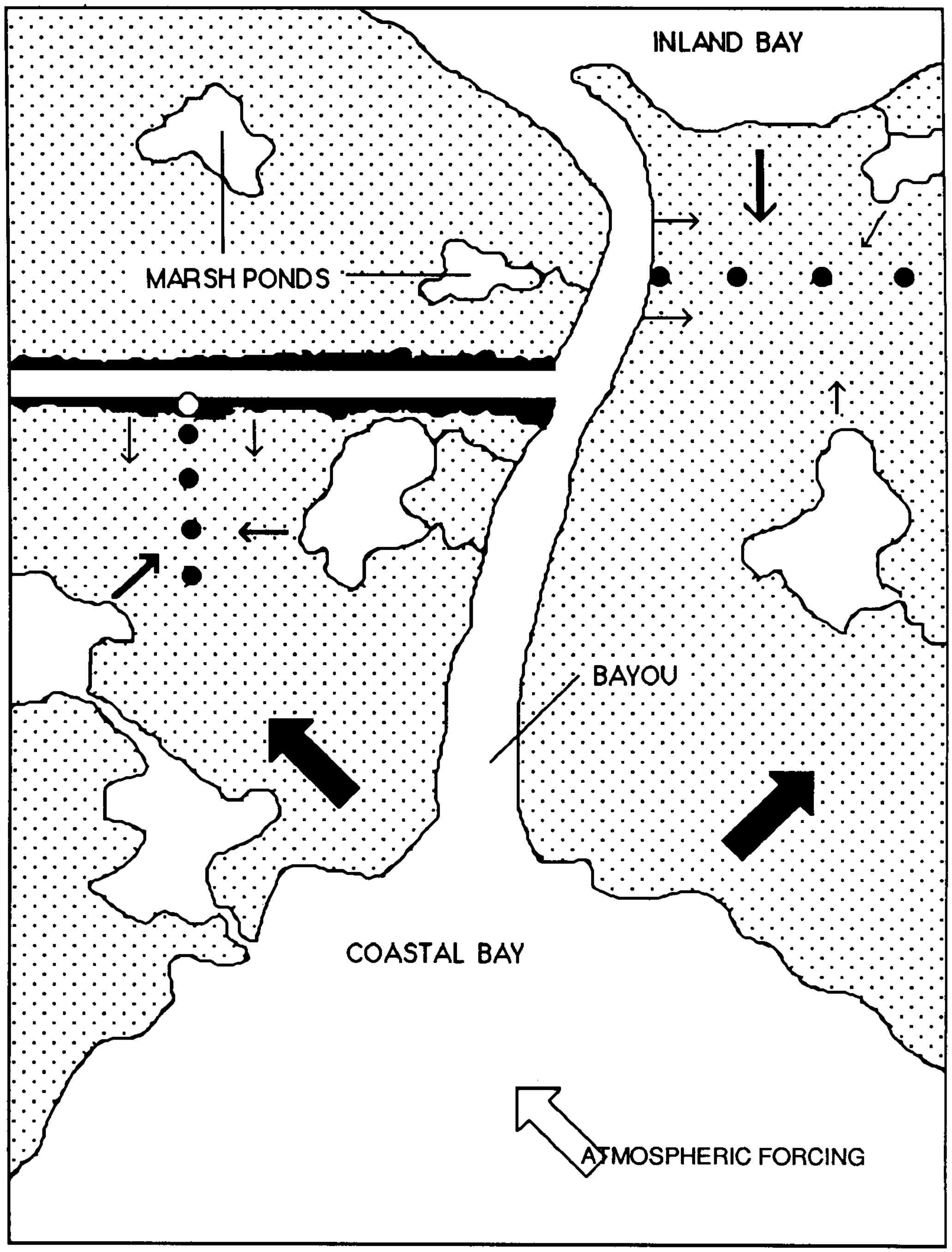


Figure 7-12. Sketch map of a marsh system showing the various pathways (arrows) through which salt can be advected to the gages along a transect (indicated by the circles). The size of the arrows indicate the relative importance of an individual pathway.

The results in Chapter 8 point out the progressively more negative effects of extending the time of exposure of marsh vegetation to elevated salinities. We expect that impounding an area will restrict the ability of saltwater to enter the marsh through over-bank flooding, but, once having entered the marsh, its ability to freely flow off the marsh will also be inhibited. Previous work by Swenson and Turner (1987) showed that in a marsh where 75% of the natural edges have been replaced by spoil banks, the hydrologic regime is altered. The partially impounded marsh had fewer but longer flooding events and fewer but longer drying events per month than did a marsh with natural edges along the surrounding bayous.

Conclusions

Although the data sets we were able to collect are short, we feel comfortable in drawing a number of definite conclusions.

(1) Lateral groundwater flow from the bayou to the adjacent marsh is of little importance on time scales of a month or less. We do not have the data available to assess its importance on seasonal time scales or longer.

(2) On time scales of days to a few weeks, overbank flooding is the dominant natural mechanism for salt to enter the marshes of south Louisiana.

(3) This mechanism of salt transfer to the marsh is not easily modelled as a single-input, linear system. The low topographic gradient of south Louisiana suggests that any given segment of marsh has multiple sources of water from overbank flooding and, possible multiple paths by which the water may arrive at a given site in the marsh.

(4) While other studies have demonstrated the potentially detrimental effects of spoil banks, such effects were not observed in this restricted study.

Chapter 8

EXPERIMENTAL FIELD AND GREENHOUSE VERIFICATION OF THE INFLUENCE OF SALTWATER INTRUSION AND SUBMERGENCE ON MARSH DETERIORATION: MECHANISMS OF ACTION

by

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Despite the popular notion that saltwater intrusion is a major factor causing wetland loss in the coastal zone of Louisiana, data supporting this hypothesis are not comprehensive and, to the best of our knowledge, have never been reported in the refereed literature. The gradual encroachment of saline water is thought to occur in Louisiana as the Mississippi River Deltaic Plain subsides and sea level rises (Morgan, 1977), and, indeed, vegetation maps indicate a northward movement in some areas of saline marsh types from 1968 to 1978 (Chabreck and Linscombe, 1978). Canals are believed to accelerate the penetration of salt water into brackish and fresh marshes that would not normally be subject to such a change in salinity. If saltwater intrusion occurs, particularly into a fresh marsh, the vegetation probably would be negatively affected. However, the questions of how and to what extent increases in salinity would affect the vegetation of the various marsh types that occur in coastal Louisiana have not been adequately investigated. Although the physiological tolerances of many fresh, intermediate, and brackish marsh plant species suggest that they would be affected adversely by relatively small increases in salinity, little experimental evidence exists that quantifies these effects.

Although increases in salinity due to saltwater intrusion into fresh marshes could result in vegetative dieback, brackish and salt marshes contain plant species that are adapted to growth in saline water. For example, *Spartina alterniflora,* the dominant intertidal salt marsh plant along the Gulf Coast, can not only tolerate, but also can grow vigorously (albeit at lower rates of productivity; Mendelssohn and Marcellus 1976) in salinities equal to that of full-strength sea-water along the Atlantic coast. Since salinity levels in the northern Gulf of Mexico are usually lower than that in the Atlantic, the extensive dieback and deterioration observed in the *S. alterniflora*-dominated salt marshes of Louisiana may not be caused by increased salinity. Thus, mechanisms other than saltwater intrusion must be considered. Land subsidence, in combination with eustatic sea level rise, may result in an increase in water level in addition to salinity increases. Stresses associated with an increase in depth and duration of flooding may potentially affect emergent macrophytes in all the major wetland habitats in Louisiana. Canalization may exacerbate this process through increased soil waterlogging (caused by impoundments or semi-impoundments created by intersecting canals) and sediment and nutrient deprivation in inland marsh areas (caused by the prevention of overland flow by spoil banks). Factors associated with increased waterlogging have been shown to reduce the growth of *S. alterniflora* in a Louisiana salt marsh (Mendelssohn and McKee, 1987). Although information of this type is generally lacking for fresh and brackish marsh species, the potential exists for a similar response to increased water levels in these habitats. In addition, the combination of salinity and waterlogging-related stresses could lead to a more rapid deterioration of these types of marshes than through either factor alone.

The major goal of this study was to investigate the relative effects of increased salinity and water levels on the dominant plant species in each of three major marsh types by simulating saltwater intrusion (increased salinity) and subsidence (increased depth and duration of flooding) under field and greenhouse conditions. The specific objectives were to determine:

(1) the impact of increased salinity on the dominant plant species in fresh, brackish, and salt marsh habitats;
(2) the impact of increased submergence (flooding) on the dominant plant species in fresh, brackish, and salt marsh habitats; and
(3) the relative importance of salinity and submergence in controlling the growth of each dominant plant species.

Materials and Methods

Field Experiments

The following design was used for the dominant plant species in each of three major marsh habitats: *Panicum hemitomon*–fresh marsh, *Spartina patens*–brackish marsh, and *Spartina alterniflora*–salt marsh. In each case, the simulation of subsidence, saltwater intrusion, or a combination of the two was initiated in appropriate sites (as described below).

Simulation of Subsidence. An increase in water level was accomplished by removing sections of marsh (0.1 m^2 surface area, 30 cm deep) and replacing them in their original locations, but at a lower elevation (-10 cm) (Figure 8-1). Disturbed controls, which were located adjacent to the subsidence treatment plots, were removed in exactly the same manner as the treatment plots, but were replaced in their original locations and elevations. Undisturbed controls were also established by marking off plots of the same size but which were not disturbed in any way.

Saltwater Intrusion Simulation. Since a consistent and homogeneous increase in soil salinity over a long period of time would be difficult and expensive to achieve in the field by additions of NaCl or salt water, an alternative approach was taken. Saltwater intrusion was simulated by removing sections of vegetated marsh (same as above) from the original marsh (hereafter referred to as the donor marsh) and transplanted to an area where the salinity was higher (hereafter designated as the recipient marsh) (Figure 8-1). In this way, a continuous input of water at the desired salinity level (within a specific range) was assured for the duration of the study. This design closely reproduced the effect which saltwater intrusion would have on the vegetation. We recognize that nutrient and sulfide concentrations may be different between donor and recipient marshes and may affect plant response such that the result may not be exactly the same as that which would occur during saltwater intrusion. However, saltwater intrusion would not only increase salinity, but would also increase sulfide concentrations due to higher sulfate levels in the more saline water. In addition, a higher mineral content in the soil substrate (which would be more likely in the recipient marsh caused by a greater sediment input) would tend to ameliorate the plant response to salinity stress by stimulating growth. Thus, this experiment probably represents a more conservative situation than would occur with saltwater intrusion. Treatment plots were removed with the intact vegetation and transported by boat to a recipient marsh where they were inserted into the substrate at three different elevations (Figure 8-1).

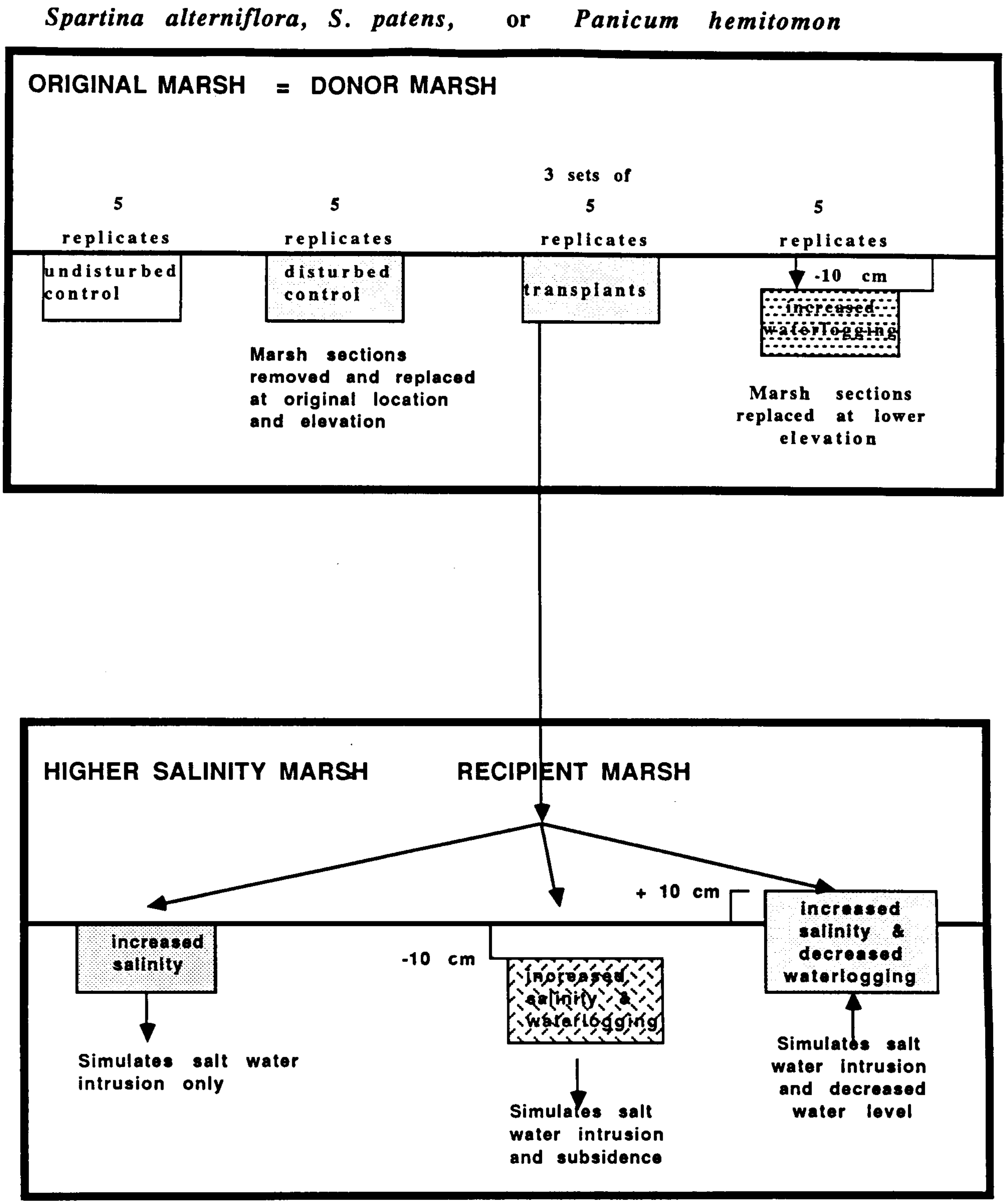


Figure 8-1. Experimental field design.

Salinity-Subsidence Interaction. Plots from the donor marshes were placed into the recipient marshes at three elevations: (1) equal to the marsh surface (simulated saltwater intrusion only); (2) 10 cm below the marsh surface (simulated saltwater intrusion and subsidence together); and, (3) 10 cm above the marsh surface (simulated saltwater intrusion but reduced the effect of submergence) (Figure 8-1). The latter treatment was established primarily to ensure that there was a salinity effect without an effect of increased submergence (due to the difference in surface elevation between the donor and recipient marshes). However, it also allowed the examination of plant response to increased salinity at reduced levels of flooding.

Vegetation surrounding the experimental plots in both donor and recipient marshes was removed by clipping to within 25 cm of the marsh surface to eliminate the influence of shading on the experimental plants. Vegetation within the treatment and control plots was trimmed to a height of 40 cm in order to eliminate dead material from the previous year and to stimulate new growth.

Study Sites. A fresh marsh site dominated by *P. hemitomon, Sagittaria lancifolia,* and *Leersia oryzoides* was selected as one of the donor marshes. This site was located at the southern end of Lac des Allemands (Figure 8-2). Plots containing a mixture of these three species were established here (subsidence treatment and controls) or transported to the recipient marsh (salinity and salinity-submergence treatments) which was located near Bayou Rigolettes (Figure 8-2). At the time of transplantation the salinity level of the donor marsh was 0 ppt and that of the recipient marsh was 6-7 ppt. The dominant vegetation at the recipient marsh was *Scirpus olneyi.*

A brackish marsh site dominated by *S. patens* was selected as another donor marsh. This site was established at the northern end of Bayou Mink near Leeville, Louisiana (Figure 8-2). Plots were transported to the recipient marsh located 30 km south near Airplane Lake (Figure 8-2). The salinity level of the donor marsh was 15 ppt and that of the recipient marsh was 21-23 ppt.

Another brackish marsh donor site dominated by *S. alterniflora* was located on Bayou Mink three km south of the *S. patens* donor site (Figure 8-2). Plots of *S. alterniflora* were transported to a recipient site adjacent to the *S. patens* transplants near Airplane Lake (Figure 8-2). The salinity level of the donor marsh was 13 ppt and that of the recipient marsh was 21-23 ppt.

The field plots were established during May 1986 and monitored over the succeeding months. The fresh marsh experiment was terminated in September 1986; the brackish and salt marsh experiments were terminated in October 1986.

Greenhouse Experiments

Experimental Design. The effect of salinity and submergence on the survival and growth of the fresh, brackish, and salt marsh species was further examined in a 3 x 5 factorial experiment in the greenhouse. Cores of marsh with intact vegetation were removed from each of the donor marshes and placed into 15-cm diameter x 15 cm height plastic pots containing drainage holes in the base. Cores from the fresh marsh contained a mixture of *P. hemitomon, S. lancifolia,* and *L. oryzoides.* Cores from the brackish and salt marshes contained monocultures of *S. patens* and *S. alterniflora,* respectively. Sixty pots of each vegetation type were brought to the greenhouse and placed randomly into five

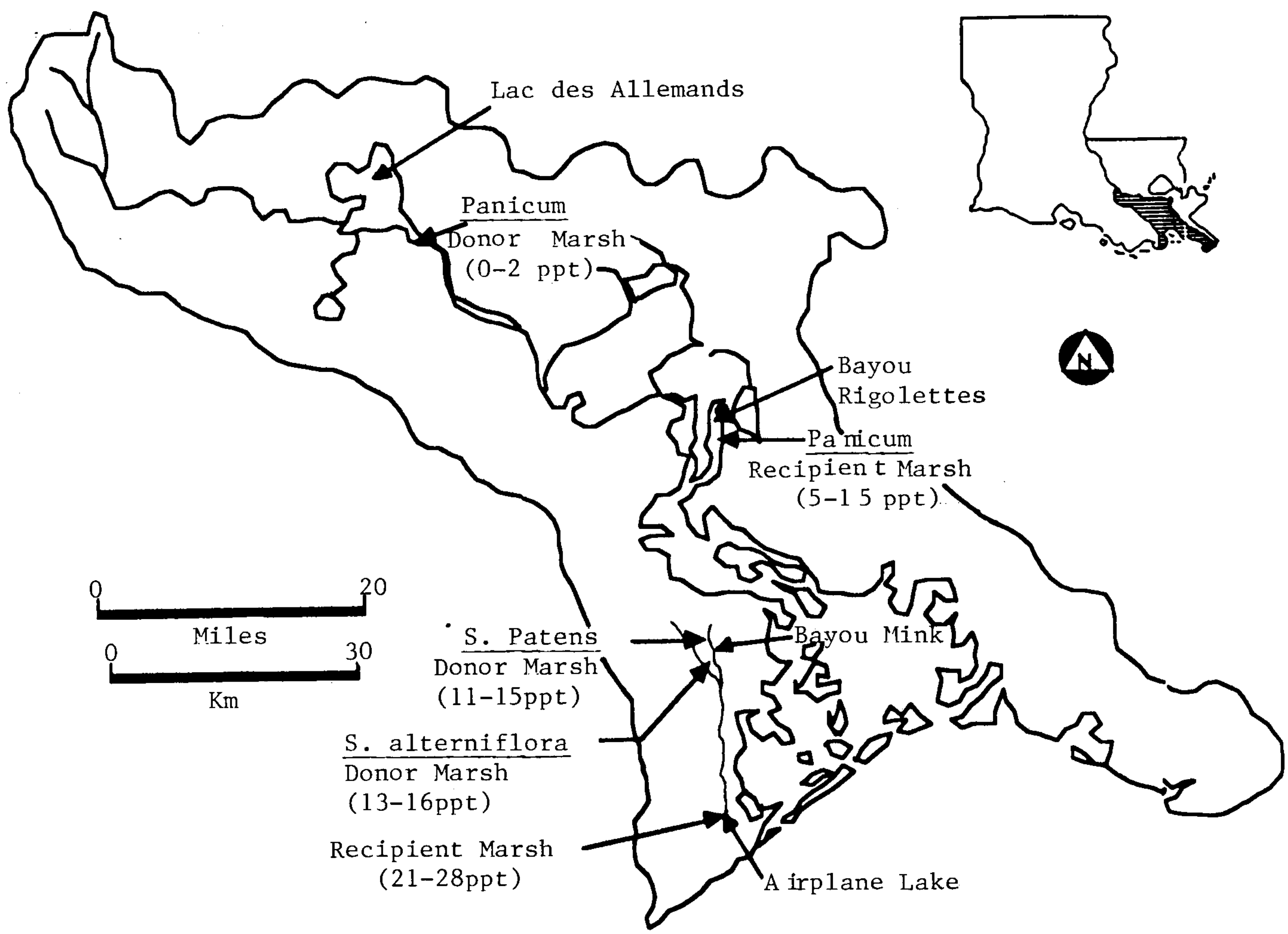


Figure 8-2. Map showing location of field sites. Salinity levels represent the range of values measured at each site during the course of the study and were determined at the following times: initial (April, 1986); interim (July, 1986); and, final (September–October, 1986).

Nalgene tanks as depicted in Figure 8-3. Experiments with each marsh type were conducted separately. Salinity levels in each tank were adjusted with Instant Ocean and ranged from 0 to 9.4 ppt for the fresh marsh plants, 0 to 28 ppt for *S. patens* and 0 to 35 ppt for *S. alterniflora.* Three flooding levels were created by placing the pots at different elevations within each experimental tank (Figure 8-3). The highest elevation was completely drained but in contact with the tank solution. These pots were flushed on a daily basis by lowering each pot into the tank solution for a few minutes; the substrate remained moist at all times. The second elevation was flooded to the soil surface; the lowest elevation was submerged to a depth of 10 cm above the soil surface (Figure 8-3). The latter two treatments were not disturbed, i.e. flushed, in any way during the experiment. A submersible pump was placed in each tank to ensure complete mixing of the salt solutions and to prevent stagnation. Salinity levels were monitored regularly and adjusted as needed with tapwater. The only sources of nutrients were those in the natural substrate and Instant Ocean. The duration of the experiment varied depending upon the response of each of the marsh types: 35 days for the fresh marsh species, 42 days for *S. patens,* and 115 days for *S. alterniflora.*

Analyses

Growth Measurements. Growth rate was monitored in the greenhouse experiments by measuring stem elongation in the fresh marsh species or leaf expansion in *S. patens* and *S. alterniflora.* Stem elongation was determined by measuring stem height of each of the three marsh species on a weekly basis. Leaf expansion was determined by placing a marker (small dot of a nontoxic silicone sealant) at the base of the youngest accessible leaf. As the leaf expanded, the distance between the dot and the leaf base lengthened. These measurements were conducted over a three-day period on a weekly basis. Biomass accumulation was used to measure treatment effects in the field experiments (see below for details).

Tissue Collection. Small amounts of leaf tissue from each sample unit (field plots or greenhouse pots) were collected for the measurement of an imino acid, proline, which accumulates in response to salinity stress (procedure described below). This tissue was quickly frozen in liquid nitrogen and stored in dry ice until transfer to a freeze dryer. All aboveground material in each sample unit was then clipped at the soil surface and placed into plastic bags. These samples were later analyzed for species composition, stem height, density, and live and dead biomass (after drying at 65° C). Brightened platinum electrodes were inserted into the soil of each sample unit at two depths (1 and 15 cm, field; 1 and 8 cm, greenhouse) and allowed to equilibrate for 1 hour prior to the measurement of soil redox potential. Eh was calculated by adding the potential of a standard calomel reference electrode (+244 mV) to the millivolt reading. Interstitial water was collected and analyzed as described below. A small sample of the viable plant roots was collected, washed of sediment and debris, and frozen in dry ice. These root samples were freeze-dried and later analyzed for the activity of the enzyme alcohol dehydrogenase (ADH) which increases in response to oxygen deficiencies (Mendelssohn et al., 1981). The remainder of the roots from the greenhouse experiments were washed and dried (65° C) for the determination of total belowground biomass.

Interstitial Water Analysis. Interstitial water was collected with an *in situ* water sampler as described in McKee et al. (in press). An aliquot of the water was added to an antioxidant buffer and later analyzed for sulfide concentration with a sulfide electrode (Lazar Model IS-146, Lazar Research Laboratories, Los Angeles, CA). The remaining

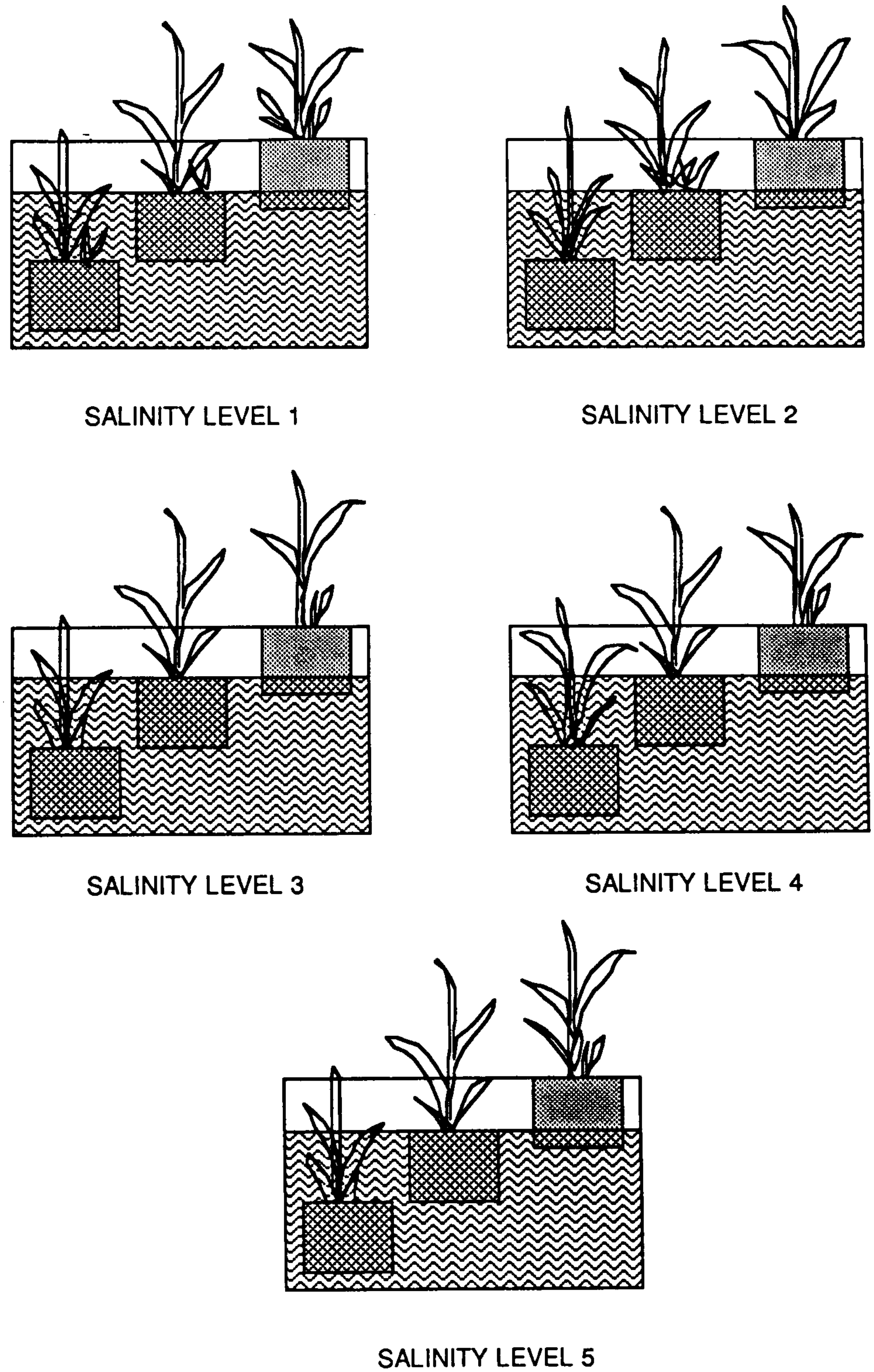


Figure 8-3. Experimental greenhouse design.

water was placed on ice and later used for the measurement of conductivity (Fisher conductivity meter, Model 152), pH, and NH_4 (EPA Method 353.2) (U.S. Environmental Protection Agency, 1979).

Plant Tissue Analyses. The oven-dried leaf tissue was ground in a Wiley Mill to pass a # 60 mesh sieve and analyzed for S, Na, K, Ca, Mg, P, Mn, Fe, Cu, and Zn (Soil Testing and Plant Analysis Laboratory, Athens, GA) and N (Perkin-Elmer C-N Analyzer, Model 240-C). The freeze-dried leaf tissue was also ground and analyzed for proline using the technique of Bates et al. (1973). The freeze-dried root samples were ground and analyzed for the activity of ADH, which is stimulated in plant roots during alcoholic fermentation (Mendelssohn et al. 1981). The degree of activity of this enzyme provided a measurement of the intensity of root oxygen deficiency and was used to quantify the degree of waterlogging stress each species experienced in the lower elevation treatment plots compared to the control plots. The activity of ADH was measured as described in McKee and Mendelssohn (1987).

Results

Field Experiments

Fresh Marsh Species. *Panicum hemitomon, Sagittaria lancifolia,* and *Leersia oryzoides* did not survive transplantation into a higher salinity marsh (Figure 8-4). Although an increase in salinity to approximately 5-7 ppt was planned, the salinity level in the recipient marsh (Figure 8-2) increased to 15 ppt following a major storm event which occurred after the experiment was initiated. Even though salinity levels subsequently returned to 5-7 ppt in July, the original species in the transplanted fresh marsh plots never recovered. However, the denuded plots in the recipient marsh were invaded by two other species, *Panicum dicotomiflorum* and *Pluchea camphorata,* which were common in this area late in the growing season.

At the donor marsh the physical disturbance of transplantation did not significantly affect density or biomass of the fresh marsh swards (Figure 8-4). The same was true for the brackish and salt marsh experiments discussed below. Density and biomass of *P. hemitomon* were significantly reduced (compared to the disturbed control) ($F = 14.23$ and 35.24, respectively, $p \leq 0.01$) by a decrease in elevation (Figure 8-4). Since this species accounted for the greatest proportion of the total biomass in the plots, its response dominated the effect of submergence on total biomass of the fresh marsh swards (Figure 8-4). Although *L. oryzoides* and *S. lancifolia* density and biomass appeared to be unaffected by elevation, the variability in frequency of occurrence of these species in the plots, as well as their relatively small contribution to overall biomass, prevents a definite evaluation of water level effects on their growth (Figure 8-4).

The soil redox potentials and sulfide concentrations indicated more reduced soil conditions in the lower elevation treatment compared to the controls (Table 8-1). *Panicum hemitomon* root ADH activity, however, was not increased significantly in the more waterlogged plants (Table 8-1). Soil salinity, pH, and NH_4 also were not significantly affected by a change in elevation although there was a trend for a decrease in NH_4 with an increase in elevation at the recipient marsh (Table 8-1, Figure 8-4). Leaf Mg, Zn, and Cu concentrations were reduced significantly in the plants growing at the lower elevation compared to the disturbed control ($F = 9.39$, 7.35, and 7.08, respectively, $p \leq 0.05$).

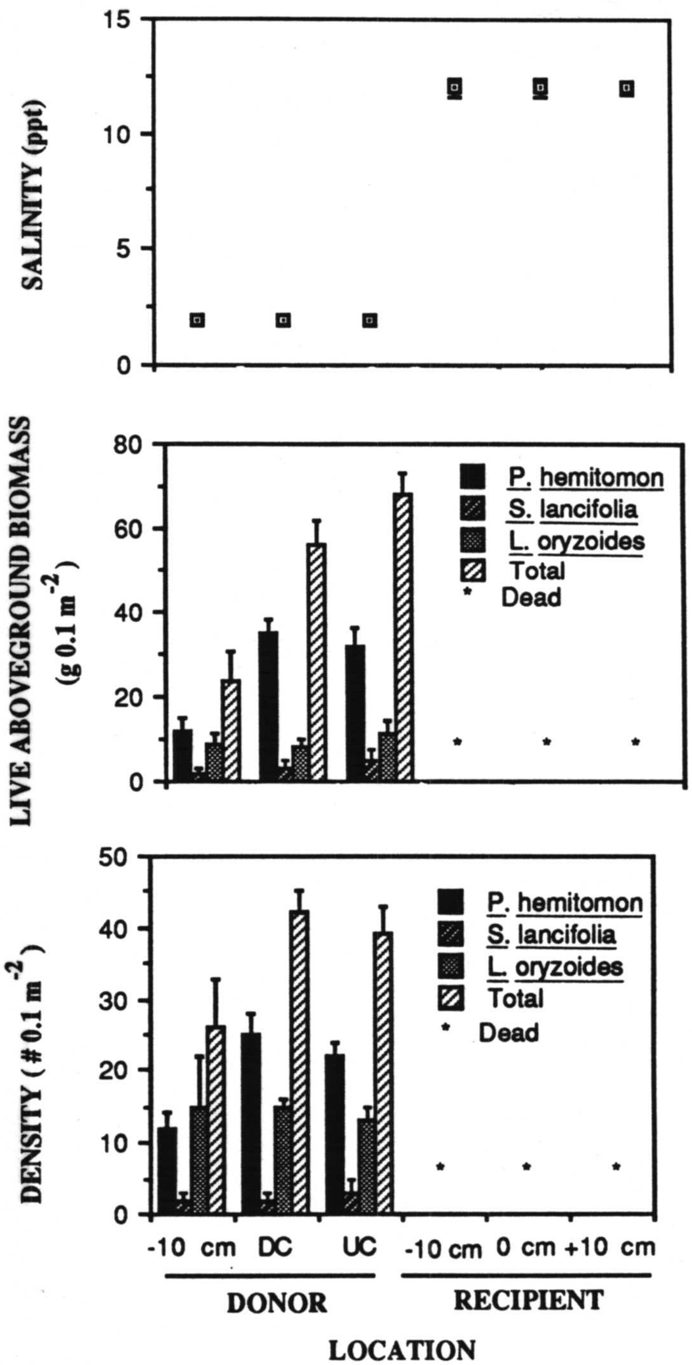


Figure 8-4. Interstitial water salinity, live aboveground biomass, and density of fresh marsh swards after six months growth at different elevations within two marshes of differing salinity. DC = disturbed control; UC = undisturbed control (n = 5). See text for further details.

Brackish Marsh Species. An increase in salinity from approximately 12 ppt to 26 ppt caused a significant reduction in height, biomass, and density of *S. patens* in the field ($F = 56.38, 93.39$, and 80.16, respectively, $p \leq 0.01$) (Figure 8-5). Although there may have been differences in nutrient concentrations between donor and recipient marsh soil, interstitial water concentrations indicated only minor differences in NH_4 between the control plots and those transplanted to the recipient marsh. Little, if any, difference was noted in pH and Eh (Table 8-2). Therefore, the primary difference between the donor and recipient marsh plots was salinity and salinity-related parameters (saltwater intrusion would not only bring in higher salt concentrations but also more sulfate, which, when reduced, would result in higher sulfide levels in the soil). The same was true for the salt marsh experiment discussed below (Table 8-3). Leaf proline concentrations were relatively low in all the field plots and indicated little accumulation in response to increased salinity (Table 8-2). An increase in salinity resulted in increases in tissue N, Na, Mg, B, Cu, and Zn ($F = 9.35, 5.12, 5.99, 64.58, 80.05, 35.01$, respectively, $p \leq 0.05$), but a decrease in K, Ca, and Mn ($F = 17.34, 11.71, 33.44$, respectively, $p \leq 0.01$).

Increased water level significantly reduced the density and biomass ($F = 31.47$ and 23.17, respectively, $p \leq 0.01$), but not height, of this species (Figure 8-5). Interstitial water sulfide concentrations were significantly higher in the lower elevation plots in both the donor and recipient marshes ($F = 19.92, p \leq 0.01$) (Table 8-2). There was also a significant negative correlation between soil sulfide and biomass at the donor ($R^2 = -0.46$) and recipient ($R^2 = -0.52$) marshes. Soil NH_4 concentrations were significantly higher in the more waterlogged plots ($F = 73.91, p \leq 0.01$), particularly in the recipient marsh (Table 8-2). There was a positive correlation between soil NH_4 and sulfide ($R^2 = 0.76$, donor; 0.64, recipient) and a negative correlation between soil NH_4 and biomass ($R^2 =- 0.47$, donor;- 0.63, recipient) and density ($R^2 = -0.32$, donor; -0.76, recipient). Root ADH activity was not affected significantly by a greater flooding depth (Table 8-2). An absence of sufficient viable roots in the -10 cm recipient marsh plots prevented the determination of ADH activity in this treatment (Table 8-2). Also, the poor condition of roots in the more waterlogged treatment at the donor marsh may have resulted in a reduced root enzyme activity, thus preventing the detection of an oxygen deficiency in the roots. Tissue nutrient concentrations in *S. patens*

Table 8-1. Soil redox potentials (Eh; mV) and interstitial water pH, sulfide (ppm) and NH_4 (ppm) concentrations, leaf proline concentrations (μmol g^{-1} dry wt), and root alcohol dehydrogenase (ADH) activity (μmol g^{-1} dry wt min^{-1}) measured in fresh marsh swards at two elevations (-10 cm depth and equal with marsh surface–disturbed control (DC) and undisturbed control (UC)) in the donor marsh and at three elevations in the recipient marsh after one growing season (n = 5).

	Donor Marsh			Recipient Marsh		
	-10 cm	DC	UC	-10 cm	0 cm	+10 cm
SOIL						
Eh (1cm)	+30± 48	+147± 15	+24 ± 12	+14 ± 50	+127 ± 39	+292 ± 42
Eh (10 cm)	-39± 6.0	+53± 20	+61± 18	-79± 26	+124 ± 40	+115 ± 30
pH	6.0± 0.1	6.0 ± 0.2	6.4 ± 0.1	6.3 ± 0.2	5.60 ± 0.3	5.7 ± 0.4
Sulfide	3.2 ± 0.0	< 0.3 ± 0	< 0.3 ± 0	6.4 ± 3.2	< 0.3 ± 0	< 0.3 ± 0
NH_4	0.6 ± 0.1	0.8 ± 0.1	0.5 ± 0.1	3.2 ± 1.3	1.1 ± 0.1	0.8 ± 0.4
PLANT						
proline	00.7 ± 00.1	00.5 ± 00.1	00.7 ± 00.2	a	-	-
ADH	72.0 ± 09.0	61.0 ± 07.0	41.0 ± 05.0	-	-	-

[a] No live tissue.

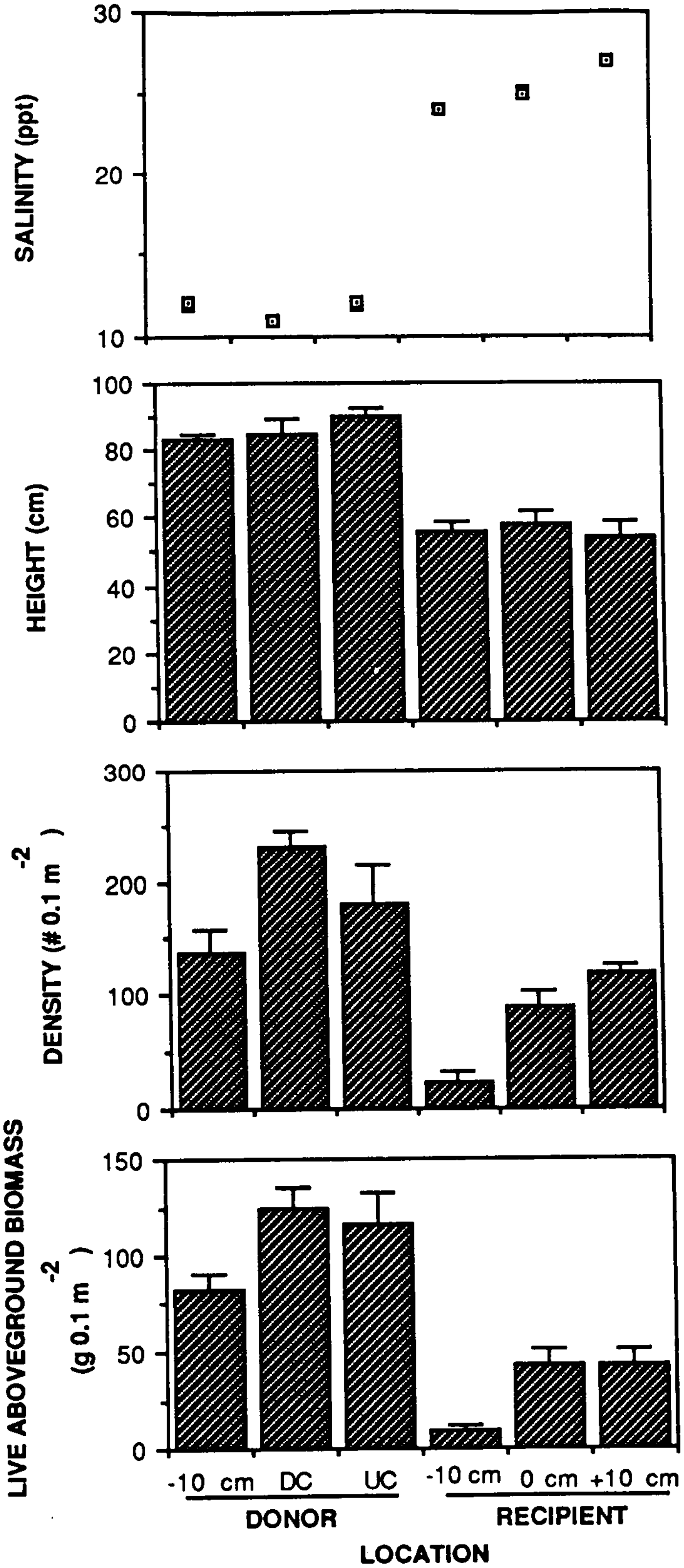


Figure 8-5. Interstitial water salinity, height, density, and live aboveground biomass of *Spartina patens* swards after six months of growth at different elevations within two marshes of differing salinity. DC = disturbed control; UC = undisturbed control (n=5).

Table 8-2. Soil redox potentials (Eh; mV) and interstitial water pH, sulfide (ppm) and NH_4 (ppm) concentrations, leaf proline concentrations (µmol g-1 dry wt), and root alcohol dehydrogenase (ADH) activity (µmol g-1 dry wt min-1) measured in *Spartina patens* swards at two elevations (-10 cm depth and equal with marsh surface-- disturbed control (DC) and undisturbed control (UC)) in the donor marsh and at three elevations in the recipient marsh after one growing season (n = 5).

	Donor Marsh			Recipient Marsh		
	-10 cm	DC	UC	-10 cm	0 cm	+10 cm
SOIL						
Eh (1 cm)	-5 ± 41	-58 ± 34	9 ± 46	-134 ± 21	38 ± 39	109 ± 34
Eh (10 cm)	-92 ± 42	-33 ± 38	-80 ± 29	-160 ± 7	-120 ± 16	-104 ± 34
pH	7.1 ± 0.1	7.1 ± 0.1	6.8 ± 0.0	7.1 ± 0.1	6.9 ± 0.3	6.5 ± 0.3
sulfide	10.9 ± 3.8	0.6 ± 0.3	0.3 ± 0.0	67.2 ± 19.5	0.3 ± 0.3	0.3 ± 0.3
NH_4	1.6 ± 0.34	0.6 ± 0.1	0.8 ± 0.2	8.3 ± 1.3	0.4 ± 0.1	0.5 ± 0.2
PLANT						
proline	2.7 ± 0.8	0.6 ± 0.2	1.5 ± 0.8	5.8 ± 1.3	1.0 ± 0.3	0.6 ± 0.1
ADH	43. ± 7	46 ± 6	40 ± 4	[a]	95 ± 20	42 ± 6

[a] Insufficient viable roots for analysis.

were affected significantly by increased waterlogging (Table 8-4). In general, N, S, Na, P, Fe, Al, and B concentrations were higher in plants growing at a lower elevation (F = 4.95, 6.95, 6.83, 27.31, 10.52, 5.01, 7.42, respectively, $p \leq 0.05$), whereas Ca and Mn were lower (F = 4.79 and 11.72, respectively, $p \leq 0.05$), primarily at the recipient marsh.

Live aboveground biomass was most reduced in the low elevation (-10 cm) treatment in the recipient marsh, indicating a greater effect of the combination of increased salinity and waterlogging on *S. patens* than either factor acting alone (Figure 8-5). However, the interaction between the two factors was not significant and their combined effect on biomass was additive. A significant interaction between waterlogging and salinity was found for leaf S concentration (F = 12.90, $p \leq 0.01$). When the elevation of the transplanted plots was increased to 10 cm above the recipient marsh surface, height, biomass, and density were not significantly increased above that of plots placed equivalent to the ambient elevation (Figure 8-5).

Salt Marsh Species. The transplantation of *S. alterniflora* from a salinity level of approximately 16 to 27 ppt did not cause a significant change in height, density, or live aboveground biomass (Figure 8-6). Leaf proline concentrations remained at extremely low levels in all plots and were not affected significantly by a change in salinity (Table 8-3). Transplantation to the higher salinity marsh resulted in higher Fe, Mn, and Al (F = 22.10, 15.69, 24.40, respectively, $p \leq 0.01$) and lower Cu and Zn (F = 16.79 and 4.26, respectively $p \leq 0.05$) concentrations in the leaves of *S. alterniflora* . Leaf Fe concentrations were more than three times greater in the plants at the lowest elevation treatment at the recipient marsh compared to that in the disturbed controls at the donor marsh (Table 8-4). Leaf Na, N, and K concentrations were similar at both marsh sites.

A 10-cm lower elevation caused a significant reduction in height, density, and biomass at donor and recipient marshes (compared to the disturbed control) (F = 4.65, $p \leq 0.05$; 17.89, 28.84, $p \leq 0.01$, respectively) (Figure 8-6). Increased waterlogging had a significant effect on soil reduction at 1- and 15-cm depths and sulfide accumulation (F = 21.71, $p \leq 0.01$; 4.44, $p \leq 0.05$; 15.39, $p \leq 0.01$, respectively) (Table 8-3). There was a negative correlation between sulfide and biomass at the recipient marsh ($R^2 = -0.62$). Increased waterlogging also resulted in an increase in interstitial water NH_4 concentration in both donor and recipient marshes ($F = 13.34$, $p \leq 0.01$) (Table 8-3). NH_4 was negatively correlated with plant density ($R^2 = -0.74$, donor; -0.58, recipient) and biomass ($R^2 = -0.67$, donor; -0.77, recipient) and positively correlated with soil sulfide ($R^2 = 0.83$) at the recipient marsh. Root ADH activity was not affected significantly by increased waterlogging (Table 8-3). However, the unhealthy appearance of the roots collected from the -10 cm elevation plots may have been a factor in causing a relatively low enzyme activity. Increased soil waterlogging caused a significant decrease in leaf concentrations of Mn ($F = 16.79$, $p \leq 0.01$) and an increase in S ($F = 13.21$, $p \leq 0.01$).

Table 8-3. Soil redox potentials (Eh) (mV) and interstitial water pH, sulfide (ppm) and NH_4 (ppm) concentrations, leaf proline concentrations (μmol g-1 dry wt), and root alcohol dehydrogenase (ADH) activity (μmol g-1 dry wt min-1) measured in *Spartina alterniflora* swards at two elevations (-10 cm depth and equal with marsh surface– disturbed control (DC) and undisturbed control (UC)) in the donor marsh and at three elevations in the recipient marsh after one growing season (n = 5).

	Donor Marsh			Recipient Marsh		
	-10 cm	DC	UC	-10 cm	0 cm	+10 cm
Soil						
Eh (1 cm)	-163 ± 17	-111 ± 13	-127 ± 25	-156 ± 10	-93 ± 8	+184 ± 26
Eh (10 cm)	-168 ± 13	-142 ± 15	-163 ± 12	-177 ± 7	-159 ± 5	-129 ± 14
pH	7.5 ± 0.1	7.4 ± 0.1	7.3 ± 0.1	7.4 ± 0.1	7.2 ± 0.1	6.8 ± 0.1
sulfide	122 ± 13	80 ± 16	106 ± 19	125 ± 26	29 ± 3	3 ± 0
NH_4	12.8 ± 4.1	1.9 ± 0.3	3.1 ± 0.1	5.5 ± 1.6	0.3 ± 0.1	0.4 ± 0.1
Plant						
Proline	0.6 ± 0.1	0.5 ± 0.0	0.5 ± 0.0	1.8 ± 0.8	0.6 ± 0.1	0.9±0.2
ADH	68 ± 9	40 ± 7	58 ± 12	34 ± 2	43 ± 17	28 ± 12

Greenhouse Experiments

Fresh Marsh Species. Increased salinity, but not water level, significantly reduced the live aboveground biomass of the fresh marsh swards during one month's growth in their natural substrate in the greenhouse (Figure 8-7, Table 8-5). This result was primarily caused by changes in the growth rate of *S. lancifolia* which was significantly affected by salinity (Figure 8-8). Stem elongation measurements indicated that *P. hemitomon* and *L. oryzoides* were the least affected by increases in salinity, while *S. lancifolia* was most sensitive (Figure 8-8). Proline concentrations in *P. hemitomon* leaves were significantly increased at the 9.4 ppt salinity level (Figure 8-9). Comparable changes in proline occurred to a smaller extent in *L. oryzoides,* but were absent in *S. lancifolia* (Figure 8-9). *Panicum hemitomon* leaf Na, K, Mg, S, Zn, B, and Mn concentrations increased with increasing salinity (F = 26.73, 5.88, 8.16, 3.84, 6.24, 12.59, and 7.07, respectively, $p \leq 0.01$) (Table 8-6). No discernible pattern was evident for N, P, or Ca (Table 8-6).

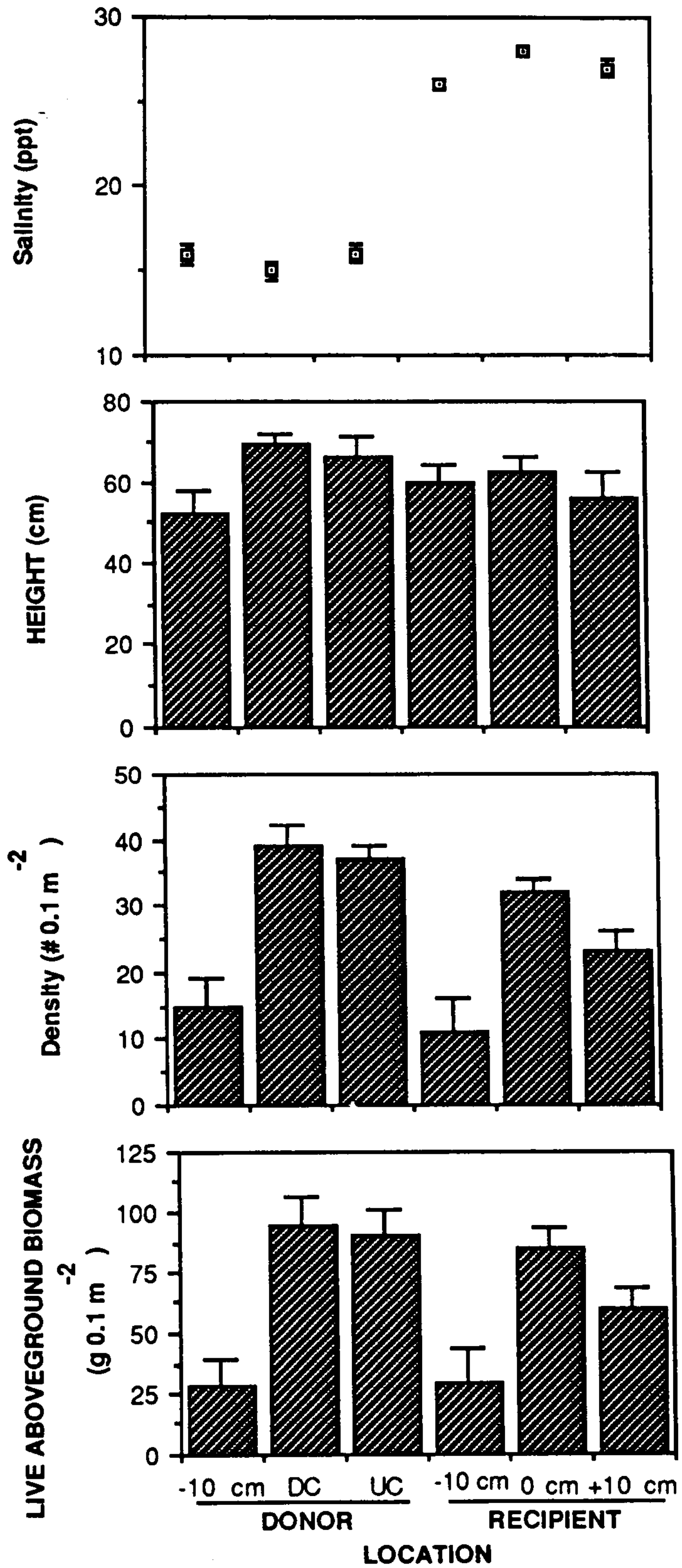


Figure 8-6. Interstitial water salinity, height, density, and live aboveground biomass of *Spartina alterniflora* swards after six months of growth at different elevations within two marshes of differing salinity. DC = disturbed control; UC = undisturbed control (n =5).

Table 8-4. Spectrographic analysis of leaf tissue collected from *Panicum hemitomon*, *Spartina patens*, and *Spartina alterniflora* swards at two elevations (-10 cm depth and equal with marsh surface–disturbed control (DC) in the donor marsh and at three elevations in the recipient marsh after one growing season. N, K, Ca, Mg, S, P values are in %; remaining values in ppm (n = 5).

	Donor Marsh			Recipient Marsh		
	-10 cm	DC	UC	-10 cm	0 cm	+10 cm
Panicum hemtomon						
N	2.17 ± 0.22	2.02 ± 0.10	2.12 ± 0.10	-[a]	-	-
K	0.7952 ± 0.1511	0.5837 ± 0.0666	0.5584 ± 0.0210	-	-	-
Ca	0.0673 ± 0.0096	0.0861 ± 0.0122	0.0813 ± 0.0058	-	-	-
Mg	0.0831 ± 0.0196	0.1602 ± 0.0157	0.1384 ± 0.0116	-	-	-
S	0.4040 ± 0.0240	0.3840 ± 0.0121	0.3940 ± 0.0136	-	-	-
P	0.1437 ± 0.0112	0.1323 ± 0.0119	0.1369 ± 0.0084	-	-	-
Na	69.15 ± 4.28	64.5 ± 5.93	59.02 ± 3.37	-	-	-
Fe	146.3 ± 16.9	124.9 ± 15.9	117.6 ± 8.1	-	-	-
Mn	100.4 ± 20.2	160.7 ± 23.3	152.7 ± 10.6	-	-	-
Al	33.41 ± 5.05	29.48 ± 3.92	26.80 ± 5.32	-	-	-
B	3.12 ± 0.50	5.32 ± 1.02	3.51 ± 0.21	-	-	-
Cu	5.11 ± 0.48	6.40 ± 0.06	6.81 ± 0.27	-	-	-
Zn	17.72 ± 2.37	26.43 ± 2.16	26.21 ± 1.19	-	-	-
Spartina patens						
N	1.16 ± 0.10	0.85 ± 0.02	0.90 ± 0.05	1.46 ± 0.09	1.21 ± 0.16	1.04 ± 0.09
K	0.5279 ± 0.0215	0.5466 ± 0.0445	0.5309 ± 0.0260	0.3946 ± 0.0145	0.4412 ± 0.0199	0.5427 ± 0.0311
Ca	0.1676 ± 0.0312	0.2539 ± 0.0418	0.2268 ± 0.0203	0.1105 ± 0.0042	0.1335 ± 0.0112	0.1312 ± 0.0193
Mg	0.1706 ± 0.0162	0.2089 ± 0.0245	0.1738 ± 0.0152	0.2402 ± 0.0072	0.2219 ± 0.0110	0.2042 ± 0.0226
S	0.2250 ± 0.0266	0.2600 ± 0.0294	0.2360 ± 0.0087	0.3940 ± 0.0329	0.2380 ± 0.0124	0.2000 ± 0.0180
P	0.1022 ± 0.0059	0.0836 ± 0.0039	0.0824 ± 0.0032	0.1071 ± 0.0040	0.0802 ± 0.0041	0.0800 ± 0.0049
Na	93.35 ± 9.80	67.70 ± 10.61	66.76 ± 5.47	110.7 ± 7.7	91.00 ± 7.78	82.41 ± 8.97
Fe	259.4 ± 25.8	162.4 ± 17.3	138.2 ± 6.3	250.7 ± 22.3	177.3 ± 34.9	142.5 ± 19.4
Mn	105.0 ± 12.6	168.8 ± 23.3	129.9 ± 13.0	42.79 ± 3.16	69.05 ± 8.07	68.72 ± 10.86
Al	191.1 ± 32.5	103.5 ± 28.0	105.6 ± 11.7	203.5 ± 33.3	143.6 ± 37.3	137.5 ± 24.1
B	6.76 ± 0.43	5.12 ± 0.33	5.71 ± 0.34	10.63 ± 0.57	9.79 ± 0.68	7.50 ± 0.62
Cu	3.67 ± 0.04	3.88 ± 0.20	4.96 ± 0.19	5.64 ± 0.27	5.71 ± 0.21	6.29 ± 0.44
Zn	7.01 ± 0.30	6.57 ± 0.26	7.77 ± 1.08	14.85 ± 1.59	12.38 ± 1.40	13.19 ± 0.53
Spartina alterniflora						
N	1.18 ± 0.11	1.11 ± 0.04	1.06 ± 0 .04	1.25 ± 0.09	1.18 ± 0.02	1.22 ± 0.05
K	1.0700 ± 0.0641	0.8829 ± 0.0873	1.1607 ± 0.0825	0.9489 ± 0.0941	1.0646 ± 0.1066	0.9057 ± 0.0830
Ca	0.1530 ± 0.0220	0.2544 ± 0.0927	0.2248 ± 0.0620	0.1739 ± 0.0447	0.2211 ± 0.0408	0.2622 ± 0.0394
Mg	0.3043 ± 0.0231	0.3450 ± 0.0585	0.3038 ± 0.0242	0.2845 ± 0.0273	0.2460 ± 0.0145	0.3054 ± 0.0260
S	0.6260 ± 0.0579	0.4475 ± 0.0668	0.6400 ± 0.0975	0.5800 ± 0.0870	0.3180 ± 0.0190	0.2780 ± 0.0107
P	0.0920 ± 0.0080	0.0648± 0.0078	0.0756 ± 0.0107	0.0887 ± 0.0084	0.0822 ± 0.0069	0.0714 ± 0.0044
Na	128.3 ± 3.2	121.8 ± 12.6	109.6 ± 9.8	141.3 ± 13.0	118.8 ± 8.9	116.9 ± 16.6
Fe	102.6 ± 8.6	117.9 ± 24.8	89.31 ± 14.03	404.5 ± 84.9	247.5 ± 11.3	166.9 ± 16.4
Mn	23.85 ± 2.18	43.44 ± 8.55	44.45 ± 2.71	47.10 ± 9.97	111.8 ±17.2	100.9 ± 17.8
Al	78.64 ± 11.65	97.13 ± 36.43	55.66 ± 3.80	451.0 ± 95.0	248.0 ± 21.3	157.9 ± 24.9
B	5.23 ± 0.15	7.68 ± 1.20	5.13 ± 0.31	9.31 ± 1.60	7.45 ± 0.85	8.21 ± 1.49
Cu	5.94 ± 0.38	5.56 ± 0.44	5.45 ± 0.25	4.62 ± 0.30	4.15 ± 0.19	3.83 ± 0.33
Zn	13.16 ± 1.26	15.2 ± 2.08	12.41 ± 1.37	11.19 ± 1.414	11.06 ± 1.12	17.79 ± 2.71

[a] No live tissue.

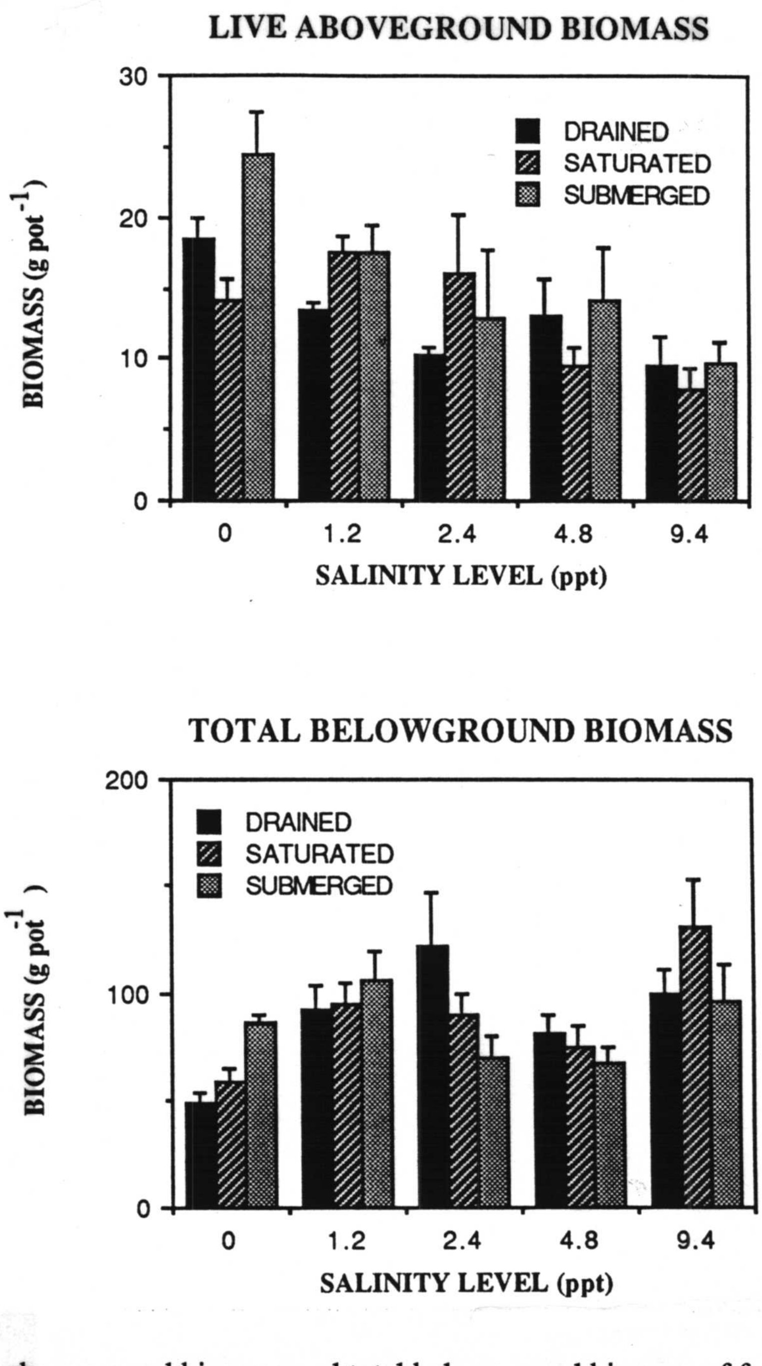


Figure 8-7. Live aboveground biomass and total belowground biomass of fresh marsh swards at five salinity levels and three water level depths in the greenhouse experiment for 35 days (n = 4). LSD $_{0.05}$ = 6 and 25 g pot^{-1} above and belowground, respectively.

Table 8-5. Soil redox potentials (Eh) and interstitial water pH, salinity, and sulfide measured in fresh marsh swards at three elevations and five salinity levels in the greenhouse (n = 4).

	Salinity Level (ppt)					
	0	1.2	2.4	4.8	9.4	$LSD_{.05}$
Eh at 1 cm (mV)						
Drained	197 ± 38	136 ± 40	48 ± 46	155 ± 89	254 ± 65	152
Saturated	-23 ± 41	10 ± 96	-119 ± 44	-23 ± 73	-89 ± 25	
Submerged	-34 ± 53	-15 ± 67	-47 ± 25	-127 ± 35	-132 ± 20	
Eh at 8 cm (mV)						
Drained	206 ± 109	146 ± 89	227 ± 66	137 ± 109	131 ± 100	175
Saturated	-8 ± 22	-43 ± 16	-83 ± 14	-83 ± 26	-78 ± 12	
Submerged	-75 ± 44	-31 ± 63	-29 ± 21	-92 ± 22	-130 ± 22	
pH						
Drained	7.3 ± 0.1	6.9 ± 0.2	6.9 ± 0.0	6.7 ± 0.1	6.8 ± 0.2	0.3
Saturated	7.2 ± 0.1	7.3 ± 0.0	7.3 ± 0.1	7.4 ± 0.0	7.5 ± 0.1	
Submerged	7.0 ± 0.0	7.3 ± 0.0	7.1 ± 0.0	7.4 ± 0.1	7.6 ± 0.1	
Salinity (ppt)[a]						
Drained	0.1 ± 0.1	2.4 ± 0.3	4.4 ± 0.2	7.5 ± 0.7	11.3 ± 0.3	0.4
Saturated	0.0 ± 0.0	1.1 ± 0.1	2.4 ± 0.1	5.4 ± 0.4	8.6 ± 0.1	
Submerged	0.0 ± 0.0	0.9 ± 0.0	2.2 ± 0.0	5.0 ± 0.1	8.6 ± 0.3	
Sulfide (ppm)						
Drained	< 0.01 ± 0.00	< 0.01 ± 0.00	< 0.01 ± 0.00	< 0.01 ± 0.00	0.08 ± 0.04	1.44
Saturated	0.40 ± 0.12	0.28 ± 0.04	0.64 ± 0.12	1.16 ± 0.50	4.32 ± 0.76	
Submerged	0.24 ± 0.02	0.24 ± 0.02	0.36 ± 0.10	2.38 ± 1.52	8.22 ± 1.00	

[a] Salinity values obtained at the end of this experiment were corrected for a malfunctioning conductivity electrode. Comparison with a new electrode revealed that the platinum black coating on the probe elements had been partially abraded (due to normal wear); this abrasion reduced the accuracy of the probe, particularly at higher salinity levels. Salinity values determined with the abraded probe could be corrected since a highly significant polynomial regression ($y = -0.299 + 1.067 x^2 + 0.057 x^2$, $R^2 = 0.99$) existed between values obtained with the two probes. This correction was applied only in this case since the new probe was used for the field study and the other greenhouse experiments.

The growth response of each species to salinity was modified by water depth (Figure 8-8). At the intermediate salinity levels tested, stem elongation of *S. lancifolia* was increased by increased waterlogging. Total belowground biomass varied with increasing salinity, but showed no coherent pattern (Figure 8-7). Because of the difficulty of separating roots of different species, it was not possible to determine the effect of salinity or waterlogging on the root biomass of individual species. The determination of root ADH activity also was not possible for the same reason. Leaf Ca, Mg, P, Fe, Mn, and Zn concentrations generally decreased with increased waterlogging, although in some cases there was an increase from the saturated to submerged treatment ($F = 13.01, 20.98, 15.84, 3.29, 10.46$, and 13.51, respectively, $p \leq 0.05$) (Table 8-6).

Brackish Marsh Species. Measurement of leaf expansion demonstrated that the growth of *S. patens* was affected significantly by increased salinity even though live aboveground biomass at harvest was not significantly different among salinity levels (Figure 8-10). The salinity level at which growth was significantly reduced depended upon the water level depth. In the most flooded treatment, leaf expansion was significantly reduced at the 20 ppt salinity level. However, in the saturated (flooded to the soil surface) and drained treatments, leaf expansion was significantly reduced at the 12 and 6 ppt levels, respectively. Proline accumulated in the leaves of plants growing in the most waterlogged

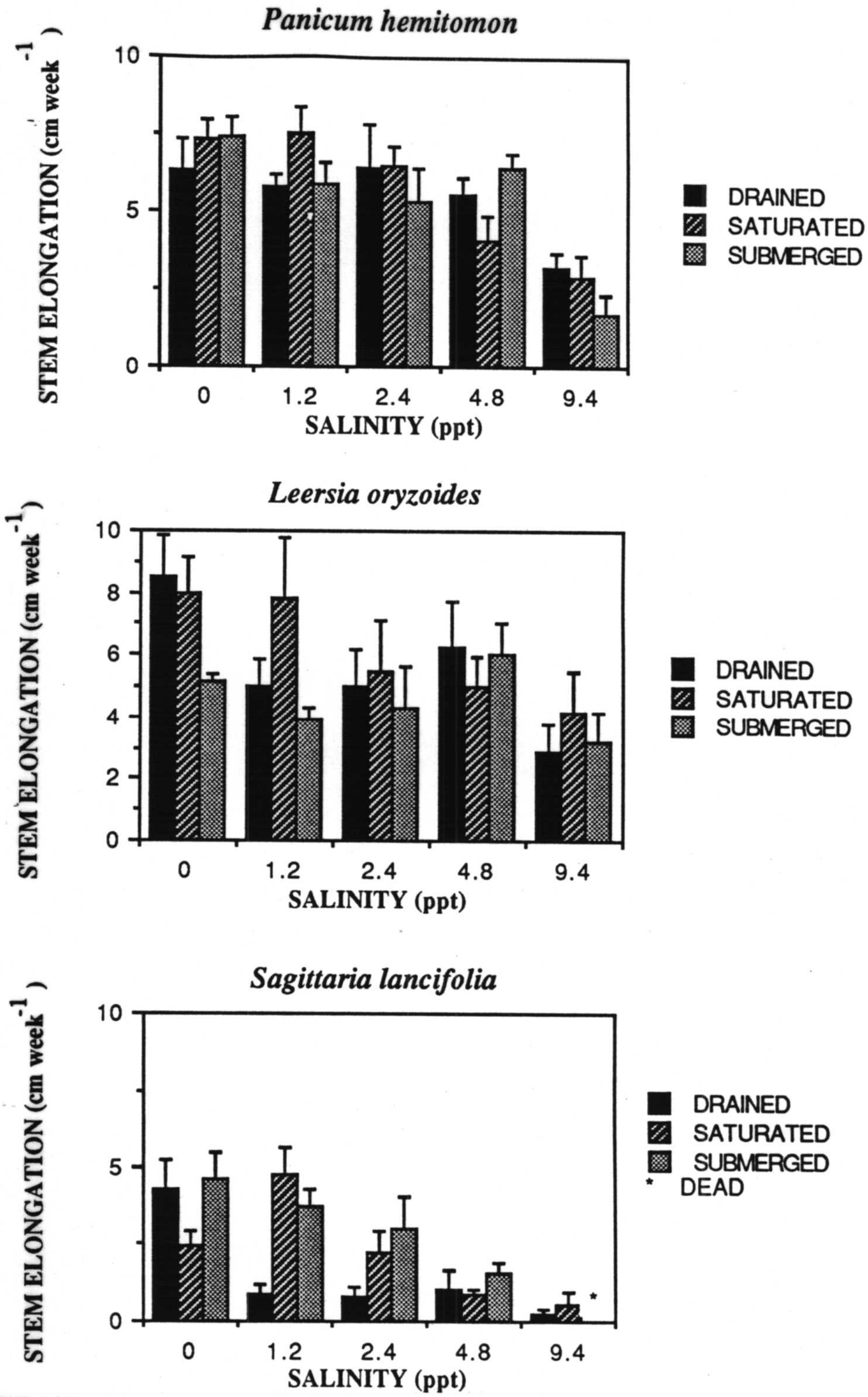


Figure 8-8. Stem elongation rate of three fresh marsh species during 35 days growth at five salinity levels and three water level depths in the greenhouse (n = 4). $LSD_{0.05}$ = 2.1, 3.4, and 1.6 cm week $^{-1}$ for *P. hemitomon, L. oryzoides,* and *S. lancifolia,* respectively.

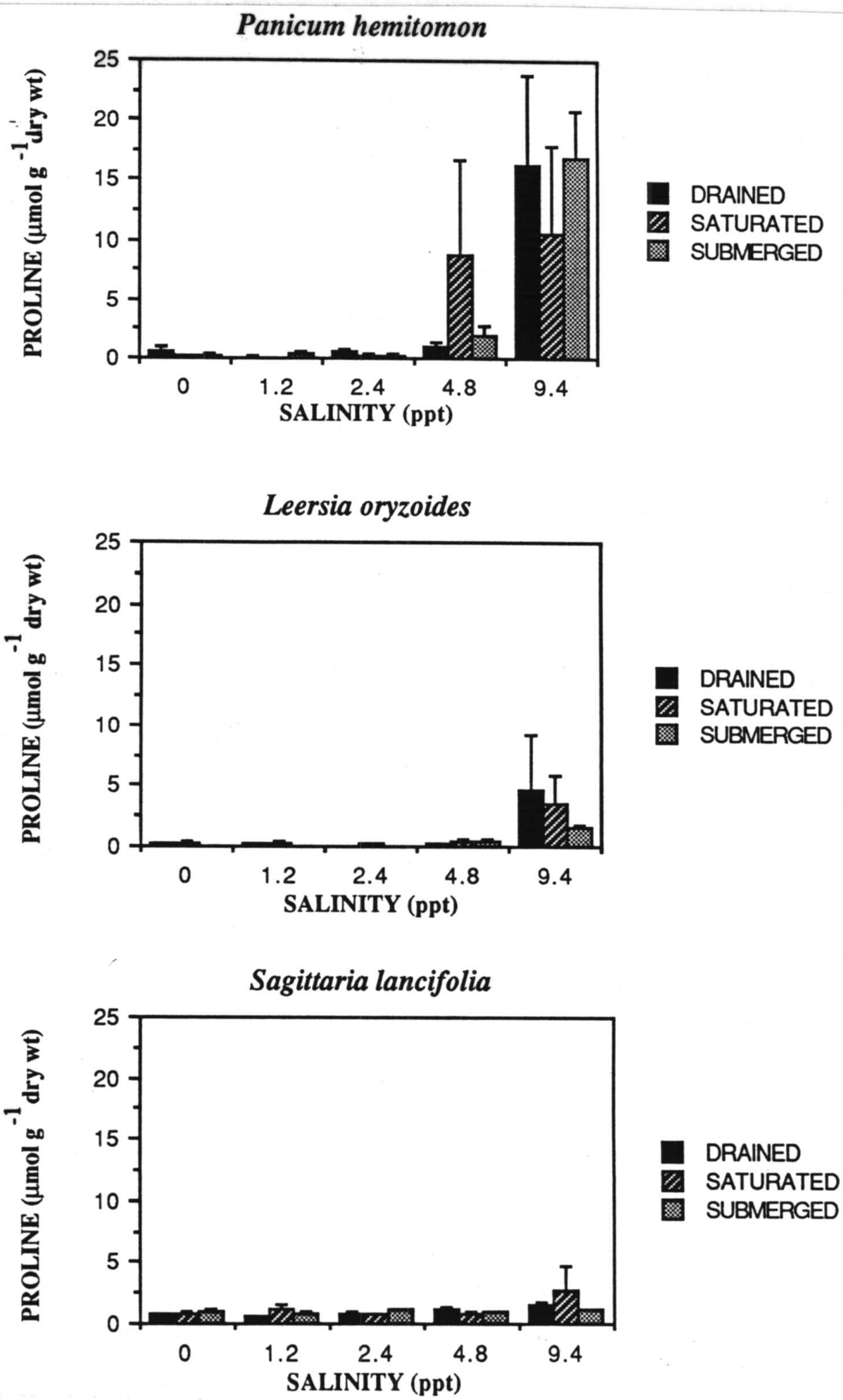


Figure 8-9. Leaf proline concentrations in three fresh marsh species during 35 days growth at five salinity levels and three water level depths in the greenhouse (n = 4). $LSD_{0.05}$ = 10.1, 4.3, and 0.4 µmol g^{-1} dry wt for *P. hemitomon, L. oryzoides,* and *S. lancifolia,* respectively.

Table 8-6. Spectrographic analysis of leaf tissue collected from *Panicum hemitomon* swards grown at five salinity levels in the greenhouse. N, K, Ca, Mg, S, P values are in %; remaining values in ppm (n = 4).

	Salinity Level (ppt)				
	0.0	1.2	2.4	4.8	9.4
Drained					
N	1.74 ± 0.07	1.75 ± 0.16	1.80 ± 0.04	2.16 ± 0.18	1.75 ± 0.10
K	0.6415 ± 0.1088	0.6044 ± 0.0636	0.6502 ± 0.1420	0.7001 ± 0.0629	0.8872 ± 0.1442
Ca	0.0909 ± 0.0090	0.1147 ± 0.0085	0.0829 ± 0.0150	0.0830 ± 0.0050	0.1198 ± 0.0311
Mg	0.1679 ± 0.0243	0.1963 ± 0.0120	0.1363 ± 0.0391	0.1971 ± 0.0135	0.2463 ± 0.0586
S	0.2900 ± 0.0379	0.3525 ± 0.0307	0.3033 ± 0.0186	0.3367 ± 0.0433	0.3725 ± 0.0852
P	0.1365 ± 0.0104	0.1959 ± 0.0117	0.1307 ± 0.0221	0.1485 ± 0.0167	0.1503 ± 0.0217
Na	51.66 ± 10.02	79.25 ± 2.41	73.42 ± 26.20	99.74 ± 11.07	172.8 ± 39.7
Fe	89.5 ± 14.9	86.03 ± 18.29	48.59 ± 3.49	102.0 ± 26.7	82.50 ± 12.50
Mn	90.95 ± 20.41	70.99 ± 9.79	47.96 ± 13.04	91.05 ± 8.36	162.1 ± 44.1
Al	20.96 ± 2.13	31.62 ± 11.94	12.61 ± 5.21	9.84 ± 2.90	6.62 ± 6.62
B	21.85 ± 8.29	28.27 ± 4.27	17.38 ± 5.14	44.43 ± 10.59	49.03 ± 13.72
Cu	3.849 ± 0.280	2.97 ± 0.20	2.94 ± 0.27	6.17 ± 1.08	5.50 ± 1.33
Zn	16.71 ± 1.73	11.76 ± 0.72	12.43 ± 2.04	22.81 ± 1.80	24.40 ± 4.35
Saturated					
N	1.51 ± 0.12	1.45 ± 0.15	1.33 ± 0.08	1.55 ± 0.13	1.59 ± 0.06
K	0.5678 ± 0.1219	0.4441 ± 0.0317	0.5241 ± 0.0501	0.7057 ± 0.0938	0.6436 ± 0.1732
Ca	0.0493 ± 0.0074	0.0637 ± 0.0053	0.0590 ± 0.0173	0.0532 ± 0.0044	0.0715 ± 0.0210
Mg	0.0756 ± 0.0183	0.0520 ± 0.0157	0.0727 ± 0.0170	0.1414 ± 0.0140	0.1431 ± 0.0440
S	0.3100 ± 0.0153	0.2300 ± 0.0303	0.2533 ± 0.0470	0.3433 ± 0.0376	0.3967 ± 0.0484
P	0.1462 ± 0.0258	0.1186 ± 0.0085	0.1051 ± 0.0102	0.1035 ± 0.0041	0.1025 ± 0.0321
Na	62.43 ± 5.20	74.17 ± 17.91	97.64 ± 16.63	177.1 ± 9.5	205.2 ± 24.9
Fe	50.28 ± 5.96	67.09 ± 16.21	84.60 ± 10.45	64.28 ± 2.59	60.72 ± 8.38
Mn	35.45 ± 7.07	30.29 ± 6.78	36.19 ± 14.02	55.09 ± 10.53	69.91 ± 15.58
Al	18.21 ± 13.56	16.24 ± 5.58	22.26 ± 6.78	6.74 ± 3.07	9.27 ± 6.71
B	9.87 ± 3.51	12.12 ± 4.81	21.16 ± 10.64	63.75 ± 6.49	33.94 ± 13.30
Cu	2.80 ± 1.41	3.05 ± 0.67	5.74 ± 4.17	4.08 ± 0.65	3.03 ± 0.94
Zn	10.12 ± 2.77	10.11 ± 1.54	11.59 ± 3.26	13.02 ± 1.58	12.11 ± 1.21
Submerged					
N	1.71 ± 0.14	1.43 ± 0.22	1.62 ± 0.21	1.43 ± 0.12	1.49 ± 0.18
K	0.5545 ± 0.0741	0.4540 ± 0.0317	0.5412 ± 0.0417	0.6824 ± 0.0252	0.8606 ± 0.0554
Ca	0.0568 ± 0.0058	0.0456 ± 0.0065	0.0806 ± 0.0129	0.0444 ± 0.0114	0.0933 ± 0.0119
Mg	0.0467 ± 0.0081	0.0406 ± 0.0074	0.1054 ± 0.0216	0.0943 ± 0.0264	0.1990 ± 0.0226
S	0.2275 ± 0.0048	0.2325 ± 0.0293	0.2233 ± 0.0233	0.3000 ± 0.0274	0.3300 ± 0.0268
P	0.0951 ± 0.0074	0.0977 ± 0.0066	0.0887 ± 0.0171	0.1040 ± 0.0070	0.1280 ± 0.0124
Na	49.13 ± 3.86	59.80 ± 7.10	80.25 ± 19.68	131.9 ± 20.9	190.8 ± 15.3
Fe	79.25 ± 10.63	57.36 ± 4.51	100.6 ± 25.4	119.9 ± 29.6	101.7 ± 8.3
Mn	44.00 ± 9.42	27.15 ± 2.42	77.15 ± 14.36	50.74 ± 11.88	97.12 ± 19.54
Al	17.82 ± 1.00	20.03 ± 6.02	39.80 ± 14.61	16.60 ± 6.65	32.26 ± 7.16
B	8.92 ± 1.24	7.97 ± 1.50	26.02 ± 7.59	41.83 ± 5.88	47.87 ± 12.03
Cu	2.09 ± 0.17	2.71 ± 0.14	3.25 ± 0.46	3.32 ± 0.34	3.88 ± 0.63
Zn	8.95 ± 2.19	8.04 ± 0.36	13.65 ± 3.43	11.00 ± 0.99	15.94 ± 1.48

Table 8-7. Soil redox potentials (Eh) and interstitial water pH, salinity, NH_4, and sulfide measured in *Spartina patens* swards at three elevations and five salinity levels in the greenhouse (n = 4).

	Salinity Level (ppt)					
	0	1.2	2.4	4.8	9.4	$LSD_{.05}$
Eh at 1 cm (mV)						
Drained	238 ± 58	192 ± 71	211 ± 44	170 ± 51	296 ± 86	127
Saturated	-127 ± 3	-133 ± 59	-108 ± 45	-115 ± 45	-167 ± 43	
Submerged	-169 ± 6	-186 ± 7	-200 ± 15	-229 ± 8	-220 ± 14	
Eh at 8 cm (mV)						
Drained	29 ± 118	-31 ± 41	88 ± 46	179 ± 56	77 ± 92	142
Saturated	-100 ± 18	-161 ± 21	-150 ± 22	0 ± 55	-214 ± 61	
Submerged	-190 ± 8	-168 ± 11	-200 ± 16	-230 ± 8	-198 ± 7	
pH						
Drained	7.2 ± 0.1	7.1 ± 0.1	7.1 ± 0.1	7.1 ± 0.2	7.5 ± 0.1	0.3
Saturated	7.2 ± 0.1	6.9 ± 0.1	7.0 ± 0.0	7.0 ± 0.1	7.5 ± 0.1	
Submerged	6.9 ± 0.1	6.8 ± 0.1	6.8 ± 0.1	6.9 ± 0.0	7.3 ± 0.2	
Salinity (ppt)						
Drained	1.8 ± 0.1	6.4 ± 0.2	12.5 ± 0.3	22.5 ± 1.2	28.8 ± 0.8	0.9
Saturated	1.5 ± 0.4	6.3 ± 0.3	12.4 ± 0.2	21.3 ± 0.6	27.0 ± 0.0	
Submerged	1.3 ± 0.3	6.3 ± 0.3	12.4 ± 0.2	21.0 ± 0.0	27.5 ± 0.3	
NH_4 (ppm)						
Drained	0.63 ± 0.19	0.04 ± 0.01	0.11 ± 0.03	0.08 ± 0.01	0.30 ± 0.11	0.55
Saturated	0.14 ± 0.02	0.07 ± 0.05	0.27 ± 0.12	0.15 ± 0.03	0.28 ± 0.08	
Submerged	0.14 ± 0.05	0.11 ± 0.03	0.40 ± 0.06	1.69 ± 0.67	0.81 ± 0.20	
Sulfide (ppm)						
Drained	< 0.05 ± 0.00	< 0.05 ± 0.00	< 0.05 ± 0.00	< 0.05 ± 0.00	< 0.05 ± 0.00	13.60
Saturated	3.84 ± 0.96	4.16 ± 1.28	8.00 ± 1.28	5.12 ± 3.2	6.08 ± 2.24	
Submerged	5.12 ± 0.32	7.04 ± 1.28	12.16 ± 2.88	23.04 ± 4.16	31.36 ± 7.04	

treatment at 20 and 28 ppt salinity levels (Figure 8-11). Proline was low or not detectable at lower salinity levels and less waterlogged treatments. Interstitial water salinity levels were not significantly different among water level treatments at each salinity level (Table 8-7).

Tissue Na concentrations increased when interstitial salinities were elevated from 0 to 4 ppt, but not with further increases in salinity level ($F = 14.81$, $p \leq 0.01$) (Table 8-8). Salinity also had a significant effect on leaf N, Mg, S, P, B, and Zn concentrations ($F = 4.48$, 12.98, 20.95, 14.33, 25.52, 33.48, respectively, $p \leq 0.01$). Zn and N concentrations increased in the salinity treatments to 12 ppt, then declined (Table 8-8). Leaf S, B, and Mg concentrations generally increased with increasing salinity, while P decreased (Table 8-8).

Increased water levels significantly reduced the total belowground biomass but not live aboveground biomass or leaf expansion (Figure 8-10). Redox potentials were low and sulfide accumulated in the most submerged treatment (Table 8-7). Root ADH activity in the saturated and submerged water level treatments was also increased above that of the drained treatment at most salinity levels (Figure 8-11). Water level had a significant effect on leaf Ca, S, P, Fe, Mn, and Zn concentrations ($F = 10.39$, 27.84, 3.69, 3.28, 8.64, and 11.80, respectively, $p \leq 0.05$)

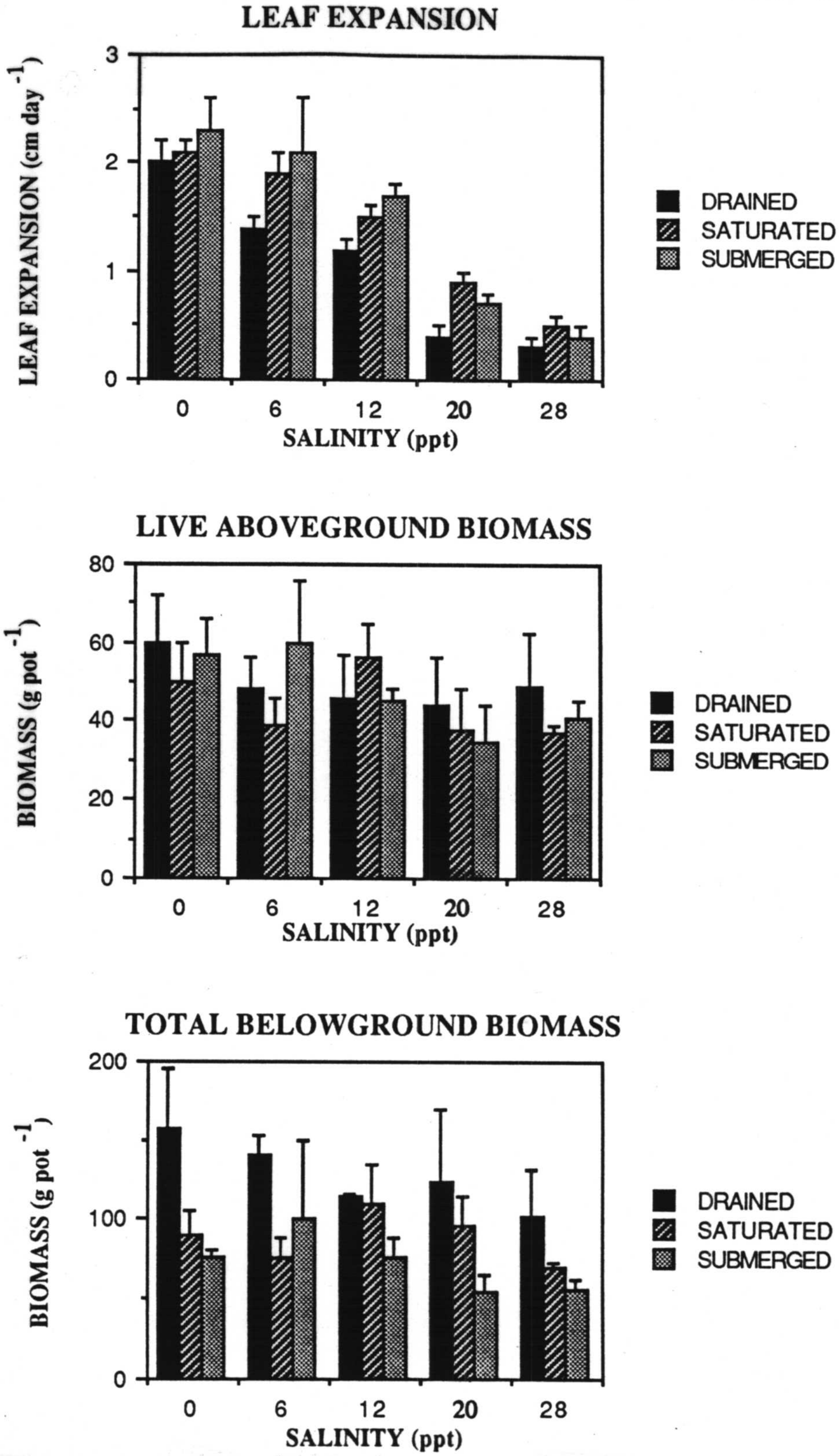
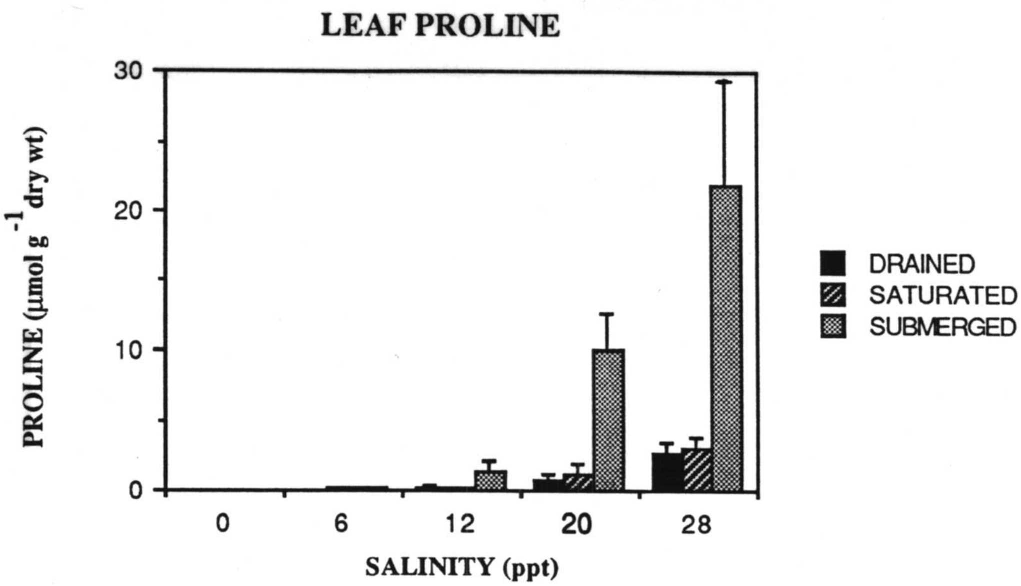


Figure 8-10. Leaf expansion rate, live aboveground biomass, and total belowground biomass of *Spartina patens* swards at five salinity levels and three water level depths in the greenhouse for 42 days (n = 4). $LSD_{0.05}$ = 0.51 cm day $^{-1}$ for leaf expansion, and 28 and 69 g pot^{-1} for above and belowground biomass, respectively.



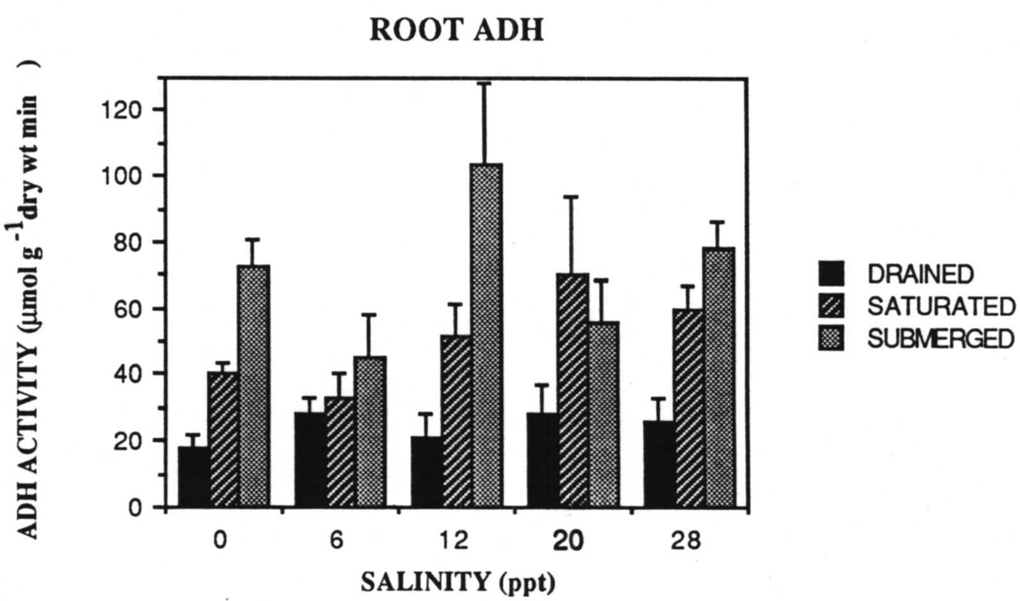


Figure 8-11. Leaf proline concentrations and root alcohol dehydrogenase (ADH) activity in *Spartina patens* at five salinity levels and three water level depths in the greenhouse after 42 days (n = 4). $LSD_{0.05}$ = 13.6 µmol g^{-1} dry wt for proline and 33 µmol g^{-1} dry wt min^{-1} for ADH.

Salt Marsh Species. Leaf expansion was affected significantly by both water level and salinity (Figure 8-12). Increased waterlogging resulted in an increase in leaf expansion at intermediate salinity levels (6 to 24 ppt) but had little effect on it at the highest (35 ppt) or the lowest (0 ppt) salinities. There was also a stimulatory effect of salinity on leaf expansion up to 12 ppt, but declined thereafter. Neither increased salinity nor waterlogging significantly reduced total live aboveground biomass, stem density or belowground biomass of *S. alterniflora* during four months of growth (Figure 8-12). Live, aboveground biomass was significantly reduced in the drained treatment at the highest salinity level (35 ppt) and the saturated 24-ppt salinity level, however. Although the interstitial water salinities indicated a slight increase in the drained 24- and 35-ppt treatments (presumably caused by evapotranspiration), the difference between these and the submerged treatments was only 1.8 - 3 ppt (Table 8-9).

Interstitial water salinities increased with increasing tank salinity level (Table 8-9). Significant increases in leaf proline concentration occurred only at the highest salinity level (Figure 8-13). At 35-ppt, proline was highest in the drained treatment and significantly lower in the saturated and submerged treatments. Spectrographic analysis of the leaf tissue showed increases in N, Ca, Mg, S, Na, and B ($F = 7.74$, 36.23, 29.28, 20.34, and 22.33, respectively, $p \leq 0.01$) and decreases in K and P ($F = 18.32$ and 2.87, respectively, $p \leq 0.01, 0.05$) with increasing salinity level (Table 8-10).

Soil redox potentials indicated that soil aeration was significantly greater in the drained treatments compared to the saturated and submerged treatments which were not significantly different (Table 8-9). Although interstitial water sulfide concentrations were considerably higher in the submerged and saturated pots compared to the drained treatments, the levels were substantially lower than measured in the field (Tables 8-3 and 8-9). Interstitial water NH_4 and pH did not vary greatly with salinity or degree of waterlogging (Table 8-9). Root ADH activities were significantly higher in roots in the saturated and submerged pots compared to those from the drained pots (Figure 8-13). Activities were low in the drained treatment at all salinity levels. Although there was some variation in root ADH activity among the salinity level treatments, no discernible pattern occurred. Increased water level also had a significant effect on leaf Na, Ca, Mg, S, P, Mn, and B concentrations ($F = 14.83$, 7.5, 4.45, 17.30, 9.79, 35.78, 7.35, respectively, $p \leq 0.01$).

Discussion

Salinity Effects

Previous investigations have reported variable responses of *S. alterniflora* to increasing salinity. Adams (1963) found that growth was not affected by salinity levels up to 20 ppt, while Haines and Dunn (1976) actually found a stimulation in growth at this level. Mooring et al. (1971) found that growth was reduced only at 40 ppt. Parrondo et al. (1978), however, reported growth reductions at levels less than 20 ppt. Aboveground biomass was not reduced in the current study by salinity levels up to full sea strength applied to plants growing in a natural substrate in the greenhouse. However, the significant reduction in leaf expansion at 35 ppt (compared to 0 ppt) indicated an effect on growth that may have eventually resulted in a decline in biomass production. Although a longer experimental period may have resulted in biomass differences in response to the highest salinity levels, the results of the 6-month long field experiment demonstrated that salinities in the range of 21 to 28 ppt had no significant effect on aboveground biomass during a growing season. Hence, it would appear that a salinity level above 28 ppt is necessary for a significant effect on growth of this species.

Table 8-8. Spectrographic analysis of leaf tissue collected from *Spartina patens* grown at five salinity levels in the greenhouse. N, K, Ca, S, P values are in %; remaining values are in ppm (n = 4).

	Salinity Level (ppt)				
	0	6	12	20	28
N	1.31 ± 0.11	1.17 ± 0.03	1.45 ± 0.08	1.21 ± 0.07	1.24 ± 0.19
K	0.7435 ± 0.0574	0.6602 ± 0.0307	0.6582 ± 0.0248	0.5531 ± 0.05821	0.5913 ± 0.0640
Ca	0.2460 ± 0.0275	0.2381 ± 0.0233	0.2442 ± 0.0099	0.1566 ± 0.0053	0.2183 ± 0.0156
Mg	0.1516 ± 0.0152	0.1425 ± 0.0192	0.1964 ± 0.0059	0.1758 ± 0.0251	0.2655 ± 0.0178
S	0.2500 ± 0.0168	0.2325 ± 0.0095	0.2625 ± 0.0111	0.2725 ± 0.0144	0.4250 ± 0.0614
P	0.0878 ± 0.0061	0.0882 ± 0.0060	0.0872 ± 0.0028	0.0676 ± 0.0065	0.0679 ± 0.0064
Na	171.2 ± 20.1	228.6 ± 16.6	275.4 ± 12.4	203.2 ± 16.2	266.5 ± 14.3
Fe	153.9 ± 12.7	111.2 ± 11.4	141.5 ± 14.4	128.8 ± 19.2	136.9 ± 4.9
Mn	73.54 ± 3.58	85.46 ± 10.97	98.47 ± 14.84	67.82 ± 6.34	78.6 ± 11.7
Al	110.2 ± 14.3	73.84 ± 11.64	101.5 ± 13.3	90.68 ± 19.26	109.0 ± 12.5
B	4.08 ± 0.34	5.78 ± 0.99	7.57 ± 0.42	6.85 ± 0.19	12.05 ± 1.19
Cu	5.09 ± 0.20	6.50 ± 1.11	5.31 ± 0.22	3.46 ± 0.21	4.06 ± 0.30
Zn	14.38 ± 0.94	15.04 ± 1.82	16.25 ± 1.33	6.11 ± 0.46	6.49 ± 0.35
Saturated					
N	1.28 ± 0.08	1.49 ± 0.07	1.47 ± 0.13	1.20 ± 0.07	1.26 ± 0.10
K	0.6708 ± 0.1085	0.7488 ± 0.0685	0.7122 ± 0.1349	0.7196 ± 0.1210	0.7928 ± 0.0973
Ca	0.1556 ± 0.0179	0.1955 ± 0.0155	0.1707 ± 0.0180	0.1939 ± 0.0108	0.1709 ± 0.0119
Mg	0.1086 ± 0.0127	0.1663 ± 0.0078	0.1526 ± 0.0310	0.2023 ± 0.0129	0.2178 ± 0.0296
S	0.2075 ± 0.0189	0.3325 ± 0.0405	0.5975 ± 0.1132	0.6503 ± 0.1374	0.7475 ± 0.0626
P	0.1026 ± 0.0071	0.1072 ± 0.0054	0.0818 ± 0.0090	0.0780 ± 0.0031	0.0713 ± 0.0072
Na	151.5 ± 18.2	232.1 ± 6.3	252.1 ± 21.4	236.0 ± 19.3	241.5 ± 8.5
Fe	159.0 ± 23.9	154.4 ± 8.2	152.0 ± 21.8	147.8 ± 28.9	139.7 ± 8.0
Mn	53.04 ± 2.77	72.51 ± 7.57	52.24 ± 3.49	64.47 ± 5.25	62.7 ± 4.4
Al	111.2 ± 22.7	112.6 ± 3.9	117.8 ± 17.3	77.48 ± 2.08	116.9 ± 3.6
B	3.81 ± 0.38	5.88 ± 0.48	6.00 ± 0.48	7.68 ± 0.78	7.47 ± 0.78
Cu	5.40 ± 0.90	8.73 ± 0.28	4.75 ± 0.25	4.68 ± 0.66	23.04 ± 19.20
Zn	13.84 ± 1.09	20.42 ± 0.76	25.31 ± 4.12	8.53 ± 0.78	15.02 ± 2.70
Submerged					
N	1.50 ± 0.04	1.30 ± 0.07	1.43 ± 0.14	1.17 ± 0.03	1.12 ± 0.04
K	0.7531 ± 0.0391	0.7202 ± 0.0549	0.7253 ± 0.1106	0.8184 ± 0.0648	0.7191 ± 0.1034
Ca	0.1755 ± 0.0091	0.1950 ± 0.0287	0.1895 ± 0.0172	0.1735 ± 0.0202	0.1325 ± 0.0178
Mg	0.1432 ± 0.0027	0.1251 ± 0.0135	0.1881 ± 0.0062	0.2245 ± 0.0176	0.1938 ± 0.0351
S	0.2575 ± 0.0048	0.4550 ± 0.0437	0.6150 ± 0.0210	0.8650 ± 0.0775	0.5933 ± 0.0636
P	0.1174 ± 0.0059	0.0952 ± 0.0104	0.0946 ± 0.0070	0.0675 ± 0.0056	0.0775 ± 0.0093
Na	167.8 ± 12.3	231.1 ± 25.7	273.5 ± 12.5	257.2 ± 16.9	199.3 ± 31.0
Fe	228.6 ± 31.5	135.9 ± 16.2	438.8 ± 238.9	195.4 ± 25.6	152.4 ± 28.2
Mn	69.08 ± 2.40	63.78 ± 6.22	68.60 ± 3.97	73.3 ± 7.8	55.8 ± 14.4
Al	128.4 ± 41.2	108.9 ± 22.2	161.9 ± 32.6	115.8 ± 21.7	102.1 ± 11.4
B	4.41 ± 0.27	5.92 ± 0.67	6.62 ± 0.22	8.23 ± 0.80	9.11 ± 1.47
Cu	5.41 ± 0.15	7.75 ± 0.88	7.71 ± 2.82	4.61 ± 0.83	3.73 ± 0.28
Zn	18.27 ± 1.71	17.59 ± 1.28	42.49 ± 6.90	6.49 ± 0.45	7.13 ± 0.35

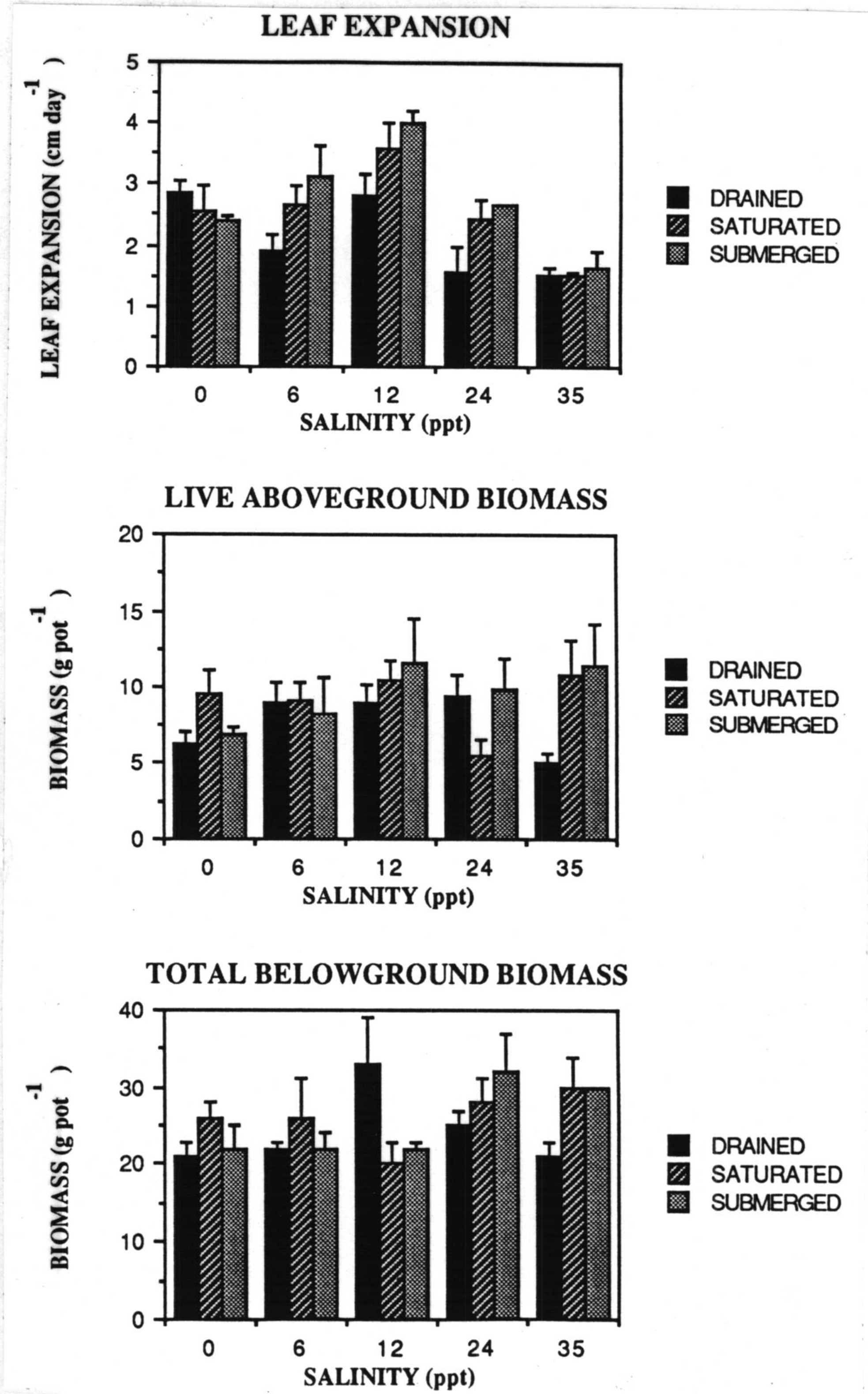


Figure 8-12. Leaf expansion, live aboveground biomass, and total belowground biomass of *Spartina alterniflora* swards at five salinity levels and three water level depths in the greenhouse for 115 days (n = 4). LSD $_{0.05}$ = 0.78 cm day $^{-1}$ for leaf expansion, and 5 and 9 g pot $^{-1}$ for above and belowground biomass, respectively.

Table 8-9. Soil redox potentials (Eh) and interstitial water pH, salinity, NH_4, and sulfide measured in *Spartina alterniflora* swards at three elevations and five salinity levels in the greenhouse (n = 4).

	Salinity Level (ppt)					
	0	6	12	24	35	$LSD_{.05}$
Eh at 1 cm (mV)						
Drained	456 ± 34	411 ± 36	434 ± 14	379 ± 24	316 ± 10	79
Saturated	-171 ± 20	-168 ± 28	-165 ± 21	-190 ± 10	-146 ± 6	
Submerged	-179 ± 34	-177 ± 25	-187 ± 13	-214 ± 12	-188 ± 27	
Eh at 8 cm (mV)						
Drained	228 ± 78	44 ± 43	-23 ± 28	7 ± 22	218 ± 31	78
Saturated	-150 ± 10	-178 ± 5	-214 ± 9	-193 ± 11	-190 ±8	
Submerged	-204 ± 10	-198 ± 9	-200 ± 12	-211 ± 7	-222 ± 19	
Salinity (ppt)						
Drained	3.0 ± 0.0	8.3 ± 0.3	13.5 ± 0.5	26.8 ± 0.5	38.8 ± 0.9	0.9
Saturated	3.0 ± 0.0	8.5 ± 0.4	14.3 ± 0.3	26.8 ± 0.5	36.8 ± 0.5	
Submerged	3.0 ± 0.0	7.3 ± 0.1	13.0 ± 0.0	25.0 ± 0.0	35.8 ± 0.3	
pH						
Drained	7.1 ± 0.0	6.9 ± 0.1	6.8 ± 0.1	6.8 ± 0.1	6.8 ± 0.2	0.3
Saturated	7.1 ± 0.2	6.9 ± 0.0	7.1 ± 0.0	7.2 ± 0.2	7.1 ± 0.1	
Submerged	7.2 ± 0.1	7.1 ± 0.0	7.3 ± 0.1	7.2 ± 0.0	7.3 ± 0.0	
NH_4 (ppm)						
Drained	0.46 ± 0.11	0.20 ± 0.03	0.37 ± 0.15	0.22 ± 0.02	0.16 ± 0.07	0.3
Saturated	0.31 ± 0.05	0.12 ± 0.02	0.30 ± 0.10	0.35 ± 0.10	0.26 ± 0.09	
Submerged	0.21 ± 0.07	0.12 ± 0.02	0.22 ± 0.05	0.51 ± 0.17	0.29 ± 0.07	
Sulfide (ppm)						
Drained	1.43 ± 0.97	1.43 ± 0.73	0.17 ± 0.16	0.36 ± 0.13	0.00 ± 0.00	7.3
Saturated	9.19 ± 3.29	14.65 ± 2.36	8.38 ± 2.82	9.27 ± 2.54	7.00 ± 3.34	
Submerged	11.64 ± 3.43	17.98 ± 6.13	8.35 ± 2.46	13.01 ± 1.03	3.69 ± 0.77	

Decreased growth of plants subjected to increased soil salinity can occur for three reasons: (1) a water deficit caused by a low external water potential (elevated osmotic pressure); (2) an excess of ions in the tissue due to uptake of salts from the soil solution; or, (3) an inhibition of nutrient ion uptake from excessive Na^+ or Cl^- concentrations (Greenway and Munns, 1980). Halophytes (salt-tolerant plant species) are unique in their ability to accumulate high concentrations of salts in their tissues for osmotic adjustment without causing adverse effects. Ions are compartmentalized in vacuoles and balanced by neutral organic solutes in the cytoplasm (Greenway and Munns, 1980). Halophytes also efficiently control tissue Na+ and Cl- concentrations through exclusion at the root, control of translocation of ions to shoots, and/or extrusion from the leaves through salt glands (Flowers, 1985). Although facultative halophytes are tolerant of salt water, their growth in such conditions exacts a cost. Therefore, optimal growth in these plants is usually seen in freshwater; decreases in growth occur with increasing salinity level. Thus, a decrease in growth or biomass of salt-tolerant species in response to increased salinity cannot necessarily be interpreted to mean that that species will not survive at that salinity level. However, if its competitive ability is compromised, then it might be competitively excluded by a more salt-tolerant species.

S. alterniflora can apparently exclude salt (Smart and Barko, 1980), accumulate it in the leaf (Nestler, 1977) and extrude it onto leaf surfaces through salt glands (Anderson, 1974). In this way *S. alterniflora* can effectively control tissue electrolyte concentrations. However, an increase in external salinity may cause a plant water deficit through an inability of the plant to maintain a gradient in water potential with the external rooting media. Drake and Gallagher (1984) showed that *S. alterniflora* is capable of adjusting osmotically, i.e., increasing inorganic or organic ion concentrations that lower tissue osmotic potentials and maintain turgor pressure. A failure to completely adjust osmotically (resulting from an inability to increase the concentration of Na+ or organic osmotica to sufficient concentrations) would result in a decrease in leaf turgor pressure and, consequently, reduced leaf expansion (Bradford and Hsiao, 1982). *S. alterniflora* is apparently able to reduce its osmotic potential in response to increased interstitial water salinity but may not be able to maintain turgor pressure and high rates of growth when external salinities are high (Drake and Gallagher, 1984; Figure 8-6). The decrease in leaf expansion at 35-ppt (greenhouse, Figure 8-12) was, thus, most likely the result of decreased water uptake, since the maintenance of leaf water volume and turgor pressure is required for growth.

Salinity levels of 21-28 ppt produced a significant reduction (compared to controls at approximately 12 ppt) in aboveground biomass of *S. patens* in the field. The same salinity levels had no effect on *S. alterniflora*. The more controlled greenhouse experiment demonstrated that leaf expansion was reduced, depending on the degree of submergence, by salinity increases up to 12 ppt, but was substantially reduced at 20 ppt. However, the reductions in leaf expansion in the greenhouse were not mirrored by differences in live aboveground biomass. Palmisano (1970) found that growth of *S. patens* was reduced by 50 % at a salinity level of 8 ppt. Differences in experimental conditions between this (natural substrate) and Palmisano's (1970) (sand culture) study may explain the lack of agreement, however. The results of the greenhouse experiment showed that the level of flooding significantly modified the response of *S. patens* to salinity. An increase in salinity to only 6 ppt produced a significant reduction in leaf expansion in *S. patens* plants growing in the drained treatment, while growth inhibition did not occur until 20 ppt in the submerged treatment. Additional water stress in the drained treatment may have caused this result.

Pezeshki et al. (1987a) found significant effects on *S. patens* stomatal behavior and photosynthesis in response to increases in salinity (4 - 22 ppt). Stomatal conductance and net photosynthetic rate were reduced to approximately 45% of pre-salt levels when salinities were increased to 9-12 ppt (Figure 3 in Pezeshki et al., 1987a). However, this physiological response can, at best, only be considered a short-term one, since their measurements were taken over a period of approximately one week following each of two sudden applications of salt. Their data cannot be interpreted with respect to direct effects of salt stress on *S. patens'* photosynthetic mechanism (and, thus, long-term growth response) because of the probable interference of a short-term stomatal response (to control water loss upon sudden exposure to physiological drought conditions). The results of the present study showed that leaf expansion of *S. patens* was reduced only 9 and 26% at 6 and 12 ppt, respectively (submerged treatments, Figure 8-10) when salinity levels were increased gradually and plant response was monitored over a longer period of time (42 days).

Table 8-10. Spectrographic analysis of leaf tissue collected from *Spartina alterniflora* grown at five salinity levels in the greenhouse. N, K, Ca, Mg, S. P values are in %; remaining values in ppm (n = 4).

	Saliinity Level (ppt)				
	0	6	12	24	35
Drained					
N	1.18 ± 0.08	1.00 ± 0.08	1.33 ± 0.09	1.18 ± 0.04	1.66 ± 0.09
K	1.4053 ± 0.0842	1.2740 ± 0.0913	1.0811 ± 0.0654	0.9507 ± 0.0495	0.9970 ± 0.0179
Ca	0.1440 ± 0.0071	0.2311 ± 0.0133	0.3222 ± 0.0411	0.3619 ± 0.0161	0.4205 ± 0.0581
Mg	0.2087 ± 0.0192	0.3537 ± 0.0216	0.4590 ± 0.0313	0.4547 ± 0.0369	0.5742 ± 0.0265
S	0.5075 ± 0.0397	0.5000 ± 0.0438	0.5175 ± 0.0403	0.5775 ± 0.0155	0.8050 ± 0.0629
P	0.0688 ± 0.0038	0.0754 ± 0.0072	0.0830 ± 0.0101	0.0739 ± 0.0038	0.0815 ± 0.0018
Na	105.7 ± 11.3	181.8 ± 7.8	250.7 ± 28.8	231.1.3 ± 48.2	434.1 ± 26.6
Fe	188.9 ± 52.7	267.6 ± 60.6	261.2 ± 78.2	258.3 ± 48.2	205.3 ± 20.4
Mn	92.6 ± 13.0	119.5 ± 21.1	114.8 ± 26.8	125.4 ± 21.8	104.0 ± 14.9
Al	238.8 ± 66.3	302.5 ± 60.2	286.9 ± 104.5	236.8 ± 54.7	350.4 ± 40.2
B	4.95 ± 0.78	10.65 ± 1.59	12.77 ± 2.12	14.68 ± 2.45	33.64 ± 1.79
Cu	4.66 ± 0.08	5.52 ± 0.51	5.46 ± 0.44	4.36 ± 0.12	4.83 ± 0.10
Zn	16.70 ± 1.15	14.69 ± 0.68	16.45 ± 1.54	13.76 ± 1.16	18.18 ± 0.85
Saturated					
N	1.33 ± 0.07	1.35 ± 0.11	1.38 ± 0.14	1.54 ± 0.09	1.66 ± 0.23
K	1.3703 ± 0.0533	1.2785 ± 0.0334	1.2128 ± 0.0675	1.1335 ± 0.0618	1.0400 ± 0.0763
Ca	0.1189 ± 0.0071	0.2185 ± 0.0310	0.2539 ± 0.0282	0.3245 ± 0.0402	0.3552 ± 0.0226
Mg	0.2916 ± 0.0185	0.5055 ± 0.0918	0.5191 ± 0.0292	0.5184 ± 0.0789	0.5842 ± 0.0221
S	0.7675 ± 0.0592	0.7900 ± 0.0492	0.7300 ± 0.0829	0.8025 ± 0.1210	0.7300 ± 0.0540
P	0.0905 ± 0.0036	0.0926 ± 0.0100	0.0952 ± 0.0064	0.1010 ± 0.0032	0.0831 ± 0.0071
Na	277.4 ± 10.9	341.3 ± 55.2	332.8 ± 20.7	303.1 ± 51.3	401.7 ± 29.0
Fe	271.6 ± 41.8	321.4 ± 118.4	226.9 ± 43.7	145.1 ± 22.9	205.3 ± 20.4
Mn	37.83 ± 1.75	44.1 ± 4.9	46.41 ± 0.80	43.47 ± 5.16	35.09 ± 5.98
Al	378.0 ± 56.6	382.6 ± 150.3	233.5 ± 55.44	118.9 ± 27.3	192.0 ± 23.5
B	15.53 ± 1.27	19.59 ± 7.12	17.26 ± 2.47	19.77 ± 5.16	31.50 ± 1.61
Cu	4.35 ± 0.17	4.85 ± 0.31	4.80 ± 0.07	4.26 ± 0.14	4.17 ± 0.27
Zn	13.25 ± 0.54	16.11 ± 1.05	15.36 ± 1.58	12.17 ± 0.94	14.44 ± 4.91
Submerged					
N	1.45 ± 0.06	1.40 ± 0.08	1.69 ± 0.12	1.90 ± 0.15	1.81 ± 0.14
K	1.3825 ± 0.0488	1.3430 ± 0.0946	1.1655 ± 0.1050	1.1533 ± 0.0404	0.8837 ± 0.0543
Ca	0.1204 ± 0.0083	0.1707 ± 0.0102	0.2433 ± 0.0138	0.3007 ± 0.0288	0.3144 ± 0.0219
Mg	0.2163 ± 0.0102	0.3023 ± 0.0168	0.4828 ± 0.0616	0.5179 ± 0.0358	0.5888 ± 0.0287
S	0.5575 ± 0.0269	0.6075 ± 0.0581	0.5075 ± 0.0419	0.5900 ± 0.0289	0.6600 ± 0.0316
P	0.1190 ± 0.0044	0.1125 ± 0.0118	0.0805 ± 0.0046	0.0792 ± 0.0063	0.0694 ± 0.0044
Na	161.7 ± 6.2	182.8 ± 11.8	273.1 ± 54.2	272.4 ± 18.2	336.3 ± 17.9
Fe	219.8 ± 26.8	236.8 ± 67.3	281.0 ± 35.9	204.3 ± 29.2	317.5 ± 31.9
Mn	67.38 ± 5.41	76.02 ± 14.29	68.92 ± 6.85	82.69 ± 12.70	68.23 ± 3.30
Al	351.3 ± 45.5	268.4 ± 58.2	304.6 ± 60.3	184.4 ± 36.5	335.5 ± 46.4
B	6.25 ± 0.53	6.51 ± 0.50	14.17 ± 5.27	15.55 ± 1.48	26.15 ± 1.56
Cu	4.73 ± 0.09	5.02 ± 0.41	4.15 ± 0.22	4.47 ± 0.18	4.42 ± 0.26
Zn	17.11 ± 1.35	22.38 ± 2.66	18.61 ± 1.22	14.78 ± 0.41	12.55 ± 0.58

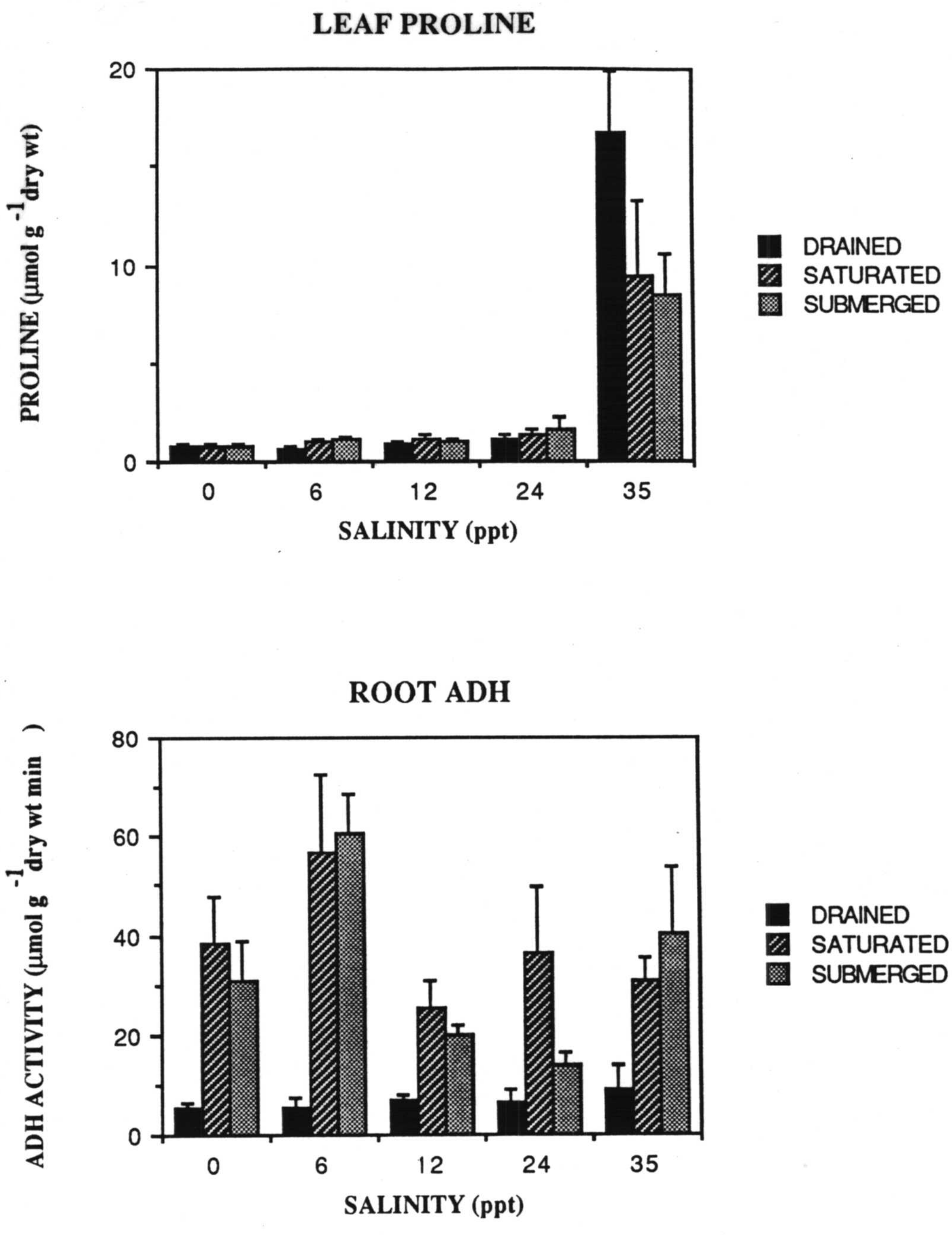


Figure 8-13. Leaf proline concentrations and root alcohol dehydrogenase (ADH) activity in *Spartina alterniflora* at five salinity levels and three water level depths in the greenhouse after 115 days (n = 4). $LSD_{0.05}$ = 4.3 µmol g^{-1} dry wt for proline and 22 µmol g^{-1} dry wt min^{-1} for ADH.

Because *S. patens* possesses salt glands (Anderson, 1974), this species has the capability of extruding excessive salt, thus aiding in control of internal electrolyte concentrations. The significant reduction in aboveground biomass of *S. patens* in response to increased salinity in the field probably was caused by a water deficit that resulted in substantial tissue death upon sudden exposure to more saline conditions. Two months after transplantation to the higher salinity marsh, approximately 50% of the original tissue was dead. However, new, green shoots had also appeared in these plots. The absence of significant differences in live aboveground biomass in the greenhouse experiment reflects the fact that gradual exposure to higher salinities did not cause significant death of the original material. These results illustrate the importance of sufficient time for osmotic adjustment and that the response (of even a salt-tolerant species) to increased salinity may depend not only on the final level, but upon pre-adaptation and abruptness of exposure to that salinity level.

The fresh marsh field results (Figure 8-4) agree with previous work which showed that tissue death of *P. hemitomon* occurred within 4 days after sudden application of salt equivalent to 10-12 ppt (Pezeshki et al., 1987b). None of the fresh marsh species in the current study survived a sudden increase in salinity to 15 ppt. However, the results of our greenhouse experiment showed that although growth was reduced, *P. hemitomon* and *L. oryzoides* were relatively tolerant of salinities of 8-11 ppt for at least one month (Figure 8-8, Table 8-5.). A gradual increase in salinity level probably allowed these species time to osmotically adjust in the greenhouse experiment. *S. lancifolia* was more sensitive to increases in salinity and showed symptoms of tissue damage at 4-5 ppt. Visual symptoms (other than reduced biomass) were not apparent in *P. hemitomon* or *L. oryzoides* in this experiment.

Investigations with other aquatic macrophytes have shown a variable response to increased salinity (Haller et al., 1974). A salinity level of 2.5 ppt inhibited the growth of a number of aquatic macrophytes, including *Eichornia crassipes* and *Pistia stratiotes.* However, growth of *Lemna minor* was stimulated by salinities up to 3.33 ppt and not decreased until 6.66 ppt. *Myriophyllum brasiliense* could tolerate salinities up to 13.32 ppt. Haller et al. (1974) concluded that the morphology of aquatic plants may be a factor determining salt tolerance since the larger floating species were the most susceptible to low levels of salinity and the submersed or small floating species the most tolerant. The more xerophytic nature of the leaf structure of the grasses, *P. hemitomon* and *L. oryzoides,* may have been a factor in their relatively greater salt tolerance compared to *S. lancifolia* which has broader leaves and a greater surface area for water loss.

Nonhalophytes usually fall into two major categories of response to salinity: (1) those that cannot avoid ion uptake and which experience adverse effects when electrolyte concentrations become excessive, and (2) those that avoid ion uptake (Greenway and Munns, 1980). Nonhalophytes which exclude ions must adjust osmotically by producing compatible organic solutes or else suffer a water deficit. In the latter case, the cost of osmotic adjustment would limit plants using this strategy to relatively low external salt concentrations (Yeo, 1983). *P. hemitomon* was accumulating Na+ in its shoots which would contribute to osmotic adjustment, but this accumulation apparently became excessive and inhibited growth in the plants growing at 8 to 11 ppt (Figure 8-8; Table 8-5). The Gramineae includes some of the more salt tolerant of nonhalophytes and may partly explain the relatively greater tolerance of *P. hemitomon* and *L. oryzoides* compared to *S. lancifolia.* Some grasses which are more tolerant of salt appear to have the ability to adjust osmotically by regulating the influx of Na+ and Cl- (Gorham et al., 1985). Work with several species of *Plantago* demonstrated that a halophytic species was able to restrict salt intake to some extent but that the nonhalophytic species could not prevent excessive accumulation of electrolytes (Koningshofer, 1983). The sudden demise of the three freshwater marsh species in the field study was presumably due to an inadequate period of time for osmotic adjustment and a resultant water deficit. This conclusion is supported by the greenhouse results which demonstrated that these species could survive a more gradual exposure to increased salinities which apparently allowed for osmotic adjustment.

Proline accumulates in response to salt stress in both halophytes and nonhalophytes (Aspinall and Paleg, 1981). Although accumulation of proline has been shown to occur in *S. alterniflora* and *S. patens* in response to salinity stress (Cavalieri and Huang, 1979, 1981; Cavalieri, 1983) and has been proposed as a cytoplasmic osmoticum in many halophytes (Flowers et al., 1977; Wyn Jones et al., 1977; Rozema et al., 1978; Cavalieri and Huang, 1979), its importance in osmoregulation has been questioned (Cavalieri, 1983). However, more recent evidence suggests that proline may protect protein (enzymes) systems against dehydration (Paleg et al., 1985). Regardless of its function, proline accumulation appears to indicate a stress response. Proline accumulation in *S. alterniflora* is apparently affected by nitrogen availability (Cavalieri, 1983) and does not occur until a threshold level of salinity is reached (Cavalieri and Huang, 1979; Cavalieri, 1983) (Figure 8-13). The threshold level for proline accumulation appears to be between 24 and 35 ppt for *S. alterniflora* and between 20 and 28 ppt for *S. patens* (Cavalieri and Huang, 1979; Figures 8-11 and 8-13). The relatively low proline levels in the *S. alterniflora* and *S. patens* plants transplanted to the higher salinity marsh (21-28 ppt) indicated that these plants were not accumulating proline, perhaps because the threshold level had not been reached.

The significant increase in leaf proline in *P. hemitomon* at the highest salinity level and where growth was significantly reduced supports the view that proline accumulation indicates a water stress and poor osmotic adjustment (Stewart and Hanson, 1980). Proline apparently does not contribute to osmotic adjustment in this species at lower salinity levels. Although proline did not accumulate to any great extent in the leaves of *S. lancifolia* or *L. oryzoides*, other compatible solutes such as glycinebetaine or sucrose may have served this function (Greenway and Munns, 1980). Gorham et al. (1985) found that sucrose and glycinebetaine, but not proline, contributed to the osmotic adjustment of *Leymus sabulosus* and *Elytrigia juncea* to increased external salinity.

Waterlogging Effects

Flooding with salt water may cause stresses in addition to those resulting from increased electrolyte concentrations or physiological drought. If the influx of salt water is accompanied by an increase in depth or duration of flooding, then the plants may also experience root oxygen deficiencies, decreased nutrient uptake, and/or a buildup of toxic compounds such as hydrogen sulfide in the highly reducing soil environment (Kozlowski, 1984; Hook, 1984; Mendelssohn and Burdick, in press). Inland salt marshes in Louisiana are characterized by low productivity, decreased elevations due to subsidence, increased waterlogging, and high soil sulfide concentrations (Mendelssohn et al., 1981; DeLaune et al., 1983). Decreased growth of *S. alterniflora* in these inland locations does not appear to be caused by salt accumulation, since salinity levels are typically equal to or lower than that adjacent to tidal creeks (Mendelssohn and McKee, 1987). A recent study has shown that decreased growth and dieback of *S. alterniflora* in inland Louisiana marshes is caused by factors related to increased soil waterlogging (Mendelssohn and McKee, 1987). Soil waterlogging may lead to an increased soil oxygen demand, increased root anaerobic metabolism, reduced nitrogen uptake, ion toxicity, and a buildup of toxic compounds such as hydrogen sulfide (see Mendelssohn et al., 1982).

The results of the field experiments in the current study agree with previous findings that factors associated with chronic waterlogging reduce plant growth and survival

(Mendelssohn and McKee, 1987). This response was true for species from all three habitats. A primary candidate causing decreased growth of saline marsh vegetation in chronically waterlogged soils is hydrogen sulfide. Although the exact mechanism by which sulfide exerts its toxic effect on plants is not well understood, hydrogen sulfide has been shown to inhibit enzymes involved in aerobic respiration (Allam and Hollis, 1972). Sulfides may combine with metallo-enzymes such as cytochrome oxidase and polyphenol oxidase and inhibit their activity directly (Allam and Hollis, 1972) or may complex with essential metals and reduce their availability for enzyme reactions (Havill, et al., 1985). The significant negative correlation between interstitial water sulfide and biomass in the current study further supports the potential role of this toxic compound in causing reduced growth and dieback of vegetation in saline and brackish marshes.

Changes in interstitial water and tissue nutrient concentrations indicated an effect of waterlogging on uptake of several elements. Increased NH_4 concentrations in the interstitial water of the lower elevation plots may reflect a decreased uptake by the plants in all three habitats. The significant correlation between NH_4 and sulfide supports previous statements that nitrogen uptake by marsh plants may be inhibited by increases in sulfide concentrations in reduced sediments (Mendelssohn, 1979; Morris, 1979). Although tissue N concentrations did not reflect the proposed reduced uptake (Table 8-4), changes in plant biomass may have caused a concentration of N in the leaves which effectively obscured any reduction in uptake and assimilation of N. However, increased NH_4 concentrations in the interstitial water could also have resulted from changes in the soil microflora, i.e., less efficient uptake by anaerobic microorganisms.

Changes in other tissue nutrient concentrations suggested the possibility of an excessive uptake of potentially toxic ions in the field experiment. Iron is more soluble and readily taken up by plants in the reduced form. The availability of cations such as Fe^{+2} are, thus, directly related to the redox potential of the rooting medium. The greater Fe concentration in the leaves of the *S. patens* and *S. alterniflora* plants growing in the low elevation treatment in the recipient marsh shows a greater uptake of Fe by the plants growing in the more submerged treatment.

The absence of a negative response to increased waterlogging in the greenhouse experiments may reflect an inability to precisely re-create the conditions present in the field environment which lead to reduced growth. The primary factor may have been an edge effect, i.e., plants growing in pots typically produce roots which encircle the inside surface of the pot where aeration would be greater than in the interior of the soil core. This response was observed for all three vegetation types growing in the greenhouse. Thus, even though sulfide levels and soil Eh values indicated a highly reducing environment in the interior of the greenhouse pots, the plants may have been able to avoid any adverse effects by directing root growth to the outer edge of the soil cores. A similar strategy was not possible in the field where the soil cores were surrounded by sediment and where no zones of aeration were available for exploitation. In addition, the shorter-term greenhouse experiments may not have allowed sufficient time for a stress response to be manifested as a change in aboveground production. It is likely that the decreased growth observed at lower elevations in the field was due to some combination of decreased root aeration, increased soil phytotoxin concentrations, ion toxicity, and reduced nitrogen uptake. However, further research is necessary to completely characterize the mechanism(s) involved in reducing plant growth in chronically waterlogged soils.

A relatively small but statistically significant decrease in aboveground biomass in the drained treatments in the greenhouse and the +10 cm elevation treatment in the field showed that *S. alterniflora* was adversely affected by highly drained soil conditions even though the soil remained moist (Figures 8-6 and 8-13). This effect was not due to an increase in salt concentration in the soil since salinities in the field were not significantly different among elevations and differed only slightly among elevations at the highest salinity levels in the greenhouse (Tables 8-3 and 8-8). *S. patens* displayed a similar response, but to a lesser degree (Figure 8-11). Reduced growth of *S. patens* and *S. alterniflora* in the continuously-drained substrate was most likely a result of a water deficit since availability of water would be less than in the more flooded treatments. A water deficit would lead to a decrease in leaf water volume and turgor pressure which would, in turn, result in reduced leaf expansion (Bradford and Hsiao, 1982). Parrondo et al. (1978) and Mendelssohn and Seneca (1980) have also shown reduced growth of *S. alterniflora* in more drained conditions.

Conclusions

The results of this study showed that *S. alterniflora* growth is normally not inhibited by salinity levels equal to that found in Louisiana's coastal waters. However, increased soil waterlogging brought about by a 10 cm decline in surface elevation significantly inhibited the growth of this species. These results indicate that the reduced vigor of *S. alterniflora*-dominated salt marshes in Louisiana may be primarily caused by factors associated with the chronic waterlogging characteristic of specific sites, i. e., accumulation of toxic compounds in the highly reduced soil substrate and/or root oxygen deficiencies. Reduced growth in this species also occurred when the substrate became too highly drained of soil water.

Spartina patens was not only sensitive to increased soil waterlogging, but was less tolerant of increases in salinity than *S. alterniflora.* The results of this study suggest that if the salinity in a *S. patens*-dominated brackish marsh is increased above 21 ppt, (1) the aboveground biomass would be significantly reduced in a single growing season and (2) that the combined effect of increased waterlogging and salinity would have a greater potential for causing deterioration of a brackish marsh than that of either factor acting alone. However, *S. patens* was capable of adjusting to and did survive a salinity level of 28 ppt in the greenhouse when allowed to slowly acclimate to the increase in salinity. Regrowth of *S. patens* at similar salinity levels was also observed in the field.

Although plant species growing in fresh marsh habitats would be affected adversely by increases in salinity, the results of this study have shown that this response may vary depending upon the species. Broad-leaved species such as *Sagittaria lancifolia* may be relatively more sensitive to increases in salinity than grasses. *Panicum hemitomon* and *Leersia oryzoides* were able to survive and grow (although at a reduced rate) for one month at salinities of 8-11 ppt in the greenhouse. Even *Sagittaria lancifolia* survived salinity levels of 4-5 ppt. Thus, marshes comprised of these species might be able to survive small increases in salinity for short periods of time, but would probably quickly succumb to sudden influxes of salt water which increased salinities above 10 ppt. Although *P. hemitomon* was more sensitive to submergence in the field than the other two species, the relative flood tolerance of the three species could not be completely characterized based on the results of this study and requires further investigation. However, since flood tolerance among fresh marsh plant species can vary, the effect of subsidence in this type of habitat would likely differ among marshes of different species composition.

Implications for Louisiana's Marshes

The results of this study show that the response of marsh vegetation to increases in salinity is influenced by a number of factors including vegetation type; level, duration, and abruptness of exposure; and level of inundation. In light of these important factors that can potentially modify plant response, predictions regarding the effect of saltwater intrusion in

Louisiana's marshes must be made with caution. First of all, a distinction must be made between vegetative change and marsh deterioration. Marsh deterioration, by definition, implies a reduction in total biomass that ultimately leads to a complete elimination of emergent macrophytes in an area. Vegetative change may involve a change in species composition, biomass, or both. The elimination of one species through a change in salinity regime, for example, would not necessarily result in marsh deterioration if the eliminated species is replaced by another species more tolerant of salinity. The key factors distinguishing the two types of change may be the abruptness of exposure to the stress and the relative vulnerability of the dominant species to that stress, as well as the presence of propagules of more tolerant species.

We have avoided an overly simplistic interpretation of the data, i.e., that reductions in growth parameters in response to increases in salinity imply that saltwater intrusion would invariably result in marsh deterioration. Instead, we envision a number of possible scenarios depending upon the interaction of several biotic and abiotic factors.

A gradual change in salinity level would allow time for osmotic adjustment (within the genetic potential of each species) and, thus, preclude the sudden elimination of a vegetation type from a marsh. For example, our data showed that a freshwater marsh species that succumbed rapidly to a sudden influx of salt water survived a relatively high level of salinity (8-11 ppt) for a month if exposed to it gradually over a period of several days. A gradual change in salinity would allow time for succession to a vegetation type more tolerant of salinity. This would, of course, depend upon a source of propagules, i.e., seeds or rhizomes. The subsidence of the Mississippi deltaic plain in combination with world-wide sea level rise may allow the gradual encroachment of salt water into less saline regions in Louisiana. If this change in salinity is gradual enough, less salt-tolerant species may be replaced by more tolerant ones. However, a rapid penetration of salt water into areas which would have otherwise not experienced such a sudden increase in salinity may result in the elimination of the emergent vegetation and a subsequent deterioration (erosion) of the substrate so that invasion by new species becomes unlikely.

The response of a marsh to saltwater intrusion will also be influenced significantly by species composition. Freshwater species appear to have a varied susceptibility to saltwater and may respond differently depending on individual characteristics. The limited information from the current study prevents any reasonable predictions regarding the response of freshwater marshes in general to saltwater intrusion since we investigated only three of many freshwater species. Grasses such as *Panicum hemitomon* may be somewhat more salt-tolerant than broad-leaved species such as *Sagittaria lancifolia*. However, this observation requires additional investigation to determine the generality of this difference. The data also showed that these freshwater species could survive (although growth was reduced) small increases in salinity for short periods of time. The implication of these results is that small influxes of salt water may reduce growth temporarily but that if conditions return to normal, the plants may recover.

The effect of saltwater intrusion on a brackish marsh dominated by *Spartina patens* would depend primarily upon the rapidity of change, the final salinity level, and the level of inundation. Sudden influxes of salt water that increase water level by 10 cm and salinity to 28 ppt could cause a 90% reduction in aboveground biomass in one growing season and would most likely result in the eventual deterioration of the marsh. However, a more gradual increase in salinity would probably not result in such a dramatic decline in biomass of *S. patens* and possibly would allow the gradual replacement by more tolerant species, i.e., *S. alterniflora*.

Spartina alterniflora growth was essentially unaffected by increases in salinity to levels present in Louisiana's coastal waters. This result was not surprising since numerous other studies have documented the vigorous growth of this species in Atlantic coast marshes where salinities often equal that of seawater (36 ppt). *Spartina alterniflora* is, however, adversely affected by increased water level. The deterioration of salt marshes is, thus, most likely caused by subsidence. Any alteration in hydrology that increases either the duration or depth of flooding beyond normal limits will probably have a deleterious effect on the growth and survival of this species.

The relative contribution of saltwater intrusion or subsidence to marsh deterioration is difficult to evaluate since the two may occur simultaneously and their individual and combined impacts will depend on a number of factors. In many cases, saltwater intrusion may only result in a change in species composition. On the other hand, an increase in water level (with or without a change in salinity) has a greater potential for causing marsh deterioration since rapid colonization by an invading species (through seed) would require a period in which the marsh surface is exposed or where light requirements for germination are met. In areas where the minimum water level is increased to a point above the marsh surface, seed germination would be inhibited because of the continued presence of water over the marsh surface. Recolonization, in any case, would have to occur quickly to prevent erosion and further subsidence of the marsh surface. Succession to more salt tolerant vegetation types is possible (in areas where mean water depth does not increase) because of the existence of a range of species tolerant of salinities up to full sea-strength. Succession is less likely where salinity changes are accompanied by increased flooding levels because of (1) the requirements for seed germination and (2) salt tolerant species are not necessarily more flood tolerant than fresh marsh species. Thus, deterioration of a fresh or intermediate marsh may not only be due to an increase in salinity which initiated the process, but also to an inability of a more salt-tolerant species to establish itself or to tolerate the level of flooding if subsidence has occurred simultaneously with the salinity increase.

Salt marshes dominated by *S. alterniflora* are deteriorating partly because subsidence has increased water levels above that which this species can tolerate and the absence of another species which is more flood tolerant (but equally salt-tolerant) to replace it. In brackish, intermediate, and fresh marshes, deterioration will most likely occur in areas where the existing vegetation is rapidly eliminated, i.e., by an increase in salinity, water level, or other environmental change, and the marsh surface subsides or erodes to the point where recolonization is impossible. Recolonization by the original species or succession to more salt tolerant plant communities (in the case of a permanent change in salinity regime), however, may proceed in areas where propagule establishment can occur.

Because the vertical accretion of marshes is dependent upon the accumulation of organic matter produced by marsh macrophytes, any reduction in this source will slow the aggradation process. A sudden change in the environment which leads to a rapid reduction in biomass or even complete elimination of the emergent vegetation in a marsh would reduce the potential for the marsh accretion rate to keep pace with subsidence and/or sea level rise. The loss of belowground plant material, i.e., roots and rhizomes, that binds the sediment and provides stability would accelerate subsidence and break-up of the substrate. The rapid rate of subsidence in Louisiana's coastal zone (Baumann et al., 1984), in combination with the predicted sea level rise of 50 to 200 cm during the next 100 years (Titus, 1986), will lead to increased flooding stresses. The stresses associated with an increase in flooding depth and duration may ultimately result in the demise of emergent macrophytes in areas where vertical marsh accretion lags behind the increasing water level. Saltwater intrusion, whether natural or man-induced, may accelerate this process in fresh, intermediate, and brackish marshes.

Chapter 9

SALTWATER INTRUSION WORKING GROUP: CONSENSUS REPORT

by

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Clearly, the fundamental question we would like to answer is: has saltwater intrusion occurred in the Louisiana estuaries, affected plant growth, and subsequently resulted in enhanced land loss? The breadth of this question restricts our ability to answer it. We, thus, have posed a sequence of more manageable questions that we answer with varying degrees of reliability and completeness. After presenting these questions and answers, we will discuss the broader implications of these results for Louisiana land loss rates.

(1) Do man-made canals or man-modified bayous, as opposed to natural bayous, promote saltwater intrusion? Yes. Both field measurements and computer simulations show saltwater intrusion to be greater in the artificially deepened channels than in shallow, natural bayous.

(2) Does a change in the depth of the water course lead to further saltwater intrusion? Yes. A two-dimensional numerical model showed that artificially deepened channels allow greater saltwater intrusion than occurred prior to deepening.

(3) What are the forcing functions controlling saltwater distribution in a channel? Salinity distribution in channels is largely controlled by river inflow, tidal amplitude, and wind velocity.

(4) Are there secular trends in the statistics of long-term (years to decades) salinity records from the Louisiana estuaries? Yes. Statistically significant trends, both positive and negative, in mean salinity, salinity variance, and extreme events were observed. Only at a few stations, however, were the predicted net changes over the course of the record of such magnitude and sign that they might be expected to deleteriously affect the marsh plants. No consistent pattern of change across the entire coastal zone was observed.

(5) What relationship exists between salinities in the waterways, natural or anthropogenically altered, and in the adjacent marsh? Waterway salinities are, generally, neither consistently nor strongly coherent with salinities in the adjacent marsh at periods between 20 hours and 2 months. Concurrent records of marsh and waterway salinities longer than two months were not available to us.

(6) What is the mechanism of saltwater inflow to the marsh? Subsurface groundwater flow is not a significant contributor to the influx of salt water to the marsh on time scales shorter than the seasonal scale. Overbank flooding and flooding through breaks in either the natural levee or spoil banks is the important mechanism for transporting salt into the marsh. Once water has entered the marsh, its salinity will be altered by a number of processes which contribute to both its enhancement and decay:

groundwater flow from upland sources, evapotranspiration, mixing with existing interstitial waters, and dilution by rainfall.

(7) If an increase in salinity occurs, does it cause a decrease in plant growth? Yes. The effect of saltwater intrusion on marsh vegetation, though, depends upon species composition, rate of salinity increase, duration of salinity increase, net amplitude of salinity increase, and flooding depth. The growth of the salt marsh plant, *Spartina alterniflora,* was not significantly affected by salinities of 24 to 28 ppt. Leaf expansion was observed to be reduced in greenhouse experiments at 35 ppt, but the plant is known to grow vigorously in east coast marshes at full strength sea water, i.e. 36 ppt.

The growth of the brackish marsh species, *Spartina patens,* was significantly reduced when salinities were increased to levels approaching 24 ppt. The reduction appears greater when duration and abruptness of exposure to higher salinity levels are increased.

The fresh marsh species, *Panicum hemitomon and Leersia oryzoides,* were able to survive and grow (at a reduced rate) for one month at salinities of 8 to 11 ppt. *Sagittaria lancifolia* survived at salinities up to 4 to 5 ppt. However, a sudden increase in salinity to 15 ppt resulted in complete mortality of all three species.

(8) Does submergence, in itself, affect the growth of marsh plants? Yes. Field results demonstrated that all three marsh types, salt, brackish, and fresh, were negatively affected by greater submergence.

(9) Is the combined effect of salinity and submergence greater than that of either factor alone? Yes, for certain species.

Increased submergence will affect plant growth in all three major marsh habitats. Salt water intrusion, whether natural or man-induced, may accelerate this process in fresh and brackish marshes but not in salt marshes.

(10) Is there a difference in response to submergence and salinity as a function of marsh type? Yes. However, where species diversity is relatively higher (i.e., fresh marsh), differences in relative flood tolerance among species may influence the ultimate outcome of an increase in water level.

Deepening of the estuarine connection between the Gulf and the marsh, whether by natural (subsidence) or man-induced (dredging) processes, should result in saltwater intrusion up the estuary, all other variables being held constant. Model simulations suggest that adjustment within the estuary to new stationary conditions should take place within only a few tidal cycles. Careful examination of existing records, though, shows very little evidence of mean salinity trends of a size which would be detrimental to the marsh vegetation within the estuaries monitored. The absence of obvious trends could have resulted for a number of reasons. Altered runoff patterns could be counteracting the tendency for intrusion resulting from deeper channels. Natural variability could be such that existing records are not long enough to accurately estimate the real trends in the data set. The saltwater intrusion could be occurring below the level of the monitoring gauges. If salinity increases in the channels, though, it must find its way into the marsh proper before it can negatively impact the health of the vegetation. This net movement of salt water from a channel to the adjacent marsh appears to occur principally through overbank flooding, at least on time scales of months or less. Low topographic gradients and water level variations on a variety of spatial scales make the problem highly difficult to model.

Spoil banks, if they have any effect on this process, would probably reduce the occurrence of overbank flooding. Once a flooding event has occurred, though, the banks would also reduce the rate at which the water drains from the marsh, thus extending the time of submergence of the vegetation and the associated detrimental effects. Field and laboratory studies have demonstrated that the dominant plant in Louisiana salt marshes is tolerant to potential salinity increases, but sensitive to flooding events. The dominant brackish marsh species is affected by increased salinities. This effect is further aggravated by submergence. The fresh marsh plants studied were most sensitive to salinity increases; their response being dependent upon the amplitude, rate, and duration of change. Since increased salinity tends to occur in a marsh through increased water levels, it is difficult to consider the effects of salinity independently of submergence in the natural environment. Of particular concern is the question of succession. If salinity increases were to propagate into the marsh and kill or severely reduce the productivity of the vegetation, succession to a more salt-tolerant species would only occur if propagules were available and submergence and substrate erosion had not proceeded to a point where their establishment was precluded.

This information represents a significant improvement in our understanding of marsh–salinity interactions. These interactions are many and complex. With additional site-specific information, the sequence of events leading to marsh deterioration and future changes in salinity and vegetation will become more predictable.

PART IV

SUBSIDENCE AND SEDIMENTATION WORKING GROUP

Coastal wetlands develop through a complex interaction of geological, hydrological, and biological processes, and it is the balance or net effect of these processes considered in toto that determine wetland loss or gain. If the marsh surface sinks or the water level rises, the marshes must adjust to these changes or gradually become submerged. Marshes exist in very dynamic environments and continue to exist only if the rate of adjustment can keep pace with the rate of change. In this part of the report, the contribution of subsidence and sediment supply and distribution to wetland loss is analyzed. In particular, man's influence on these processes is evaluated.

This part consists of chapters on regional and local subsidence (Chapters 10 and 11), sediment supply (Chapter 12), and sediment distribution Chapters 13 through 17). The effort of this working group has been focused on answering the following question:

What is the contribution of organic and inorganic sedimentation in counteracting land loss (coastal submergence) on a local and regional basis and how do man's activities (OCS and non-OCS) influence this process?

These answers help define the first two of the five major questions:

> (1) If land is sinking more quickly than land is building and the rates of each process are changing, to what extent is this disparity caused by changes in: (1) sediment supply reaching the marshes; (2) organic matter accumulation; (3) subsistence rates; and (4) water level?
>
> (2) Do levee construction, canal dredging, and oil and gas production influence the rates of sedimentation, organic matter accumulation, and subsidence in coastal Louisiana? If so, how do these impacts contribute to the high rate of coastal submergence?

The results of all the chapters have been synthesized in a regional sediment budget model for Barataria Basin and is presented in Chapter 18.

Chapter 10

SUBSIDENCE AND SEA LEVEL

by

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The general purpose of this work was to estimate changes in the absolute land and sea levels in coastal Louisiana during the past 50 years. Specifically, the objective of the work was to answer the following questions:

(1) What portion of the high rate of coastal submergence is caused by changes in the subsidence rate?
(2) Have subsidence rates for south Louisiana changed since 1900?
(3) Does fluid withdrawal (e.g, oil, gas, formation water) influence subsidence rates in coastal Louisiana ?

Technical Approach

The approach taken to answer the questions presented above was based upon a combination of data analysis and modeling. The primary data base of the study, tide gage records and bench mark re-surveys, was already fixed. That is, the events being studied were historic, and there was no option to acquire new data such as the marsh elevations in 1940. However, the historic data base has problems: the data were limited to a few locations in coastal Louisiana; many data sets were incomplete; and, documentation of the data sets was spotty. Also, previous analyses of much of the data indicated that interpretation of the data was not straightforward; relative sea level changes resulted from the influence of several physical processes. In order to overcome the limitations of the existing data base, theoretical models were used (in some cases developed) to extend the knowledge of events or processes into areas not measured. Furthermore, use of theoretical models helped define the mechanisms involved (the causes behind the effects recorded in the historic data) and produced quantitative predictions of the relative importance of each factor when several were involved.

Subsidence Estimates

Sea Level

Data on sea level changes as measured by coastal tide gages were examined to separate eustatic sea level rise (absolute sea level rise) from subsidence. Subsidence is defined as the strain or vertical movement of the surface of a thickness of sediments resulting from the consolidation or density increase of the sediments over time. The values for subsidence were compared with changes in the elevations of bench marks and theoretical consolidation models so that any conclusions reached about the magnitude and causes of subsidence would be consistent.

The sea level data sets used in this study have been described in recent studies by the Louisiana Geological Survey (Penland et al., 1987). The data consisted of tide gage records taken at a number of locations along the coast of the northern Gulf of Mexico by the National Ocean Survey (NOS) and the U.S. Army Corps of Engineers (COE). The data result from water level readings that were taken every hour or once each day. The readings were averaged into mean monthly values of sea level; it is in this form that the data were used in this study. Previous analyses of these tide gage data have indicated that subsidence values in Louisiana are in the range of 10 to 20 mm/yr and have recently increased. Dating of geologic samples has indicated that the longer term (Holocene) accumulation rates of sediments is about 5 mm/yr. The purpose of the work reported here was to re-analyze the tide gage data with a scheme that would account for a factor not included in the published works, namely the effects of freshwater run-off.

The reason for considering the effects of fresh water run-off on tide gage readings is that most of these gages are located within the coastal marsh complex and are only partially affected by sea levels. It had also been observed that records of tide gage elevations that were corrected for eustatic sea level rise (sea level rise was subtracted from the records) showed periods when water elevations dropped, which represented an apparent rise in land elevation. While fluctuations in the rate of subsidence might be expected to occur over a decade, it is unreasonable to think that land levels are ever increasing. Finally, the last reason for considering the influence of fresh water or tide gage records was the relatively high subsidence rates reported for the tidal epoch of 1960 to 1980 including several years in which the Mississippi and Atchafalaya Rivers flooded. Thus, the influence of fresh water or tide records was analyzed, and the monthly tide gage records were examined in detail as a component of the data analysis.

The overall plan was to identify tide gage stations having a continuous record that included the two tidal epochs: 1940 to 1960 and 1960 to 1980. Because freshwater run-off was to be included in the analysis, stream flow records were needed for the same time periods. Several sets of tide gage records and basin stream flow data were obtained from USGS water resources publications, which met these criteria.

The first step in analyzing the water level data was to determine the eustatic sea level changes for the period of interest. This information was derived from two sources. A recent study (Barnett, 1984) analyzed sea level data for several stations in the Gulf of Mexico and presented a "coherent" sea level variation for the Gulf and the North Atlantic. Figure 10-1 shows this smoothed annual value of sea level that was coherent among several stations in the Gulf.

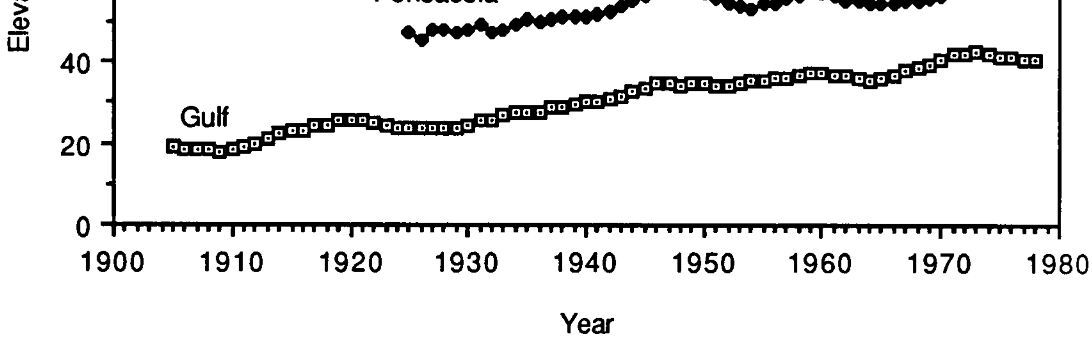

Figure 10-1. Sea level for Galveston, Pensacola, and Gulf of Mexico (5-year running mean).

The data was smoothed with a five-year running mean. Although not plotted, the annual data are listed in Table 10-1. The data in Figure 10-1 show that sea level is rising Gulf-wide at a rate of about 2.3 mm/yr when averaged over the whole record. There is little change in the underlying rate of rise, although there are periods of several years for which the rate accelerates (e.g., 1960s) and for which the rate is negative. While the data analyzed by Barnett (1984) did not include tide stations in Louisiana, it included Galveston, a site itself undergoing subsidence.

Table 10-1. Sea level variations for the Gulf of Mexico (Barnett, 1984).

Year	Sea Level (cm)	5-Year Mean (cm)	Year	Sea Level (cm)	5-Year Mean (cm)
1903	-7.6	-	1940	-1.0	-.5
1904	-13.8	-	1941	-1.2	0
1905	-12.0	-11.5	1942	1.5	.5
1906	-12.1	-12.2	1943	1.5	1.5
1907	-12.1	-11.8	1944	1.5	2.5
1908	-10.8	-12.3	1945	4.0	3.0
1909	-12.2	-12.6	1946	4.0	4.2
1910	-14.5	-12.1	1947	4.2	4.5
1911	-13.3	-11.6	1948	7.2	4.0
1912	-9.7	-10.8	1949	3.0	4.1
1913	-8.3	-9.2	1950	1.5	4.2
1914	-8.0	-7.9	1951	4.7	3.8
1915	-6.5	-7.4	1952	4.8	3.9
1916	-7.2	-7.2	1953	5.1	4.7
1917	-7.2	-6.1	1954	3.4	4.8
1918	-7.2	-6.0	1955	5.6	5.0
1919	-2.4	-5.1	1956	5.1	5.6
1920	-5.8	-4.6	1957	5.6	5.8
1921	-2.9	-4.5	1958	8.4	6.3
1922	-4.9	-5.4	1959	4.2	6.8
1923	-6.5	-5.9	1960	8.3	7.3
1924	-6.9	-6.9	1961	7.7	6.3
1925	-8.4	-6.8	1962	7.7	6.2
1926	-7.8	-7.1	1963	3.4	5.7
1927	-4.5	-6.9	1964	4.0	5.4
1928	-7.8	-7.1	1965	5.6	5.5
1929	-6.1	-6.4	1966	6.4	6.2
1930	-8.0	-6.4	1967	8.0	7.4
1931	-5.8	-5.0	1968	6.9	8.4
1932	-4.1	-5.0	1969	10.3	9.3
1933	-.9	-3.8	1970	10.3	10.5
1934	-6.1	-3.1	1971	11.1	11.9
1935	-2.0	-2.6	1972	14.0	12.0
1936	-2.5	-2.6	1973	13.6	12.3
1937	-1.7	-1.6	1974	11.0	11.8
1938	-.8	-1.4	1975	12.0	11.1
1939	-.8	-1.1	1976	8.4	10.7
			1977	10.4	10.3
			1978	11.9	10.1
			1979	8.8	-
			1980	11.0	-

To provide a second reference for eustatic sea level variations, the tide gage record for Pensacola, Florida, was obtained. The Pensacola record, also shown in Figure 10-1, is compared to the Galveston record and the Gulf data (Barnett, 1984). The Pensacola record shows more year to year variation than the Gulf data but is considerably smoother than the Galveston record. The Gulf annual sea level values were subtracted from the Pensacola and Galveston records, as shown in Figure 10-2. Figure 10-2 is important because it produces elevation records which reveal either changes in land elevation or the influence of local hydrodynamic factors, e.g.,wind set-up or freshwater run-off. The Galveston minus Gulf record clearly shows a consistent variation and indicates a rising relative water level, while the Pensacola record shows a much smaller variation, tending toward a lower relative sea level. Galveston is known from direct observation of land levels to be subsiding and the adjusted record indicates a rate of about 3.3 to 4.4 mm/yr. The Pensacola record is taken very close to Louisiana and might reflect Gulf water mass and weather influences on sea level that would affect Louisiana, therefore it was used as the reference station that defines eustatic sea level. The changes between the Pensacola record and records from locations in Louisiana should reflect Louisiana-specific conditions of subsidence and run-off. Subsidence may be occurring at the Pensacola site, but at a rate that is much lower than occurring in Louisiana and is caused by a different scale process. The Pensacola site reflects the effects of tectonic movement associated with the Northern Gulf of Mexico geo-syncline.

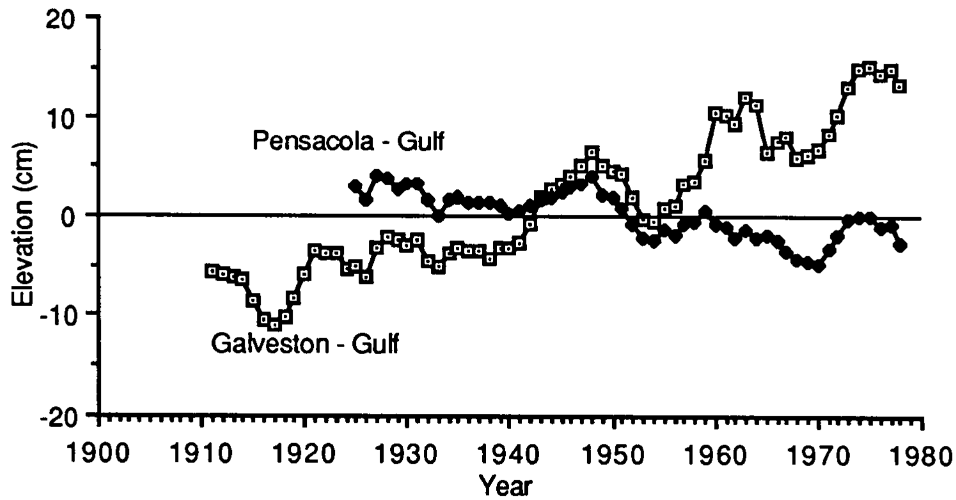

Figure 10-2. Difference between Galveston and Pensacola water levels and Gulf water level.

Tide Gage Records

The next step in the analysis was to determine the degree to which freshwater run-off might be affecting a tide gage record in Louisiana. The monthly values of water level for each year were used. In this way, the water level for a single month could be compared with water levels for other years (e.g., January 1945 with January in other years). An example of this type of graph is shown in Figure 10-3. The figure shows the monthly variation in water level at Galveston for five year averages. The curve for 1952, for example, is based upon the years 1950, 1951, 1952, 1953, and 1954. Figure 10-3 shows a well-defined seasonal pattern of water level that is being displaced upward by a small amount from one five year period to the next. The important feature of the records at this location is that the records are sub-parallel and have generally the same shape. The pattern of annual variations in water levels for a site at Cameron, Louisiana, shown in Figure 10-4, can be compared with that at Galveston. The water level at Cameron for the 1952 10-year average data set for January through July is much higher than for the same months in the 1962 data set. Thus, these water level records could not be solely the result of subsidence, since subsidence is not expected to vary within a twelve-month period. Another example of the annual variation in water level is at Morgan City, shown in Figure 10-5. For Morgan City, on the Atchafalaya River, the five-year mean annual variations are extreme. The water levels for 1967 and 1972 are much higher than the water levels for the years 1977 and 1982. This location is clearly influenced by the stage of the Atchafalaya River, and the high annual water levels from 1965 to 1975 are associated with flooding.

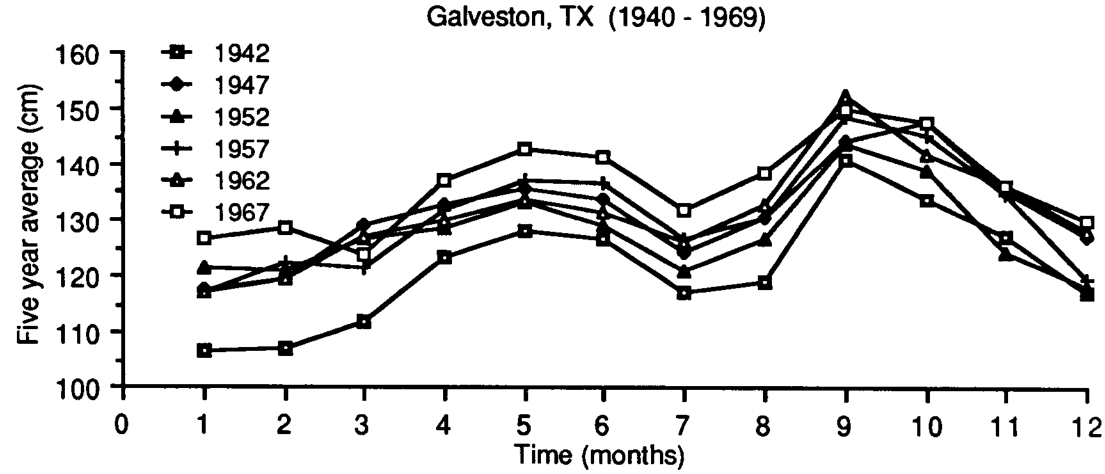

Figure 10-3. Monthly water level changes for Galveston, averaged in five-year intervals.

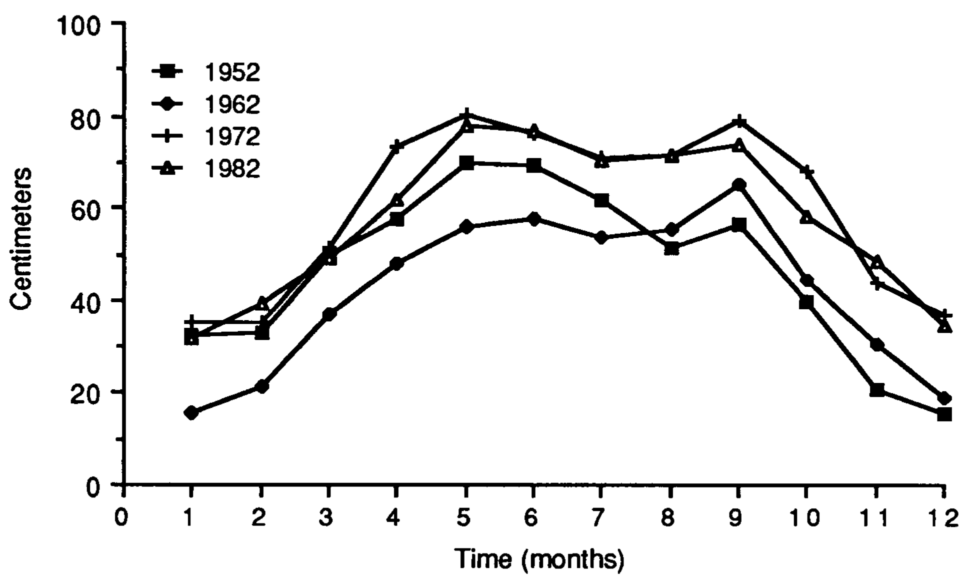

Figure 10-4. Monthly water level record for Cameron averaged in 10-year intervals.

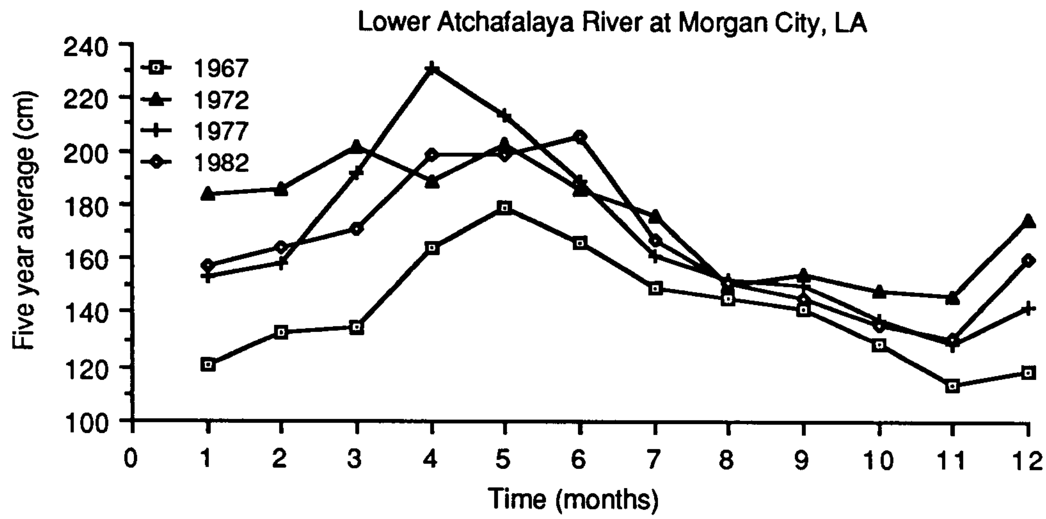

Figure 10-5. Monthly water level record for Morgan City, averaged in five-year intervals.

The second step in adjusting the tide gage records was to account for the influence of river discharge. The stage-discharge curves for the Calcasieu, Mermentau, and Atchafalaya Rivers and Bayou Lafourche were obtained. Figure 10-6 shows the curve for the Calcasieu River. The stage-discharge curve at each of the tide gage stations was not available, so a technique was developed to estimate stage-discharge that relied upon the character of the record at a location. The discharge of the waterway where the gage was located was plotted for each year versus the annual average water level. The average water level for each of several ranges in discharge was computed, resulting in an average stage-discharge value for the location. For the Cameron tide gage, a change in water level of 3 cm occurred for a change in discharge in the Calcasieu River of 3 m^3/sec. With this average stage-discharge value for each location, a second adjustment was made to the tide gage records. All the tide gage records were adjusted for discharge using the records for the rivers and bayous shown in Figures 10-7 and 10-8.

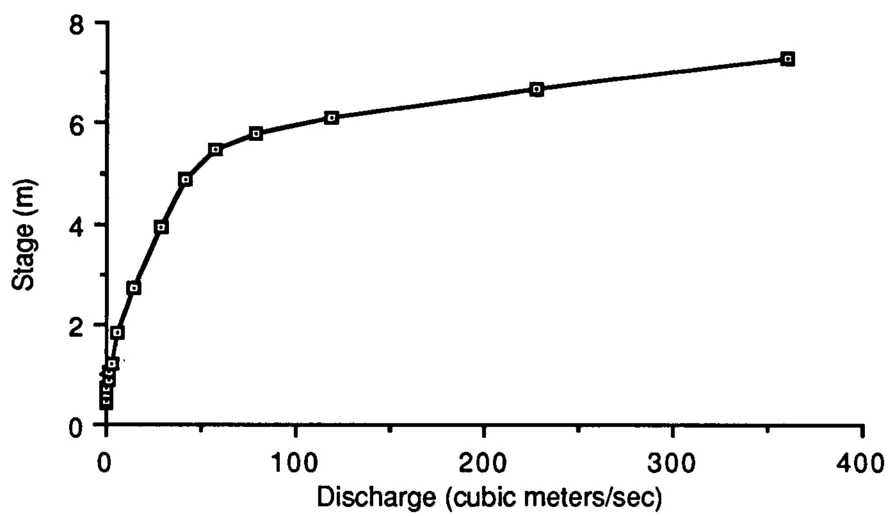

Figure 10-6. Stage discharge record for the Calcasieu River at Beckwith.

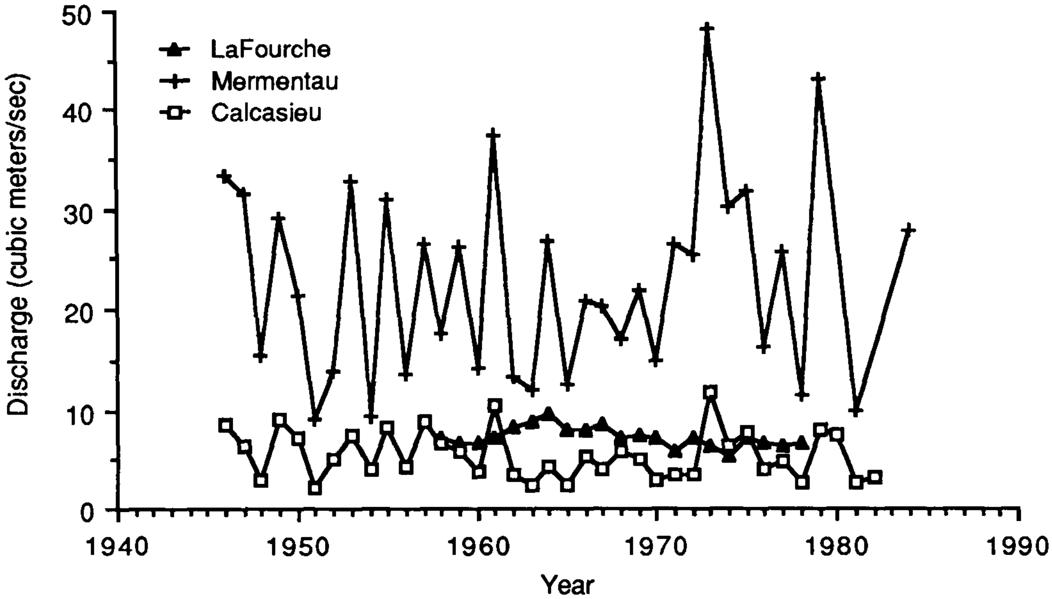

Figure 10-7. Annual discharge for Calcasieu and Mermentau Rivers and Bayou Lafourche.

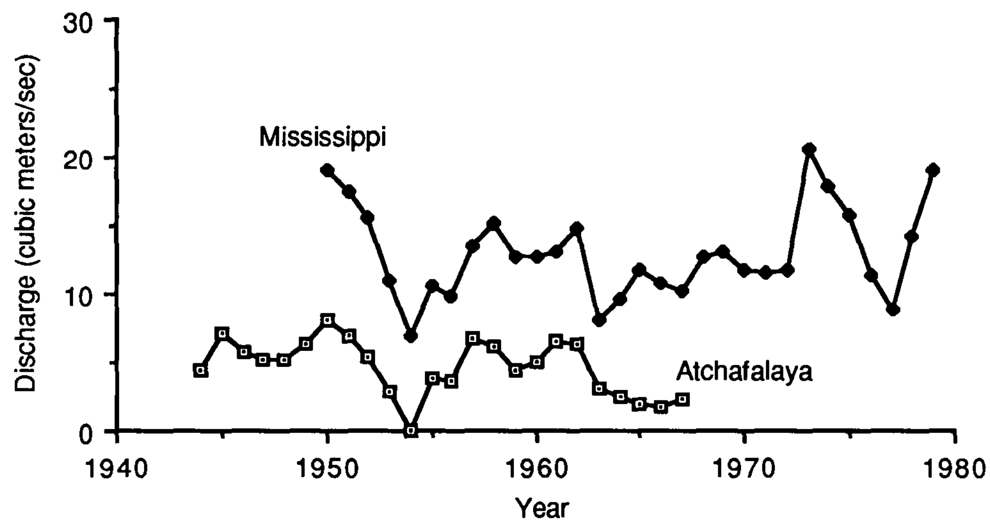

Figure 10-8. Annual discharge for Atchafalaya and Mississippi Rivers.

The results of the analysis of tide gage records on the Chenier Plain and in the Teche/Lafourche delta are shown in Figures 10-9 through 10-13. Each figure shows the annual water level for the station, with the water level corrected for the Pensacola eustatic variation and the water level corrected additionally for discharge. The lowest curve is the subsidence record for that station. The subsidence rates are constant over the two tidal epochs for most of the stations (the Grand Isle location shows a slight indication of a subsidence rate increase). The subsidence rates for each location are Cameron: 6.3 mm/yr;

Hackberry: 6.95 mm/yr; Morgan City: 13.3 mm/yr; Eugene Island: 10.1 mm/yr; and Grand Isle: 8.0 mm/yr. For each station, the R^2 value for the linear feet is given.

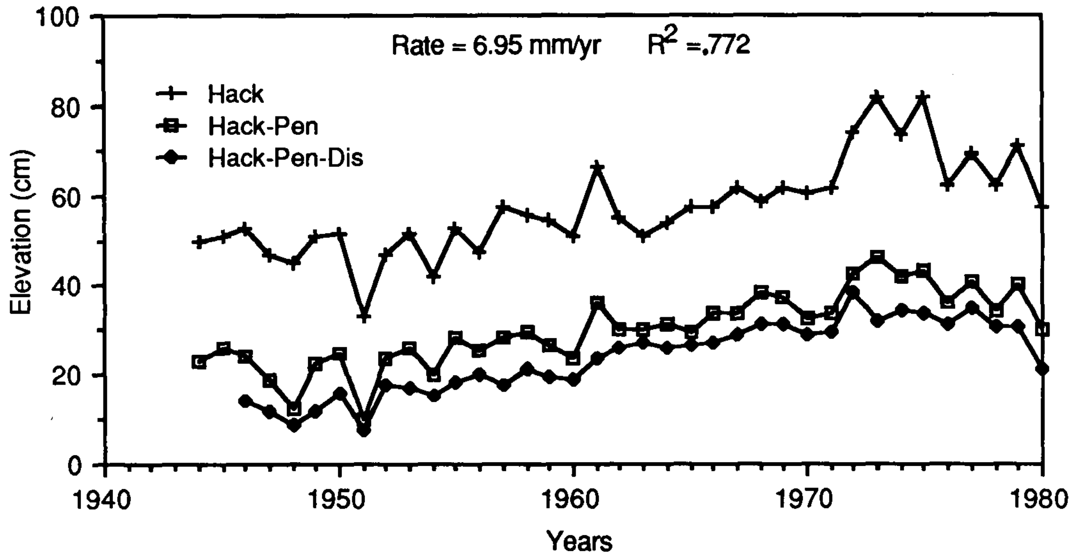

Figure 10-9. Subsidence record for Hackberry.

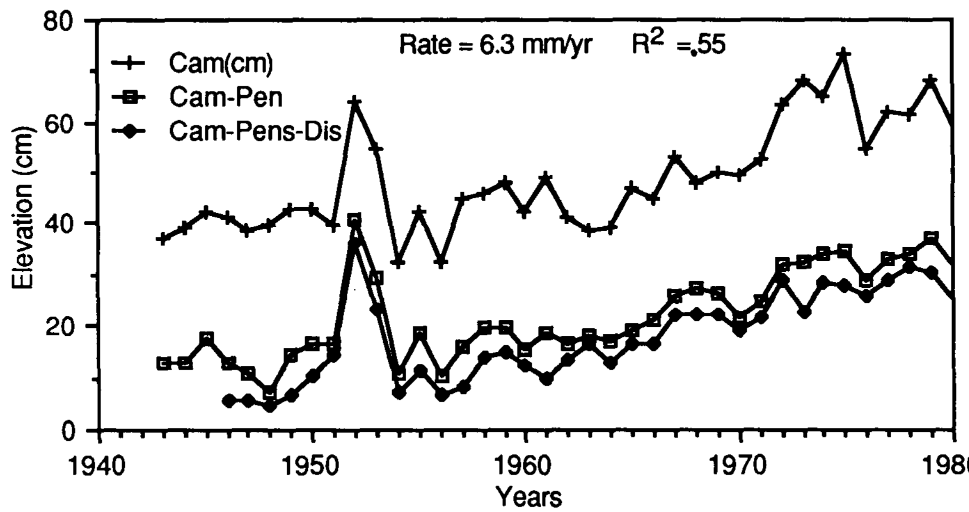

Figure 10-10. Subsidence record for Cameron.

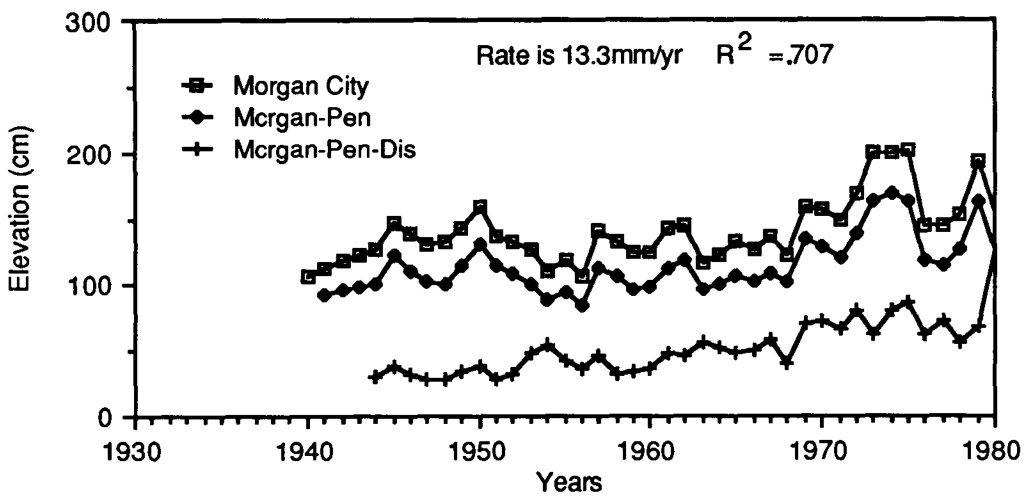

Figure 10-11. Subsidence record for Morgan City.

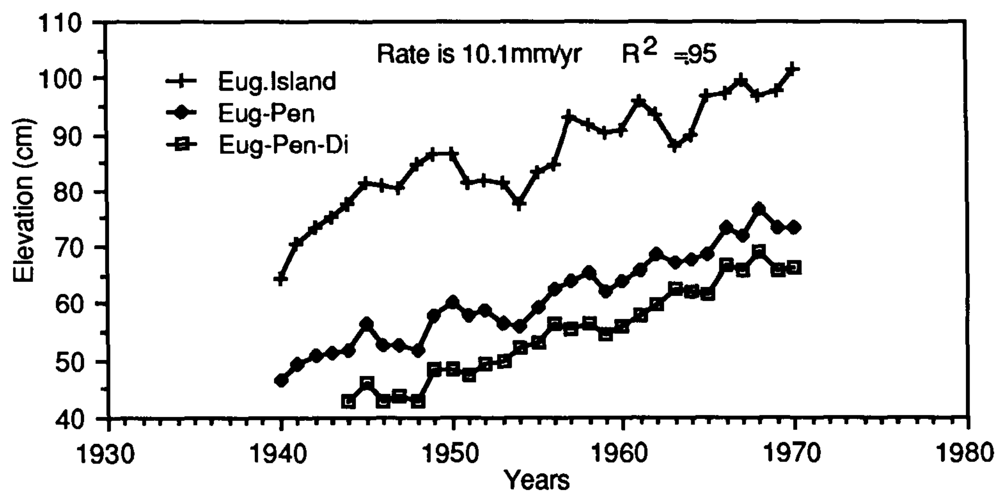

Figure 10-12. Subsidence record for Eugene Island.

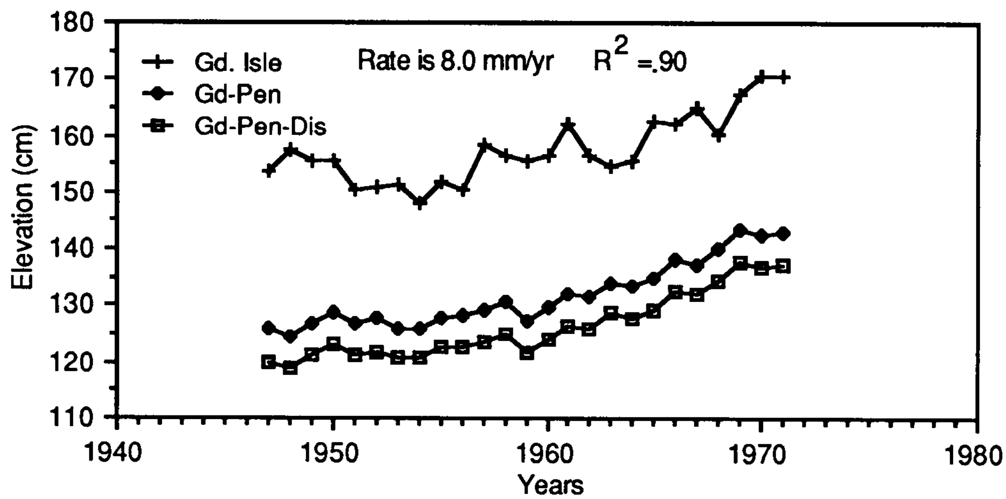

Figure 10-13. Subsidence record for Grand Isle.

Bench Mark Data

Several re-surveys of bench marks in Louisiana have been made by the National Geodetic Survey (NGS). These surveys allow another means for determining the subsidence in coastal Louisiana. The surveys provided by NGS include one line from Beaumont, Texas to New Orleans and a line from Raceland to Grand Isle. The data are summarized in Table 10-2. The subsidence rate on the Chenier Plain since 1955 is about 3 mm/yr and about 6 mm/yr on the Lafourche delta. The line from Raceland to Grand Isle shows an increasing rate from 0 mm/yr at Raceland to almost 9 mm/yr at Grand Isle.

The elevations of bench marks in Cameron Parish were adjusted in 1965 and 1983, as shown in Table 10-3. The differences in the bench mark elevations range from 15 to 25 cm, indicating a subsidence rate of 5.3 to 10.9 mm/yr.

Subsidence over Oil Fields

The effect on subsidence of the withdrawal of fluids from petroleum reservoirs was determined using a prediction model and oil/gas production data. The fluids withdrawn from reservoirs include crude oil, condensate, casinghead gas, natural gas, and water.

The first approach was based upon applying a model developed to estimate reservoir compaction. The model (Geertsma, 1973; Martin and Serdengecti, 1984) is based upon assuming a uniform reservoir having a bulk compressibility, Cm, that undergoes a reservoir pressure change of dP. The maximum compaction is given as

$$Cr = C_m \, dP \, H$$

where H is the thickness of the reservoir.

Table 10-2. Subsidence estimates based upon National Geodetic Survey lines in Louisiana.

	1918-1955		1955-1982	
Mile	mm	mm/yr	mm	mm/yr
Line - Beaumont to New Orleans				
0	-	-	-	-
50	40	1.1	-	-
100	80	2.2	100	3.7
150	0	0.0	80	3.0
200	50	1.4	80	3.0
250	50	1.4	0	0.0
300	120	3.2	160	5.9
350	120	3.2	160	5.9
400	140	3.8	-	-
450	60	1.6	-	-
Line - Raceland to Grand Isle				
0	-	-	0	0.0
20	-	-	0	0.0
40	0	0.0	30	1.8
60	40	1.3	60	3.5
80	140	4.7	50	2.9
100	20	.7	10	.6
120	270	9.0	150	8.8
Line - Biloxi to New Orleans				
Years	mm	mm/yr		
1938-1955	500	29.4		
1938-1971	600	18.2		
1955-1971	200	12.5		
1955-1976	350	16.7		

Table 10-3. Changes in elevations of bench marks in Cameron Parish.

Bench Mark No.	Elevation 1965(ft msl)	Elevation 1983 (ft NGVD)	Change (ft)
L 212	6.391	5.567	.82
M 212	4.596	3.897	.69
TT 154	4.301	3.522	.77
Q 212	5.679	4.999	.68
R 212	5.791	5.128	.66
S 212	5.791	6.099	.66
T 212	5.472	4.759	.71
U 212	3.812	3.253	.55
V 212	4.892	4.318	.57
W 212	5.925	5.370	.55
X 212	3.199	2.661	.53
Y 212	3.730	3.135	.59
TT 179	4.081	3.409	.67
Z 212	3.740	3.133	.60
A 213	4.619	4.111	.50

Cameron Bench Mark Surveys: 1965 and 1983. Rate of Change: 8.3 to 13.9 mm/yr
Estimated subsidence: 5.3 to 10.9 mm/yr. Range in Elevation Change: 0.50 to 0.82 feet
Sea Level Change Assumed Included in Surveying: 3.0 mm/yr

For a disc-shaped reservoir, the profile of the surface subsidence above the reservoir, Uz, is given as a function of the radial distance from the center of the reservoir, r, by

$$Uz(r) = -2\ C_m\ (1-\nu)\ dP\ H\ A(\rho,\eta)$$

where ν is Poisson's ratio and A is a function of the dimensionless variables ρ and η; ρ is the ratio of the radial distance, r, and the radius of the reservoir, R; η is the ratio of the depth of burial of the reservoir, D, and the radius of the reservoir, R. The function A has a maximum value of 1 at $\rho=0$ and $\eta=0$ and decreases as ρ and η increase. The model predicts that the subsidence above a reservoir can be larger than the reservoir compaction,

$$Uz(0)/Cr = 1.5\ A$$

Data were acquired from the Louisiana Department of Conservation, Summary of Field Statistics for 1973. The report contained a listing of the depth to the top of the producing formation, the areas and thicknesses of oil and gas components of each field in Louisiana. From this data, fields were selected for which the reservoir was less than 1,000 m (3,000 ft) subsurface. The oil fields selected for analysis are listed, along with the field data, in Table 10-4. The total area of the 32 fields listed is 620,000,000 m^2 (166,490 acres) or about 620 square km.

A value for C_m of 10^{-6} (1/kPa) was used for the one dimensional compressibility in the calculations of subsidence. This value is given by Martin and Serdengecti (1984) for unconsolidated sandstone at a depth of 1,000 m and is conservative by perhaps a factor of 2. Poisson's ratio was assumed to be 0.3.

In order to compute the subsidence, data were needed concerning the pressure drops for each field. These data were not readily available, and they were not available for each field or well in each field. However some pressure data were available for a few wells in a few fields. For example, in the Lake Pelto field, the pressure in a well at a depth of 1,878 m (6,160 ft) dropped from 900 kPa (450 psi) in February, 1984, to 280 kPa (140 psi) in August, 1986, and then increased to 500 kilopascals (kPa) (250 psi) by April, 1987. A well in the Leeville oil field, at a depth of 882 m (2,894 ft) changed from 1,190 kPa (595 psi) in January,1984, to 210 kPa (105 psi) in January, 1986. A second well in the Leeville field, at a depth of 767 m (2516 ft) changed in pressure from 1,400 kPa (700 psi) in May, 1984, to 300 kPa (150 psi) in January, 1985. A well at a depth of 1,058 m (3,470 ft) changed in pressure from 600 kPa (300 psi) in December 1965 to 350 kPa (175 psi) in November 1969.

Because pressure data were not available for each field, a pressure drop of 1,000 kPa was used in the subsidence calculation for each field. This value is 5% of the value used in Martin and Serdengecti (1984) for maximum reservoir pressure drops in Louisiana. It is about twice the pressure change recorded for the above mentioned wells over just a few years.

Results of the estimates of the subsidence potential for oil fields are given in Table 10-4. The largest subsidence potential (SP) is estimated for the Lake Washington field at 86.1 cm. The subsidence as a function of distance from the center of the field is given in Table 10-5. The subsidence extends to a distance of about one-and one-half field radii from the center of the field. There are 19 fields with a subsidence potential greater than 10 cm, considered to be a cut-off value for the model being used. These results should be conservative because pressure drops larger than assumed in the calculation have been recorded.

Table 10-4. Shallow oil fields in southern Louisiana. Areas and thickness are given for both oil and gas as oil/gas.

	Depth (ft)	Area (ac)	Thick. (ft)	SP (cm)
Formation Depth 0 to 1000 ft				
Iberia	800	2480/ 1300	400/ 18	15.4
Charenton	960	4380/ 1800	348/ 87	13.4
Garden Island Bay	788	4200/ 1000	1550/ 360	53.2
Potash	688	1120/ 2030	458/ 193	7.4/15.7
Formation Depth 1000 to 2000 ft				
E Golden Meadow	1406	160/ 1760	20/ 50	1.5
Leeville	1787	6520/ 4520	1201/ 201	41.2
Valentine	1103	2740/ 6360	465/ 587	22.6
Lake Pelto	1350	3760/ 2620	540/ 400	18.5
W. Lake Verret	1300	4480/ 1520	310/ 82	11.9
W. Cote Blanche Bay	1728	12160/ 2500	317/ 135	12.2
Vinton	1880	2830/ 220	135/ 135	4.1
Cameron Meadows	1269	820/ 720	37/ 140	3.8
W. Hackberry	1958	2970/ 1520	39/ x	1.2
Welsh	1200	990/ 1310	15/ 15	.5
Lake Washington	1114	9500/ 5700	2507/ 386	86.1
Bay Marchand 2	1684	14500/ 3900	1825/ 250	70.3
Grand Isle 16	1539	10000/ 2000	910/ 100	35.0
Grand Isle 18	1738	2920/ 400	570/ 20	17.4
West Cameron 33	1671	40/ 1000	20/ 160	4.3
Formation Depth 2000 to 3000 ft				
Golden Meadow	2544	9520/ 2860	365/ 125	12.5
Bay St. Elaine	2574	7800/ 4620	610/ 625	20.9
Caillou Island	2618	16520/ 9040	1600/ 520	54.9
Dog Lake	2276	4000/ 1700	255/ 146	7.8
Gibson	2403	1520/ 6370	230/ 170	5.5
Abbeville	2658	1700/ 3880	240/ 150	5.7
E. Hackberry	2700	3000/ 1120	60/ 60	1.6
Bay De Chene	2460	2080/ 1200	280/ 110	6.7
Southeast Pass	2674	2680/ 1600	105/ 50	2.8
Venice	2345	3960/ 880	1650/ 320	44.3
Breton Sound 36	2820	x / 1420	x/ 70	1.5
Main Pass 69	2244	10200/ 3500	1440/ 140	49.4
Ship Shoal 23	2603	80/ 1080	x/ 120	2.2

Table 10-5. Estimated subsidence profile above Lake Washington field.

Radius (m)	Subsidence (cm)
0	86.1
250	86.1
500	85.7
750	85.4
1000	85.4
1250	84.4
1500	83.4
1750	81.0
2000	81.0
2250	78.7
2500	73.1
2750	67.4
3000	67.4
3250	54.2
3500	41.0
3750	28.7
4000	28.7
4250	16.5
4500	12.1
4750	7.7
5000	7.7
5250	6.0
5500	4.3
5750	3.5
6000	3.5

Because field specific data were not available for each site, a second approach to determine subsidence resulting from fluid withdrawal was to consider the field production. Data were acquired from the Annual Oil and Gas Report of the Department of Conservation, State of Louisiana concerning production figures for 1954, 1965, and 1984. The 1984 data for the largest fields, Bay St. Elaine, Leeville, Lake Pelto, and Golden Meadow, are given in Table 10-6. The table shows the total production of reservoir fluids and does not include water removed. Water removal was found to vary from 1 to as much as 15 times the crude oil removed.

Table 10-6. Oil field production for major fields, to the end of 1984. Note: mbbl = million barrels; mmcf = million million or 10^{12} cubic feet. Figures do not include formation water removed.

	Golden Meadow	Lake Pelto	Bay St. Elaine	Leeville
Crude (mbbl)	131.7	114.7	162.6	139.1
Con. (mmcf)	6.0	4.3	8.0	1.0
Cas. Gas (mmcf)	99.8	213.1	274.5	160.7
Nat. Gas (mmcf)	163.0	188.7	596.3	104.6

The volume of the fluids removed from a reservoir allows a second estimate of subsidence. The compaction of the oil reservoirs that occurs because a volume of fluid has been removed from the reservoir per unit area gives an estimate of surface subsidence, assuming the surface moves downward to replace the volume removed. The volume of only crude oil per unit area for the above mentioned fields was computed, resulting in the following estimates of subsidence: Lake Pelto was 130 cm; Leeville: 91 cm; Bay St. Elaine: 89 cm; and Golden Meadow: 59 cm. These values do not include the effects of withdrawal gas or formation water nor do they reflect seepage or the pumping of water into the formation.

Spoil Bank Compaction of Marsh

Surveys were made of several natural and man-made spoil banks in the Terrebonne marsh to investigate the association between spoil banks created by man-made channels and the appearance of open water. The results of the surveys were analyzed for the effect of the load on marsh sediments.

The sediments near the surface of the marsh are dominantly clay with a high water and organic content. The dry density of the marsh sediments is 0.15 to 0.25 g/cc. These types of sediments are highly compressible, with a compression index, Cc, 100 times that of clay. Few data are available to describe the critical sediment properties that control the response to applied loads. Data were used to estimate that the marsh has a one-dimensional compaction coefficient of 0.21/kilopascals (kPa). The spoil bank material was assumed to have a dry unit weight of 4,000 newtons (N)/m^3. For a spoil bank 1 meter high, the downward depression of the marsh surface is predicted to be 34 cm, using the reservoir compaction model of Geertsma (1973). The bank was assumed to have a width of 7.68 meters. The depression caused by the spoil bank is predicted to extend a distance of only 1 meter beyond the edge of the spoil bank. In this area the downward depression of the marsh surface is predicted to be about 10 cm.

Although the spoil bank subsidence model is very rudimentary, it allowed field measurements of spoil bank geometry to be interpreted. Surveys of spoil banks at three sites in the Terrebonne marsh were conducted, as shown in Figure 10-14. Two of the spoil banks were along dredged channels, profiles at Raccourci and Superior Canals. The probe at Raccourci Bayou was across the bank of a natural channel. There is a slight depression in the profiles at the Raccourci Canal site of about 6 to 8 cm. At the highest spoil bank site, at Superior Canal, there is a slight downward tilting of the marsh sediment elevation , at distances from 11 to 15 m. The downward depression is only a few cm. The spoil bank surveys seem to suggest that a downward displacement of the marsh surface has occurred; however the magnitude of the displacement for the surveyed spoil banks is at the threshold of the surveying precision.

Summary

The results of the analysis of sea level and subsidence data indicate that subsidence is the dominant process causing wetland inundation. The eustatic sea level rise appears to have been relatively constant over the last 80 years at a rate of about 2.3 to 2.8 mm/yr. This rate has varied over decade long periods from a sea level decrease of 3 mm/yr to a maximum increase of about 10 mm/yr. Basin water level changes significantly influence marsh water levels and have produced decade long rises and lowerings of water levels at tide gage stations by as much as 60 cm. Subsidence, the compaction of coastal sediments, has been in the range of 3 to 10 mm/yr, or about 100 to 300% of the eustatic sea level rise.

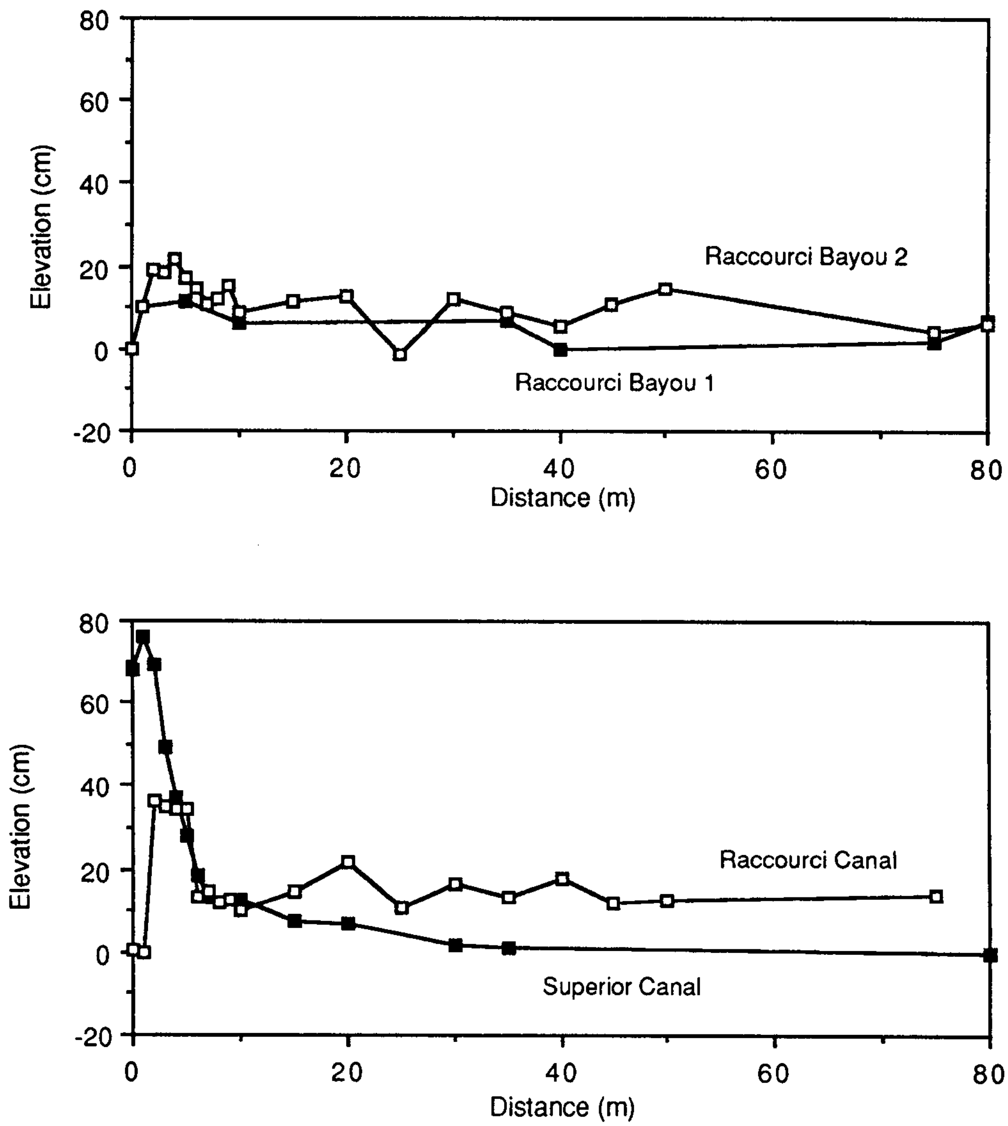

Figure 10-14. Elevation survey data. Top graph shows natural bayou; bottom graph show canal.

These subsidence rates do not seem to have varied during the last 40 to 50 years, in general. Fluid withdrawal from oil/gas reservoirs appears to have a localized influence on subsidence, amounting to as much as 80 cm of settlement directly above the reservoirs. The total area of oil/gas field having a subsidence potential greater than 10 cm is about 400 km^2.

Considering the wetlands of Louisiana as whole, subsidence is the major long-term component of relative water level rise in the wetlands, with eustatic sea level rise and basin water level changes being of secondary importance. On this scale, oil and gas production does not seem to have significantly influenced subsidence.

Chapter 11

TIDE GAGE RECORDS, GEOLOGICAL SUBSIDENCE, AND SEDIMENTATION PATTERNS IN THE NORTHERN GULF OF MEXICO MARSHES

by

R. Eugene Turner
Center for Wetland Resources

Tide gage records from the Gulf of Mexico can be used to estimate eustatic sea level rise (Gornitz et al., 1982; Hicks et al., 1983), geological subsidence due to compaction and crustal downwarping (Swanson and Thurlow, 1973), and the disparity between water level rise and vertical accretion of sediments (Baumann, 1980; Delaune et al., 1978; 1986). Analyses of tide gage records for the Gulf of Mexico wetlands have led to conclusions about increases in sea level rise and geological subsidence (Penland et al., 1987; Ramsey and Moslow, 1987), less vertical accretion than relative sea level rise (Delaune et al., 1986), related secondary interpretations about wetland loss and management (Templet, 1987), and estimates of how much oil and gas fluid withdrawal contributes to geologic subsidence (Penland et al., 1987). These latter analyses generally examined tide records of 20 years or less, thus excluding discussion of longer-term variations. It was the purpose here to analyze the available tide gage records to (1) determine sea level rise and settling rates for this century (i.e., geologic subsidence), (2) assess long-term water level variations, and, (3) re-evaluate estimates of the balance between eustatic water level changes and sedimentation rates. These analyses were basically accomplished by establishing coherence between annual water level for records throughout this century, principally at Galveston, Texas, and the many shorter records.

Methods and Materials

Tide gage records from 55 coastal stations (Table 11-1) were analyzed with the assistance of K. Ramsey and S. Penland, Louisiana Geological Survey (LGS), by cross-checking and selecting only reliable data sets from two technical reports by the LGS (Penland et al., 1986, 1987). The two LGS reports include monthly summaries of the original daily records collected by the U.S. Army Corps of Engineers (COE) and the National Oceanic and Atmospheric Administration (NOAA). Discrepancies between LGS and this analysis were resolved by examination of the original records in consultation with LGS. All staff gage readings were corrected to adjusted mean sea level, per notations in the original individual gage records. Several station records contained information to the effect that a nearby water control structure influenced water levels through pumping stations, gates, weirs, or locks; these latter stations were therefore not analyzed further. An estimate of mean global sea-level change is from Barnett (1984).

A mean yearly average of the monthly means for each gage was computed and annual coherence among stations determined using correlation analyses. The longest record of several stations was analyzed for long-term trends in annual mean water level, and for 10- and 18-year running average water level. Simple linear regression of one gage record against another was completed with a Statview 512+ microcomputer program.

Table 11-1. Tide gage record location and record length.

Station Number	Name	R^2	Years	Comparison to Galveston station: Interpreted Slope #23 vs others	Comparison to Galveston station: 72 yr water water rise rise cm/yr
1	Calcasieu R. & Pass near Cameron, LA	0.78	38	1.17	0.73
2	Calcasieu R. & Pass at Hackberry, LA (Cameron Parish)	0.76	37	1.15	0.72
3	Mermentau R. at Catfish Point Control Structure (S.Auto)	0.76	31	1.14	0.72
4	Schooner Bayou at Control Structure (E. Auto)	0.75	38	0.96	0.61
5	Schooner Bayou at Control Structure (E. Staff)	0.82	37	0.96	0.60
6	Intercoastal Waterway at Vermilion Lock (E. Auto)	0.78	38	0.85	0.54
7	Intercoastal Waterway at Vermilion Lock (E. Staff)	0.72	37	0.85	0.54
8	Intercoastal Waterway at Vermilion Lock (W.)	0.58	38	1.11	0.70
9	Wax Lake Outlet at Calumet	0.77	38	3.37	2.12
10	Atchafalaya Bay near Eugene Island - COE	0.71	4	1.69	1.07
11	Lower Atchafalaya River at Morgan City, LA	0.67	47	2.02	1.27
12	Bayou Black at Greenwood, LA	0.84	35	2.00	1.25
13	Intercoastal waterway at Houma, LA	0.51	34	•	•
14	Bayou Rigaud at Grand Isle, LA (#1 of 2)	0.53	31	•	•
15	South Pass Bar Near Port Eads, LA	0.77	32	2.04	1.28
16	Lake Pontchartrain Near South Shore	0.91	31	1.22	0.77
17	Lake Pontchartrain at Little Woods, LA	0.84	45	1.42	0.89
18	Lake Pontchartrain at West End. LA	0.60	47	0.68	0.43
19	Lake Pontchartrain at Frenier	0.48	47	•	•
20	Lake Pontchartrain at Mandeville, LA	0.75	49	0.70	0.44
21	Atchafalaya Bay near Eugene Island, LA COE	0.85	33	1.35	0.85
22	Bayou Rigaud at Grand Isle, LA (#2 of 2)	0.83	28	1.07	0.67
23	Galveston, TX	1.00	73	1.00	0.63
24	Port Isabel, TX	0.84	34	0.58	0.37
25	Gulf of Mexico at Biloxi, MS	0.25	27	•	•
26	Pensacola, FL	0.75	47	0.42	0.26
27	Cedar Key, FL	0.78	53	0.32	0.20
28	St. Petersburg, FL	0.71	33	0.31	0.20
29	Key West, FL	0.76	67	0.33	0.21
30	Charenton Drainage Canal near Floodgate	0.45	34	•	•
31	Charenton Drainage Canal at Baldwin, LA	0.10	38	•	•
32	Charenton Drainage Canal at Mud Lake, SW of Baldwin, LA	0.08	9	•	•
33	Bayou Sale at Luke's Landing	0.91	23	1.78	1.12
34	Wax Lake vicinity of Belle Isle	0.72	7	2.87	1.80
35	Wax Lake West Drainage area at control structure	0.31	25	•	•
36	Intercoastal waterway at Wax Lake W. control structure	0.84	25	2.45	1.54
37	Bayou Teche at W. Calumet Floodgate	0.88	29	1.44	0.91
38	Six mile Lake at Verdunville, LA	0.77	31	4.34	2.73
39	Round Bayou at Deer Island , LA	0.07	7	•	•
40	Lower Atchafalaya River below Sweet Bay Lake	0.72	24	1.91	1.20
41	Lower Atchafalaya River at Berwick Lock (W.)	0.81	24	1.94	1.22
42	Bayou Boeuf at B. Boeuf Lock (E.), Near Morgan City, LA	0.79	26	1.46	0.92
43	Belle River near Pierre Pass, LA	0.64	23	1.60	1.00
44	Lake Verret, Attakapas Landing, LA	0.88	25	2.08	1.31
45	Bayou Lafourche at Donaldsonville, LA	0.40	20	•	•
46	Bayou Boeuf at Amelia, LA	0.29	37	•	•
47	Bayou Black at Gibson, LA	0.78	10	2.34	1.47
48	Bayou Lafourche at Thibodaux, LA	0.01	11	•	•
49	Bayou Petit Caillou at Cocodrie, LA	0.78	11	0.99	0.62
50	Bayou Lafourche at Valentine, LA	0.84	14	1.54	0.97
51	Bayou Cheureuil near Chegby, LA	0.71	23	1.16	0.73
52	Bayou des Allemands at des Allemands, LA	0.60	30	0.84	0.53
53	Bayou Blue near Catfish Lake, LA	0.85	5	0.90	0.56
54	Bayou Lafourche at Golden Meadow, LA (oilfield)	0.86	19	1.99	1.25
55	Bayou Lafourche at Leesville, LA	0.80	23	0.87	0.55

Results and Discussion

Mean Annual Water Levels

The Galveston, Texas, tide gage records of water levels were used extensively as a surrogate for investigating long-term changes at other stations because of its long record (1908-1980) and coherence with those other gage records. Water levels at Galveston increased at 0.62 cm/yr for the last 73 years. A 10- and 18-year running average of water levels showed relatively constant variation about the mean, a constant increase, but no permanent acceleration over that time period (Figure 11-1). These results do not contradict the finding by Penland et al. (1986, 1987) that sea level rise from 1942-1962 was well below that from 1962-1980, but puts those changes from one lunar epoch (18.6 years) to another into an historical perspective. Penland et al. (1987) indicated that the intervals 1942-1962 to 1962-1982 were periods of increased water level rise, or acceleration, in the northern Gulf of Mexico. This acceleration is observed for the Galveston tide gage record (from 0.32 to 1.15 cm/yr, respectively; Figure 11-2). However, from a historical perspective, this acceleration may be short-lived since both acceleration and de-accerelation are observed to be within the historical variation. In other words, the acceleration in mean water level rise from 1942-1982 may well be temporary and retire to the lower rates of rise that will continue to fluctuate about a long-term mean. This is not to suggest that water level rise at Galveston will not eventually accelerate due to increased amounts of atmospheric carbon dioxide; the currently-available record at these stations does not conclusively indicate acceleration, but, rather, variation about a long-term mean sea level rise. Galveston water level rise will decline for this decade if the historical pattern continues.

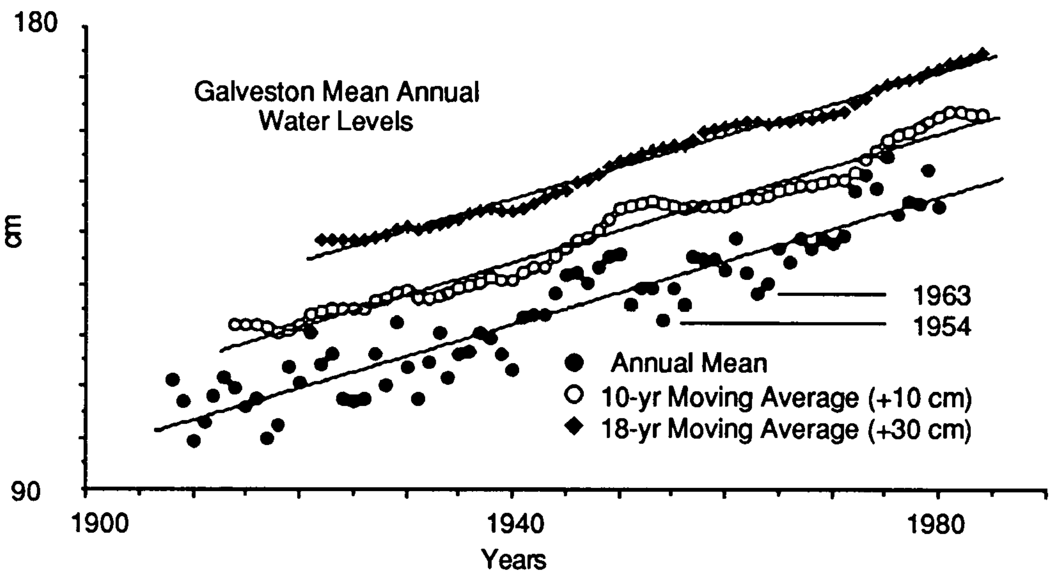

Figure 11-1. The annual water level at Galveston, Texas, from 1909 to 1980. Three different means are shown: the annual mean, a 10-year moving average of the annual mean, and an 18-year moving average of the annual mean.

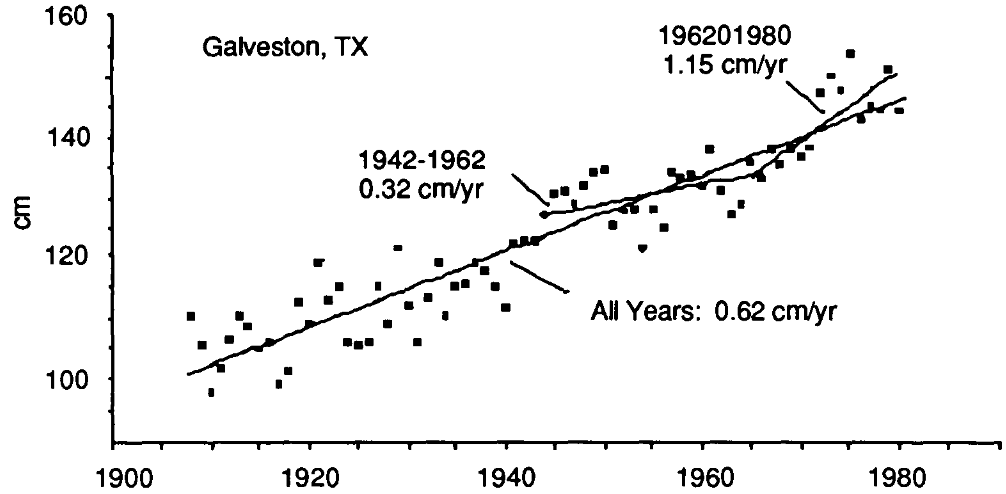

Figure 11-2. Annual changes among years at Galveston Bay, Texas. Three different linear relationships are shown: for all years of record, for 1942-1962 and for 1962-1982.

Water levels at the Galveston tide gage are well correlated with those from the west coast of Florida (Figure 11-3), which is considered a relatively geologically stable calcium carbonate platform and therefore subsiding little, if at all. A similar linear relationship is observed with the global mean sea level (Figure 11-4). These constant relationships between annual water levels at Galveston and elsewhere support the hypothesis that mean annual water level rise across the Gulf of Mexico and the world oceans are changing at relatively constant rates and are not accelerating. If geological subsidence at Galveston was accelerating, for example, then the relationship shown in Figures 11-4 would be curvilinear, not linear.

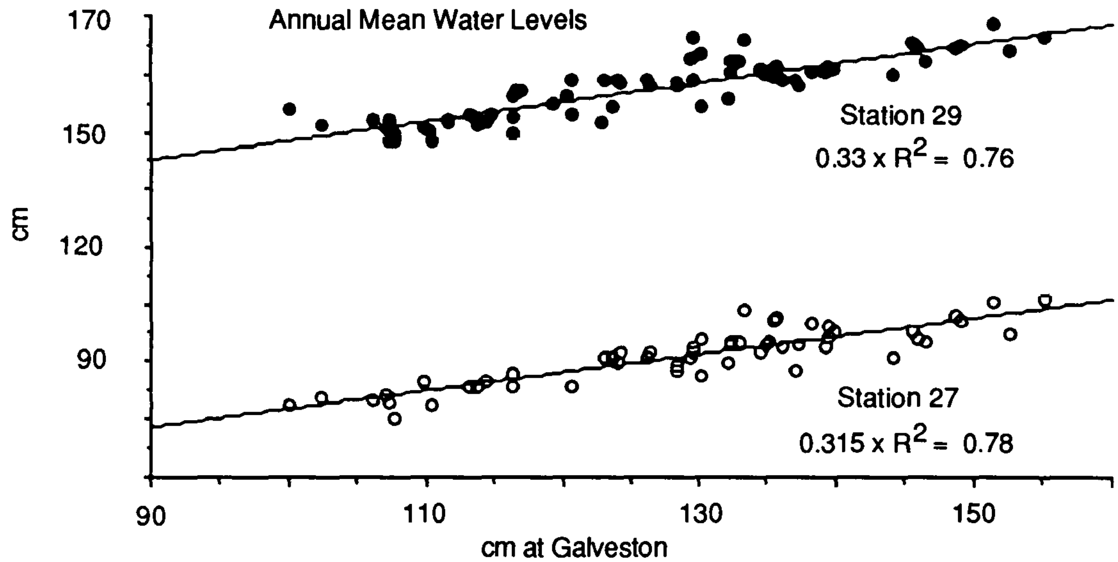

Figure 11-3. Mean annual water levels at Galveston compared to Cedar Key, Florida (Station 27; 1914-1980) and Key West, Florida (Station 29; 1913-1980).

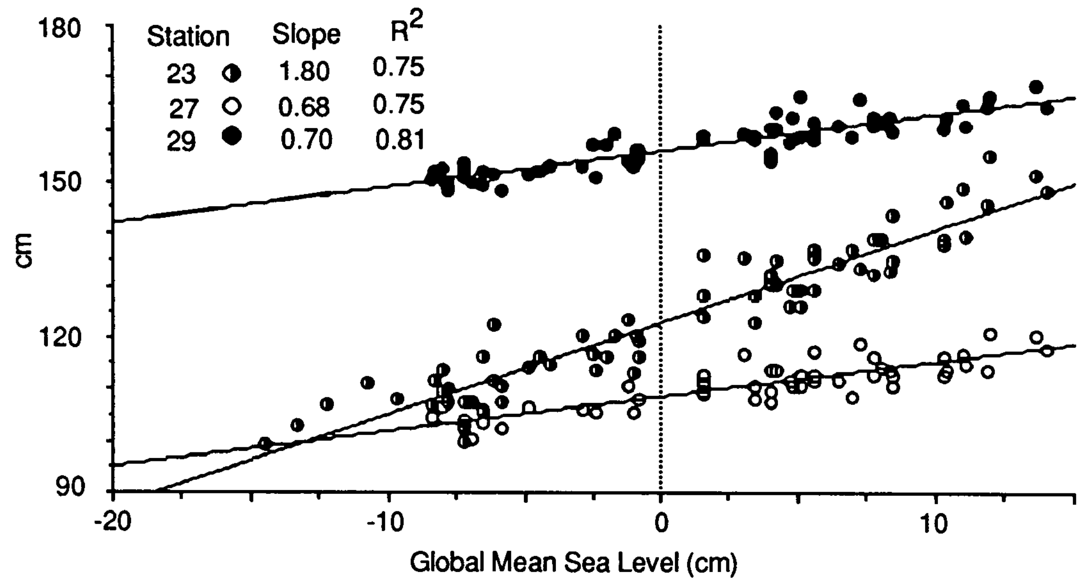

Figure 11-4. Tide gage water levels at three stations in the northern Gulf of Mexico compared to the mean global sea level (from Barnett, 1984).

Annual water levels at Galveston and Louisiana tide gages are similarly related, though at different ratios. Examples will be discussed at specific locations below. Lack of coherence was observed only at stations with short tide gage records (less than 10 years), where pumps or tidal gates were active, at one gas and oil field, or, where the newly emerging Atchafalaya Delta is growing near a major river distributary.

The Galveston tide gage record is well correlated with the shorter-term record of annual water levels in southwestern Louisiana (Figure 11-5), and Barataria Bay, Louisiana (Figure 11-6), located just west of the modern Mississippi River delta. The Galveston tide gage record can therefore be used as a surrogate for relative changes at other gages where coherence between annual water levels is high. Table 11-1 contains pertinent data on the correlation between annual mean water levels at Galveston and other stations and an estimate of the long-term water level rise based on those correlations. These estimated values range from about 0.2 to 2.7 cm/yr for all stations, and average about 0.8 cm/yr for all Louisiana stations. In general the highest rates of water level rise are where recent sedimentation is the highest (at deltas) and where waterway construction and water management is most intense (at locks). The lowest rates tend to be where the depth to the Pleistocene terrace is shallowest (estimated from Frazier, 1967) and where sedimentation is lowest.

Based on the above analyses, the percent water level rise due to eustatic sea level changes (estimated from the Florida tide gage records) can be calculated as a percent of the total relative change. For this century, eustatic sea level rise is 32% of the total water level rise at Galveston and ranges from 29 - 45% for the stations shown in Figures 11-5 and 11-6.

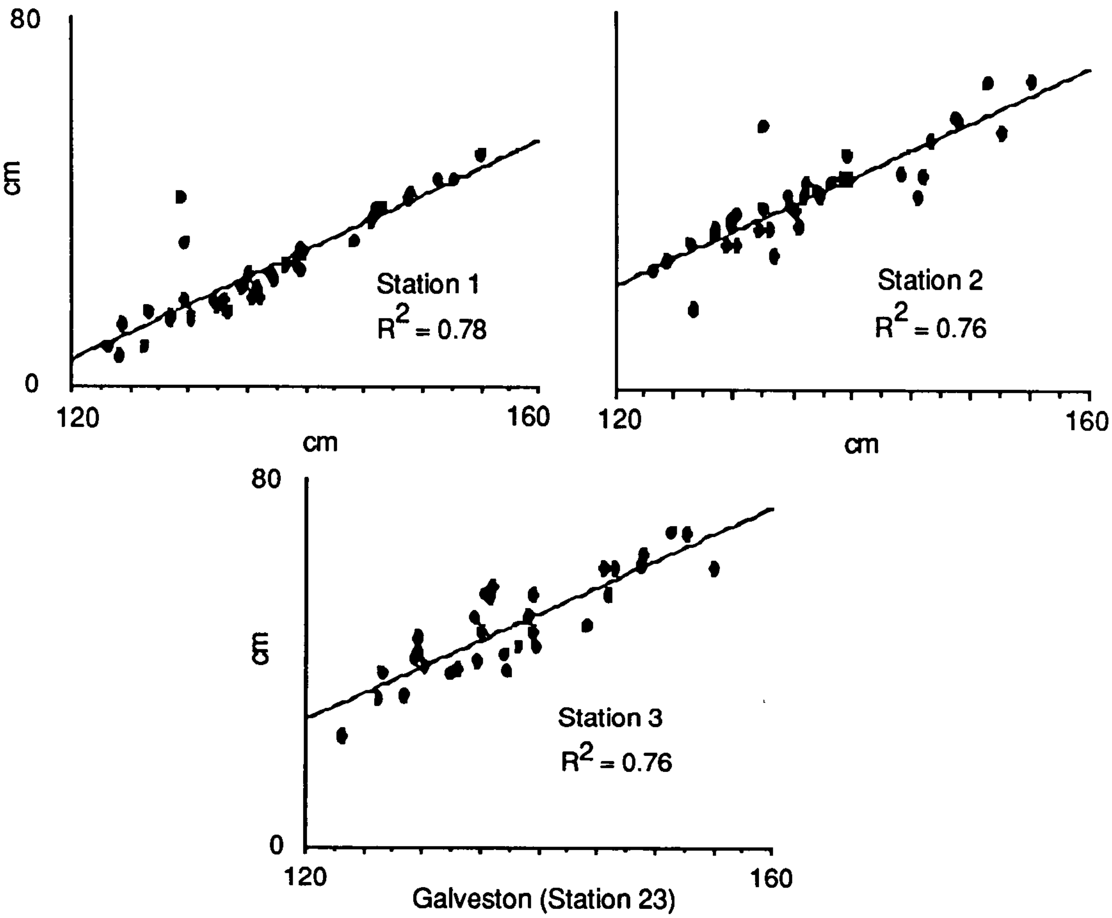

Figure 11-5. Annual mean water levels (cms) at Galveston, Texas (station 23) versus annual mean water levels at three stations in southwestern Louisiana, near Cameron.

Thus, geological subsidence and sea level rise are relatively constant at these stations for this century, and factors influencing the Galveston tide gage also affect water level throughout the northern Gulf of Mexico. A secondary implication is that geological subsidence caused by natural factors is much more significant than local changes in subsidence caused by belowground fluid withdrawal. The exception may be at the Golden Meadow oilfield (station 54), where water level rise is about double that at two nearby gages.

Sedimentation rates in coastal wetlands are not constant, but vary spatially and temporally (e.g., Baumann, 1980). Water levels also vary daily, seasonally, and over periods longer than lunar cycles (Figures 11-1 and 11-2). Water level variation among months is often ten times higher than variation among years and this can be attributed to differences in climatic influences (e.g., cold front passages in winter and storm events in summer). It takes little imagination to invoke climatic variability to explain variation among years (e.g., Chapter 2; Bradley et al., 1987; Swenson and Turner, 1987), because of the observed coincidental variations between wind, water level, temperature, and river discharge over days, months, and years.

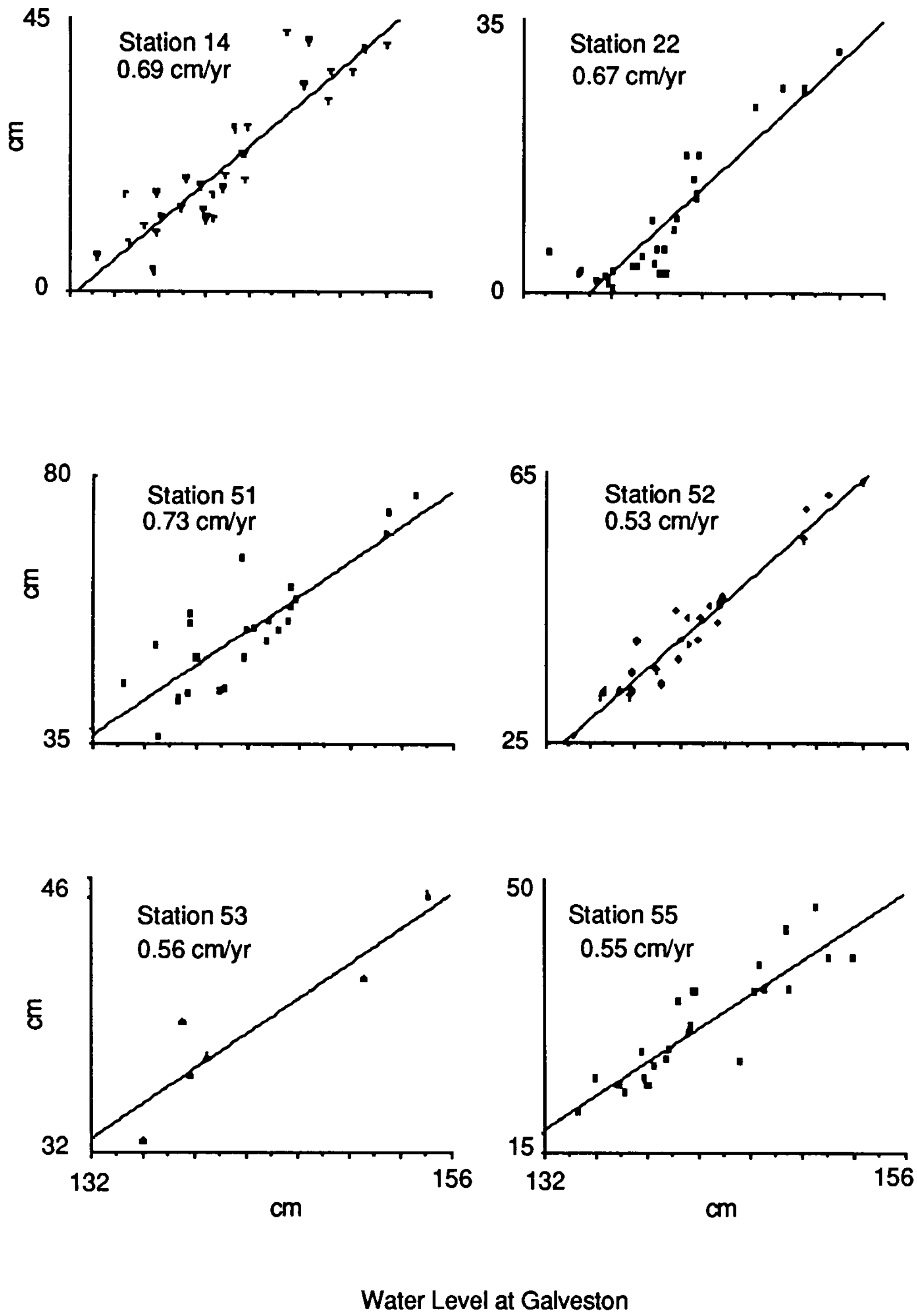

Figure 11-6. The relationship between annual mean water level at Galveston, Texas (station 23) and 6 stations in or near Barataria Bay, Louisiana. The number in the figure is the estimated long-term water level rise at each station based on this relationship and the long-term record at Galveston.

Long-term Subsidence Rates

The rate of subsidence in Louisiana coastal marshes is generally highest in the first few hundred years after deposition, and declines with age (Figure 11-7). The high rate in the first few hundred years is due partially to sediment compaction, soil dewatering, and decomposition. Biological processes contributing to subsidence probably predominate only during this first period as geological factors predominate at deeper depths. The relative water level rise in the later period (after 300 years) is around 0.2 cm/yr, and is close to the average rise in sea level this century as estimated from the analysis of the Florida tide gage records (0.23 cm/yr). This indicates that subsidence after 300 years is relatively small compared to the first 100 years or even negligible.

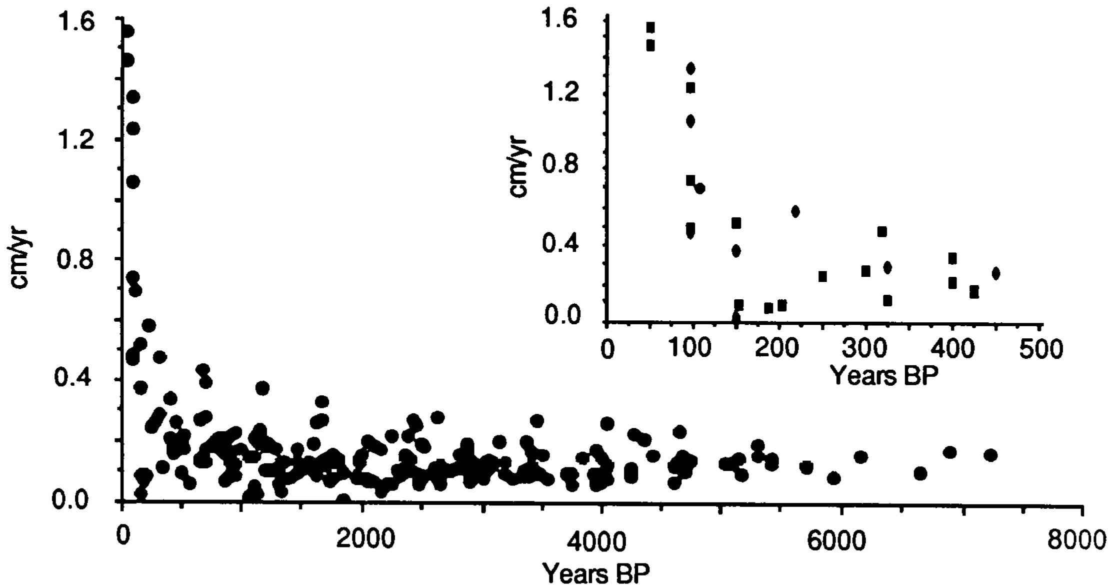

Figure 11-7. The burial rates for carbon dated sediments (from Frazier, 1967; and Penland et al., 1987).

Water level changes at these tide gages, as estimated in the previous sections, are partially correlated with sediment age and depth. Tide gages in the region of active deltas (the Mississippi and Atchafalaya River deltas) have the most rapid water level rise. In effect, water level rise is slowest where deltaic sediment accumulations are thin and old.

Vertical Accretion and "Sedimentation Deficits"

Negative differences between the vertical accumulation of sediments and water levels have been termed sedimentation deficits (e.g., Baumann, et al., 1984). The calculation of sedimentation deficits appears as a common-sense method to explore how well marshes maintain their vertical equilibrium as water rises and sediments compact. If water level rises faster than the marsh builds vertically, then increased flooding results and plant stress is likely to increase. Because water level varies over months, years, and decades, it is important to choose the correct water level record length to compare to the estimate of vertical growth of the marsh surface. The ^{137}Cesium dating method necessarily uses the 1963 fallout peak as a marker for the best estimator of vertical accretion. Tide gage records of water level from 1963 forward are then used to estimate a disparity between annual average water level rise and the measured vertical accretion. This disparity is generally on the order of 10-20 % more water level rise than vertical accretion in the inland marshes from 1963 to the late 1970s (Delaune et al., 1978, 1983, 1986, 1987; Hatton et al., 1983). As it turns out, the 1963 to 1982 water level change was about 200 % higher than the average for this century (e.g., Figures 11-1, 11-2), because the first few years after 1961 had relatively lower water levels (they declined in 1962 and 1963) and the following years above-average water level rise. Estimates of water level changes from 1963 to the late 1970s will therefore be much higher than average, and using that water level record interval in the calculation of vertical accretion disparities may suggest a sedimentation deficit when there is none in the long term. Further, if the marsh vertical accretion is constant this century, then there must have been periods of sedimentation "excess"; this probably occurred, for example, in 1962 and 1963. It is unlikely that there has been a significant century-long sedimentation deficit in the marshes sampled because all of the ^{137}Cs-dated samples came from an intact marsh. It may also be appropriate to use the century-long trend in water level because estimates of vertical accretion for 100 year old sediments (dated using ^{210}Pb isotopic residues) indicate relatively similar rates of vertical accretion from 1963 to 1986 compared to 100 years BP to 1986 (Table 11-2; Chapter 14; Delaune et al., 1987; Delaune et al., this report), especially if sediment compaction is taken into account.

Table 11-2. Estimates of recent sedimentation in back marshes using ^{137}Cs and from tide gage records. Levee (also called streamside) marshes are excluded and always have a higher estimated sedimentation rate.

Station Location	Sedimentation Rates[a]	Water Level Change[b]	Source of ^{137}Cs data
Chenier Plain, Southwestern LA	0.67	0.72 - 0.80 (1, 2, 3)	Delaune et al., 1983
Barataria Bay	0.59 - 0.75	0.53 - 0.74 (14, 22, 51-53, 55)	Hatton et al., 1983
	0.75	0.52 - 0.74 (14, 22, 51-53, 55)	Delaune et al., 1978
Barrier island south of Barataria Bay	0.55 - 0.78	0.67 (14, 22)	Delaune et al., 1986
Terrebonne marshes, central LA coast (exclusive of Wax Lake outlet marsh)	0.65 - 0.91	–	Delaune et al., 1987[c]

[a] cm/yr estimated with 137Cesium (1963 to 1978-83)

[b] cm/yr based on Galveston, TX record 1908 to 1980; station numbers in parentheses

[c] There are no other tide gage records of sufficient quality nearby to calculate marsh vertical accretion rates. Delaune et al. (1987) estimated "sedimentation rates to be equal to, or greater than, the subsidence rates close to the active delta."

The above is not meant to imply that other marshes which have fragmented or turned to open water have not undergone stress caused by too low sedimentation rates or that a decade-long disparity does not influence marsh stability. Sediments, of course, build and sustain marshes against a rising water level. But, strictly speaking, the data from the presently available sedimentation dating approaches cited herein could also be used to support the argument that a disparity between marsh growth vertically and water level rise is a temporary disparity, viewed over a time period of this century or even a few decades. The ability of the marsh to survive disparities between water level rise and vertical accretion is largely unexplored. If, as expected, sea level rise slows somewhat after 1982, then we can resurvey the marshes in 20 years to examine whether vertical accretion rates changed proportionally and coincidentally in time and space.

Chapter 12

HISTORICAL SEDIMENT DISCHARGE TRENDS FOR THE LOWER MISSISSIPPI RIVER

by

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Department of Geography and Anthropology

The Mississippi River has provided the sediment discharge responsible for the construction of the active and inactive subdeltas comprising the Deltaic Plain. The shifting of deltaic lobes and varying characteristics of the suspended and bedload portions of the sediment discharge are responsible for differences in the geomorphic forms and sedimentary environments associated with the plain. Bed load, which is composed largely of fine sand, provides much of the sediment that makes up channel, point-bar, levee, distributary mouth bar, crevasse splay, and coastal beach deposits. The fine silts and clays that comprise the suspended load are deposited seaward from the river mouth with some sediment being pushed inshore by marine currents and deposited in interdistributary troughs (Fisk et al., 1954). During overbank flow that occurs above flood stage, suspended sediments are carried into adjacent marshlands and interdistributary basins. The amount of sediment introduced into coastal wetlands no longer appears sufficient to offset the land loss. The purpose of this study is to examine and analyze the historic evidence that can be used to discover what changes in the suspended and bedload regimes have taken place in the lower Mississippi River during the last 100 years.

Since 1800 the sediment discharge of the River has been affected increasingly by man-made constructions, as well as natural events. A qualitative estimate of the magnitude and duration of these factors is illustrated in Figure 12-1. Most of the factors with the greatest influence are man-induced and have occurred since 1900. Both the bed load and the suspended load regimes of the channel have been affected. Prior to 1900, the River generally had been considered unaffected by man-made structures (Stevens et al., 1975). Recent studies have indicated that during the past 35 years there has been a 50% decrease in the suspended load of the Lower Mississippi River (e.g., Meade and Parker, 1985; Tuttle and Combe, 1981). This decline has been attributed to revetment and dam construction (Tuttle and Combe, 1981; Robbins, 1977). Studies during the periods 1877 to 1883 (Ockerson, 1892) and 1931 to 1941 (Smith and McCleave, 1945) indicate that bank caving supplied approximately 476,000 m^3 of sediment per km (1,000,000 yd^3/mile) of channel in the Lower Mississippi River above the Old River diversion. Revetment construction has largely eliminated this sediment source (Winkley, 1977).

In addition to a reduction in sediment discharge, the construction of artificial levees since 1900 has played an increasing role in confining sediment loads that, under natural conditions, would have provided overbank sediments to the adjacent flood plain and marshlands. Elliot (1932) documented that during the period 1897 to 1921, sedimentation rates on channel banks from Donaldsonville to Carrollton ranged from 0.15 to 0.52 m/yr (0.5 to 1.7 ft/yr) and that, during the period 1894 to 1917, the rate for 100 miles below Carrollton to the Forts was 0.27 m/yr (0.9 ft/yr).

Current estimates of sediment reduction on the Lower Mississippi River are based on systematic measurements of the sediment discharge taken since 1950 at Tarbert Landing, Louisiana. Because these estimates are based on such a short period of record, they may not recognize longer-term cyclic changes that may be present. These estimates are also limited to changes in the suspended load and do not provide any information on the bedload fraction. Because the bed load provides an important source of sediment for channel deposits, interdistributary troughs, levees, crevasse splays, and beaches (Fisk et.al., 1954), changes in the availability of this fraction should also be documented.

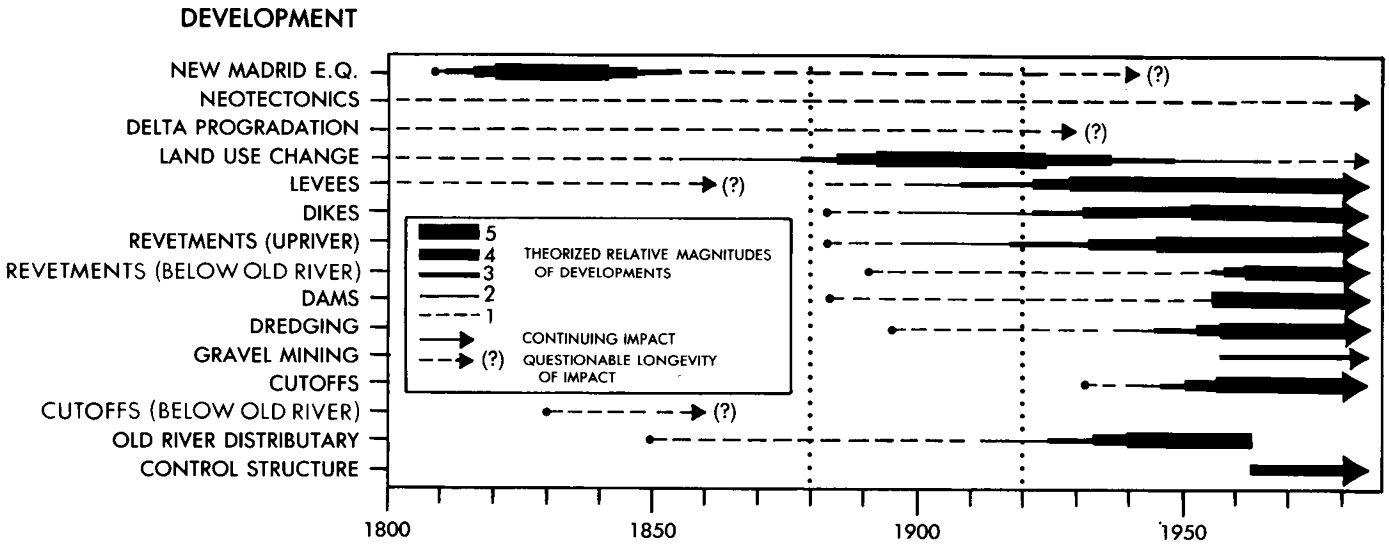

Figure 12-1. Qualitative estimates of the magnitude of events that may have caused changes in sediment discharge of the Lower Mississippi River.

Flood protection levees and channel improvements have virtually eliminated the transfer of sediment from the River to the coastal wetlands. To evaluate the influence of this loss, some estimate of the amount of sediment involved and the manner it was delivered to the coastal plain should be documented.

Literature Review

Background and History of Suspended Sediment Measurements

The first recorded sediment sample was taken in the Mississippi River Basin by Captain Talcott in 1838 near the mouth of the River. Since that date, a number of agencies have taken sediment samples on both regular and irregular bases. The history of sediment sampling up to 1931 is summarized in two papers published by the U.S. Waterways Experiment Station (WES, 1930, 1931). These documents include an inventory of 73 sites from which sediment samples have been collected. Measurements were confined to the suspended load and were gathered mostly on a sporadic basis for experimental and reporting purposes. Two surveys during the period provide measurements on a yearly basis. Humphreys and Abbot (1861) measured suspended load from 1851 to 1853 at New Orleans, and a survey by Major Quinn (1894) recorded suspended sediment load in South Pass from 1879 to 1893. The Quinn survey provides water discharge measurements for all major distributaries at the mouth of the River that can be used to estimate the total suspended load for the entire River. From 1894 to 1950, no measurements of suspended load were conducted for an entire year at gaging stations from Cairo to the Gulf. The longest records available during this period include six consecutive months, September 1930 to February 1931 (WES, 1931) and six consecutive months from February to July, 1938 (WES, 1939). The former include measurements for Vicksburg, Red River Landing, and Carrollton, while the latter is confined to measurements taken from the Passes at the mouth of the River. Starting in 1950, the suspended load of the Lower Mississippi River below the Old River diversion (into the Atchafalaya Basin) was measured at Baton Rouge until 1958; at Red River Landing from 1959 to 1963; and at Tarbert Landing from 1963 to present. These represent the longest continuous records of suspended load measurements available for a major gaging station on the Lower Mississippi River. Taking into account these data, Robbins (1977) and Koewn et al. (1981) provide the most recent summaries and analyses of the sediment regime of the Mississippi River. The only other suspended sediment record available on an annual basis for a Mississippi River gaging station below the Old River diversion is near New Orleans at Belle Chasse where the U.S. Geological Survey (USGS) has taken measurements once a month since 1977. This study has expanded the suspended sediment data base for the lower River with the acquisition of unofficial measurements taken by the New Orleans Water and Sewage Board. These measurements have been taken weekly since 1930 and represent the longest continuous record available for the Mississippi River below Cairo.

Bedload Measurement

There are no direct estimates of bedload amounts transported by the Mississippi River because of difficulties with measurement. Fisk et al. (1954) estimated bed load at 10% of the total sediment load, while Holle (1952) estimated 20%. Reviews of bedload studies are included in the reports by Robbins (1977) and Keown et al. (1981) and have examined the size, shape, and mineralogy of the sediments. The first comprehensive study of bed material of the Mississippi River was conducted by the Mississippi River Commission in 1932 and 1934, when 615 samples were collected between Cairo and the Gulf of Mexico (WES, 1935). Since 1965, studies of bedload characteristics have been conducted by the individual engineering districts (Memphis, Vicksburg, and New Orleans) of the U.S. Army Corps of Engineers (COE) which are responsible for a particular segment of the River. The 1932/34 and post 1965 data have been the basis for comparative studies to determine any long-term trends in the relative size characteristics of the bed sediments (Keown et al., 1986; Keown et al., 1981; Robbins, 1977).

Some indication of changes in the volume of bed load transported by the Mississippi River may be calculated indirectly using historical hydrographic survey maps. Within a meandering river system, point bars provide one possible storage area for bed sediments. A series of hydrographic surveys published by the Mississippi River Commission provide topographic data for active point bars in the Mississippi River. These maps are available for the periods 1879 to 1881, 1911 to 1915, 1921 to 1925, 1935 to 1938, 1948 to 1952, 1961 to 1963, and 1973 to 1975, at scales of 1:10,000 and 1:20,000.

Methods

The procedures used to identify historic trends in the sediment discharge of the Mississippi River are restricted by the limited data available. Analysis of the sediment discharge was confined to the lower Mississippi River below the Old River diversion for several reasons. First, this segment has the longest and most extensive data base. Second, it is the sediment passing through this lower segment that is available to the coastal wetlands. The analysis included data published by or from the files of the Mississippi River Commission (Vicksburg), COE (Vicksburg, New Orleans Districts), USGS (Baton Rouge), and the New Orleans Water and Sewage Board.

The analysis of suspended load data included measurements taken at New Orleans in 1851, 1852, 1930 through 1983; at South Pass from 1879 through 1893; and at Tarbert Landing from 1950 through 1983. Data from these sources provide water and suspended sediment measurements for either 12 consecutive months or a total for the entire year. Although measurement techniques employed by these sources are not always the same, some standardization of the data is possible. Where necessary, the annual suspended load was determined by procedures employed by the COE (Robbins, 1977) at Tarbert Landing. Sediment concentration (mg/l) is used to calculate the suspended load using the formula (Eqn.12.1) outlined by Porterfield (1970), which is

$$Qs = (Qw)\ (Cs)\ (K) \qquad \text{Eqn.12.1}$$

where

Qs is the suspended sediment discharge in tons(metric)/day
Qw is the water discharge in cubic meters per second (m^3/s)
Cs is the concentration of suspended sediment in mg/l, and,
K is a constant that includes the conversion of water discharge and equals 0.00245.

Where water discharge was not given, values from Red River Landing or Tarbert Landing can be used in the computations because there are no major tributaries entering the River below these stations.

The Tarbert Landing data are reported as tons/day and annual total suspended load in tons (for this report, English units are converted to metric). Supplemental data include total sand and silt loads in tons/day and sediment concentration in ppm by weight. Data from the New Orleans Water Board (1930 to 1983) consist of sediment concentration in ppm by weight for water samples taken from an intake pipe located 9 m (30 ft) from the bank line and 4.6 m (15 ft) below the average low water plane. Sediment concentration data are available on a weekly basis and were averaged to obtain monthly values. The sediment load for each month was then calculated and the total annual suspended load computed by adding these values. Water discharge data for Red River and Tarbert Landing were used in the computation. The Water Board data overlap with the Tarbert Landing data but extend the suspended sediment discharge record back from 1950, the limit at Tarbert Landing to 1930. The sediment concentration measured at New Orleans is based on a single-point sample, while at Tarbert Landing it is based on the average of a series of depth-integrated samples taken at different verticals across the River. The annual suspended sediment load determined from the New Orleans data is consistently lower than that measured for similar years at Tarbert Landing (Figures 12-2 and 12-3). This trend may represent a more accurate estimate of the sediment concentration at Tarbert or possibly the difference reflects deposition in the River reach between the two stations. Both data sets, however, show almost identical trends for the suspended sediment load of the Lower Mississippi River for the period since 1950. The New Orleans data, therefore, appear to provide a reasonable estimate for the trends in suspended sediment load between 1850 and 1930.

The Quinn survey from 1879 to 1893 provides an annual ratio of sediment to water by weight passing through South Pass. The proportion of the total annual water discharge of the River that passes through South Pass is given, as well as the proportion of sand in the suspended load. The samples from this survey were taken in the vicinity of Head of Passes, where turbulence is generally high and therefore sediment concentration and amount of sand in suspension can be higher than elsewhere in the River. However, the sediment concentrations and the calculated sediment discharges from this survey compare well with similar data compiled from the Humphreys and Abbot survey (1851-52) made in the vicinity of New Orleans (Figure 12-2). Although the values computed from the Quinn data may be somewhat higher than upriver, they appear to be representative of the suspended sediment discharge for the period. Based on these data, sediment concentration in mg/l and annual water discharge were calculated. These values were used in Eqn. 12.1; the Qs (tons/day) were determined and then multiplied by the number of days in that year. As a comparative check, the annual water discharge was also determined using stage data from Red River Landing for the same years published by the COE (Latimer and Schweizer, 1951). This value was substituted for the Quinn estimate in the formula and the Qs recalculated. The values used in the analysis were an average of the two Qs values (Table 12-1). Measurements for 1851 and 1852 included monthly sediment concentration in ppm by weight and water discharge in cfs. The sediment concentration in 1851 was based on samples taken at three levels in the River (surface, mid-depth, and bottom) and a weighted mean for the three values calculated. The sediment concentration value for 1852 was based only on measurement from surface samples. Based on the relationship between surface samples and subsurface samples in 1851, the 1852 surface concentration measurements were multiplied by a constant equal to 1.14. The suspended sediment load was computed for each month and summed to obtain the annual suspended sediment.

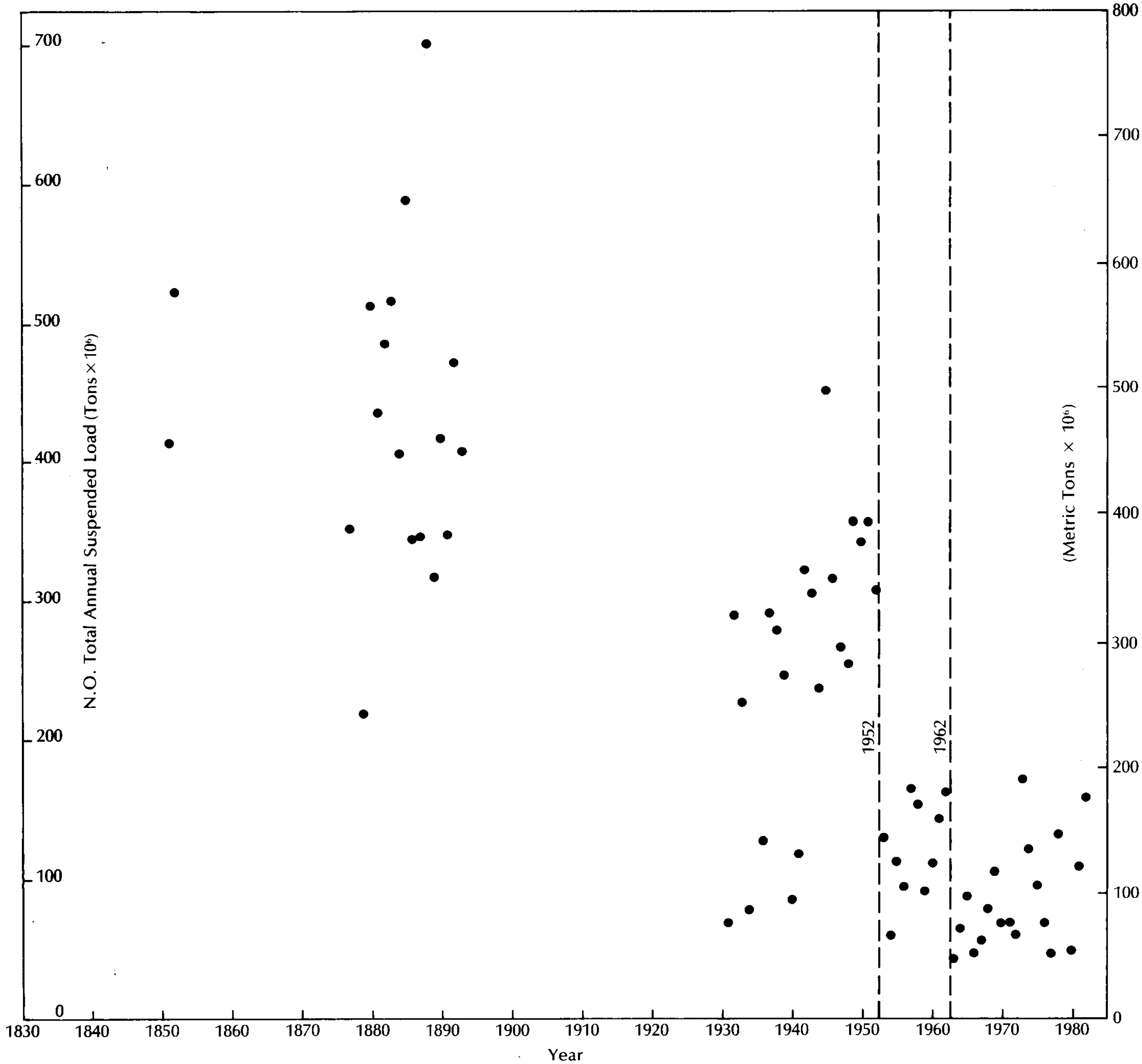

Figure 12-2. Total annual suspended load for the Mississippi River below New Orleans based on data from Humphreys and Abbot, 1851 to 1852; Quinn, 1879 to 1893; and New Orleans Water Board, 1930 to 1982.

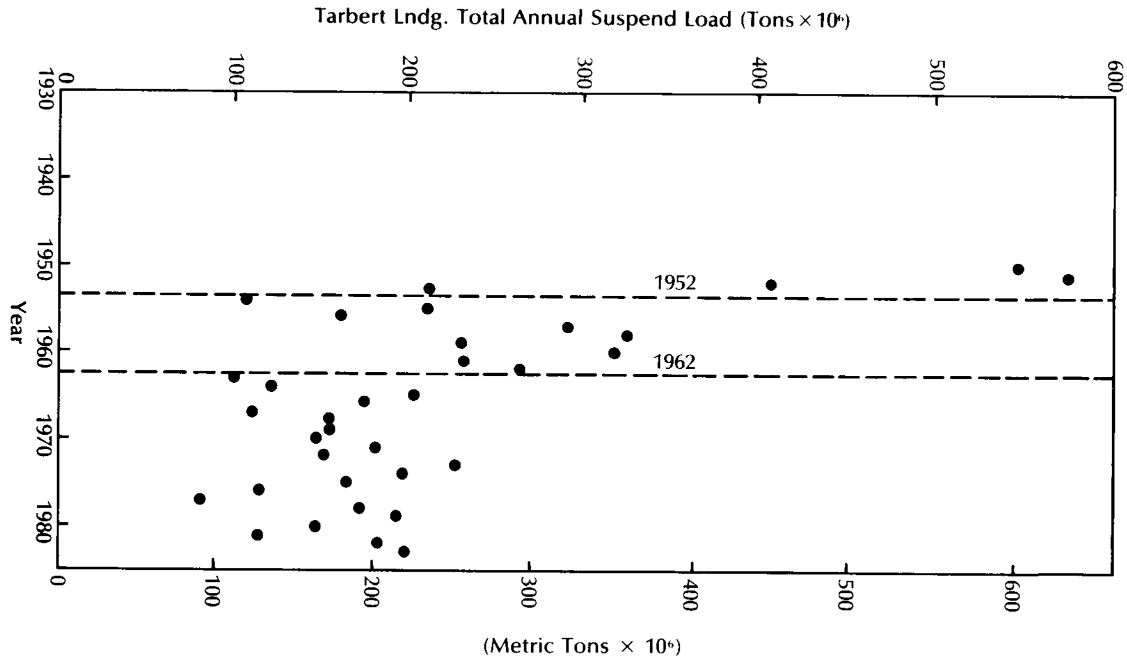

Figure 12-3. Total annual suspended load below Old River, based on data from Tarbert Landing.

Table 12-1. Suspended load estimates at Head of Passes, 1879-93.

Year	$m^3 \times 10^6$	mg/l	Quinn Est.	Formula #1 (tons x 1000)	Average
1879	4.22	456	230607	166425	198515
1880	5.98	803	518292	414654	466473
1881	5.58	826	488057	301463	394760
1882	8.50	642	411275	471518	441396
1883	6.49	817	480274	458575	469425
1884	7.42	589	359757	377993	368875
1885	6.00	877	615502	455410	535456
1886	5.66	543	360927	266009	313318
1887	4.53	641	376408	251842	314125
1888	5.58	1099	742600	531118	636860
1889	6.23	521	294712	280876	287794
1890	7.65	574	378001	379755	378878
1891	6.15	581	324478	309103	316791
1892	7.17	733	403577	454246	428911
1893	6.12	621	412290	329167	370728

The analysis of sediment volumes stored on point bars was conducted using five Mississippi River Commission (MRC) hydrographic surveys: 1879-80, 1911-15, 1935-37, 1948-52, and 1973-75. It was first necessary to standardize the datum planes, scales, and low water surface elevations used on the maps. All hydrographic and topographic data for the first two surveys, 1880 and 1915, were based on the Memphis Datum, a reference plane 58.17 m (190.84 ft) below gage zero on the Memphis, Tennessee, gage, which corresponded to -2.47 m (-8.1 ft) below sea level in 1880 and to -2.02 m (-6.63 ft) below sea level in 1915. It was necessary to subtract these values from all elevations taken from these surveys so that they would be comparable to the more recent surveys that are based on mean sea level. The 1880 and 1915 surveys also differ from the later surveys in terms of river bed elevations, which are expressed in soundings below the water surface elevation measured on the date of the survey and based on the nearest gage reading, instead of elevations above or below mean sea level. The Average Low Water Plane (ALWP), which was established by the COE for the 1963-65 survey, was used as the standard low water reference plane for the base of all point bars in each of the surveys.

Point bar volumes are average-end area volumes above the ALWP, computed with a Numonics 2400 digitizer. The volumes are computed by the digitizer by measuring the area contained within successive contour lines and multiplying the mean area by the contour interval. The results presented in Table 12-6 include the absolute or total change between each period and the volume change in kilometers of channel length. Meanders and cutoffs continually cause the length of the River to change; therefore, the volume change/km of channel provides a more meaningful basis for comparison.

Trend in Suspended Load Record

Based on the longest period of record, which includes data prior to 1900 and the 50-plus year record from the New Orleans Water Board (Figure 12-2), there has been a substantial decrease in the amount of suspended sediment transported by the Lower Mississippi River below Tarbert Landing. This decrease is reflected in both the total annual tons of sediment transported (Figures 12-2 and 12-3) and in the sediment concentration (Figure 12-4). The overall decrease from 1851 to the present may be, in part, caused by a decline in water discharge as suggested by gage readings at Vicksburg (Schweizer, 1954). Changes in landuse practices may also play a major role (Keown et al., 1986), although the lack of data between 1900 and 1930 makes it difficult to establish the magnitude of their effect. Decreases in the suspended sediment regime since 1950 appear to be the result of the construction of locks and dams on major tributaries. A decrease in the suspended discharge in 1952-53 probably can be attributed to a 55% decrease in sediments carried by the Missouri River (Figures 12-4 and 12-5). A further decrease in 1962-1963 most likely reflects an 87% decrease in sediments supplied by the Arkansas River. Similar changes can be seen in the suspended sediment records from Tarbert Landing (Figure 12-6) and the Atchafalaya River. Data from all sources indicate that there has been a 50% reduction in suspended sediment load by 1953 followed by a further 30% decrease since 1963 (Figure 12-7). The overall decrease in suspended sediment since 1850 appears to be approximately 60% (Figure 12-7).

A comparison of the data from the Quinn Survey, Tarbert Landing, and Belle Chasse (Carrollton) suggests that there has been a significant decrease in the size of sediments carried in suspension (Figure 12-8). The proportion of sand (2.0 mm to 0.05 mm) carried in suspension decreased from an average 46% in the period from 1879 to 1893 to 23% measured at Tarbert Landing from 1950 to 1983, a decrease of 50%. The data from Belle Chasse cover a much more limited time period but is in closer proximity to the South Pass where the Quinn survey was conducted. The proportion of sand at Belle Chasse averages 13%, which indicates a 72% decrease in sand since 1890. The decrease in the proportion of sand from Tarbert Landing is probably a function of downstream sorting or possibly differences in sample size.

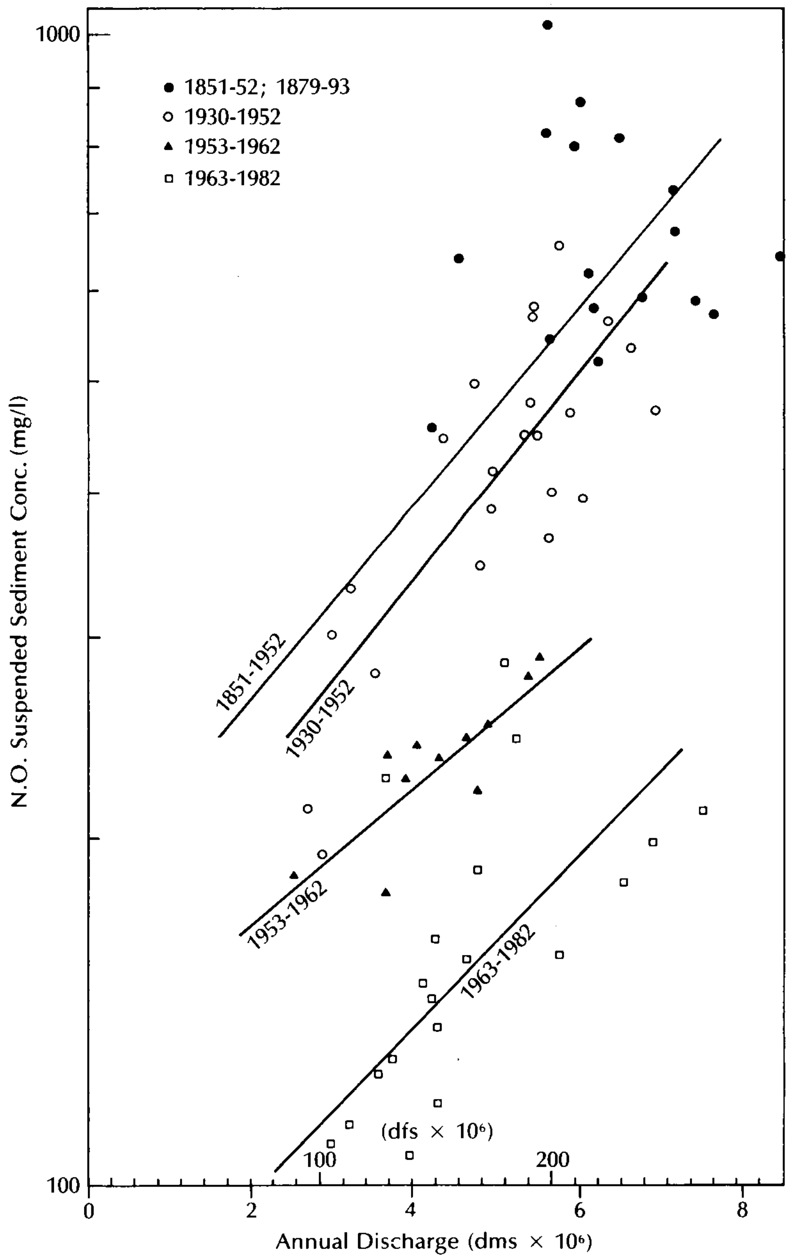

Figure 12-4. Relation between suspended sediment concentration (ppm) and annual discharge below New Orleans. Three distinct groupings can be seen in the data: 1851-1952; 1953-62; 1963-82.

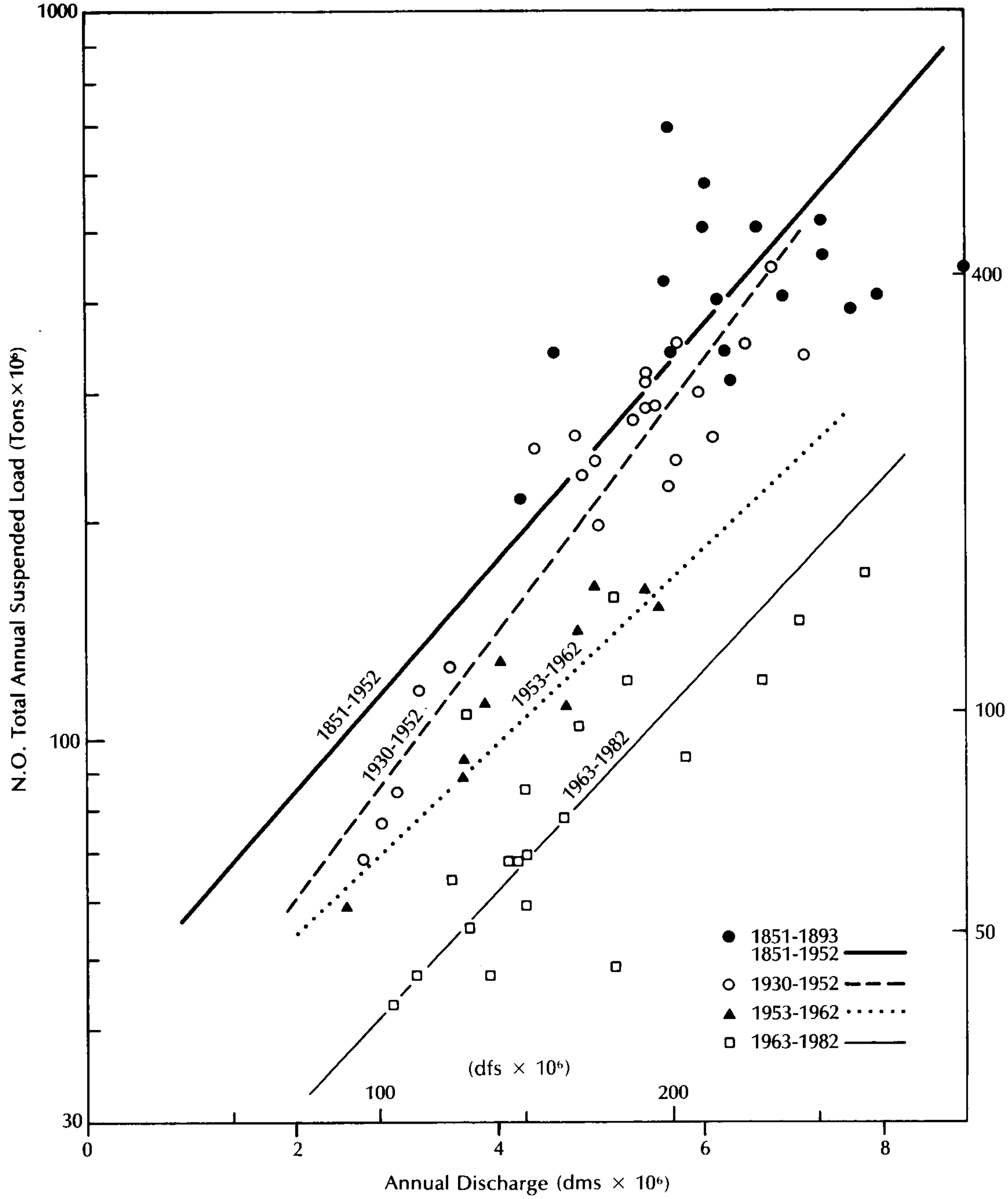

Figure 12-5. Relation between total annual suspended load and annual discharge below New Orleans. Similar grouping of the data to those in Figure 12-4 are apparent.

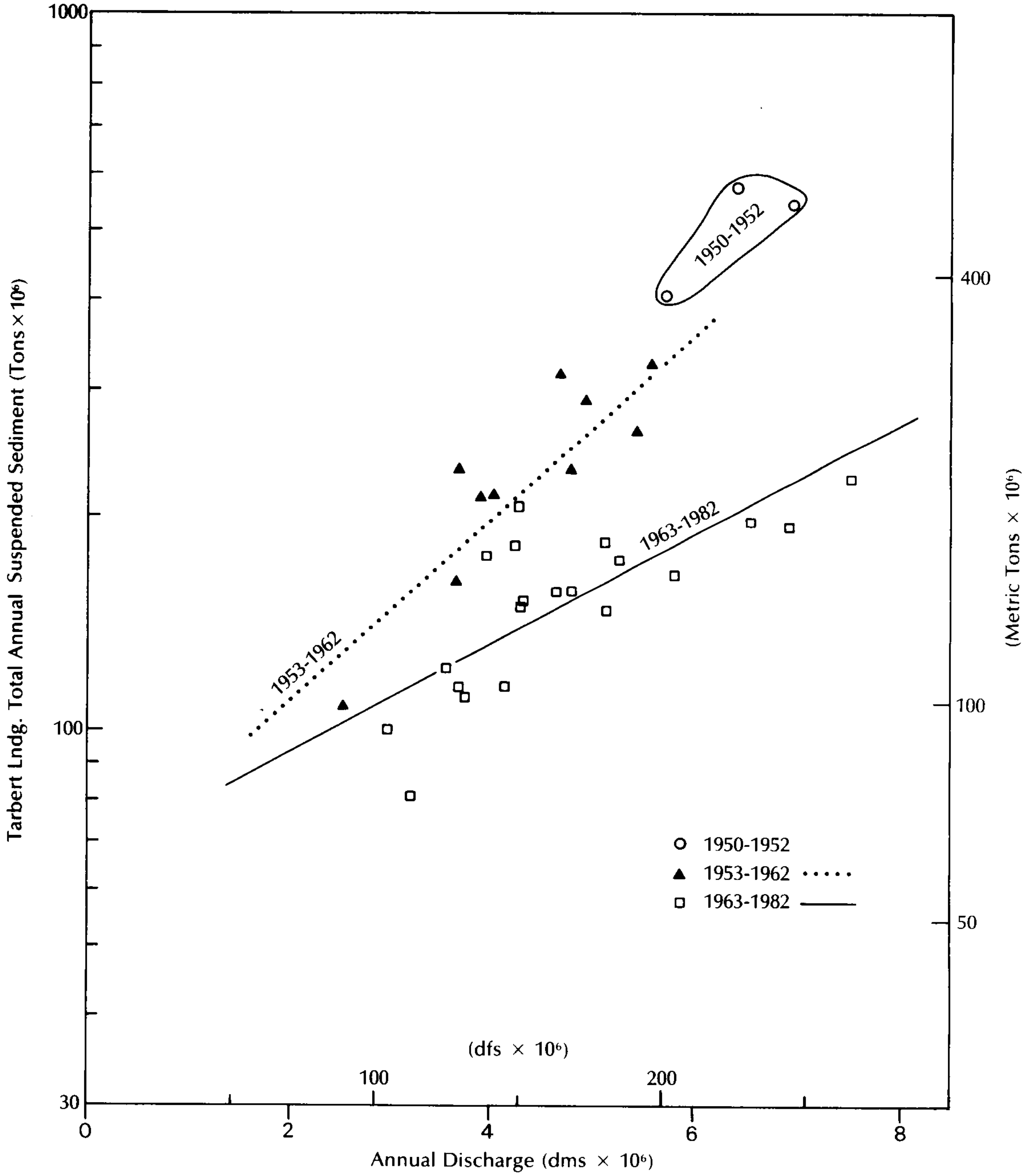

Figure 12-6. Relation between total annual suspended load and annual discharge at Tarbert Landing. Groupings of data are similar to the New Orleans data.

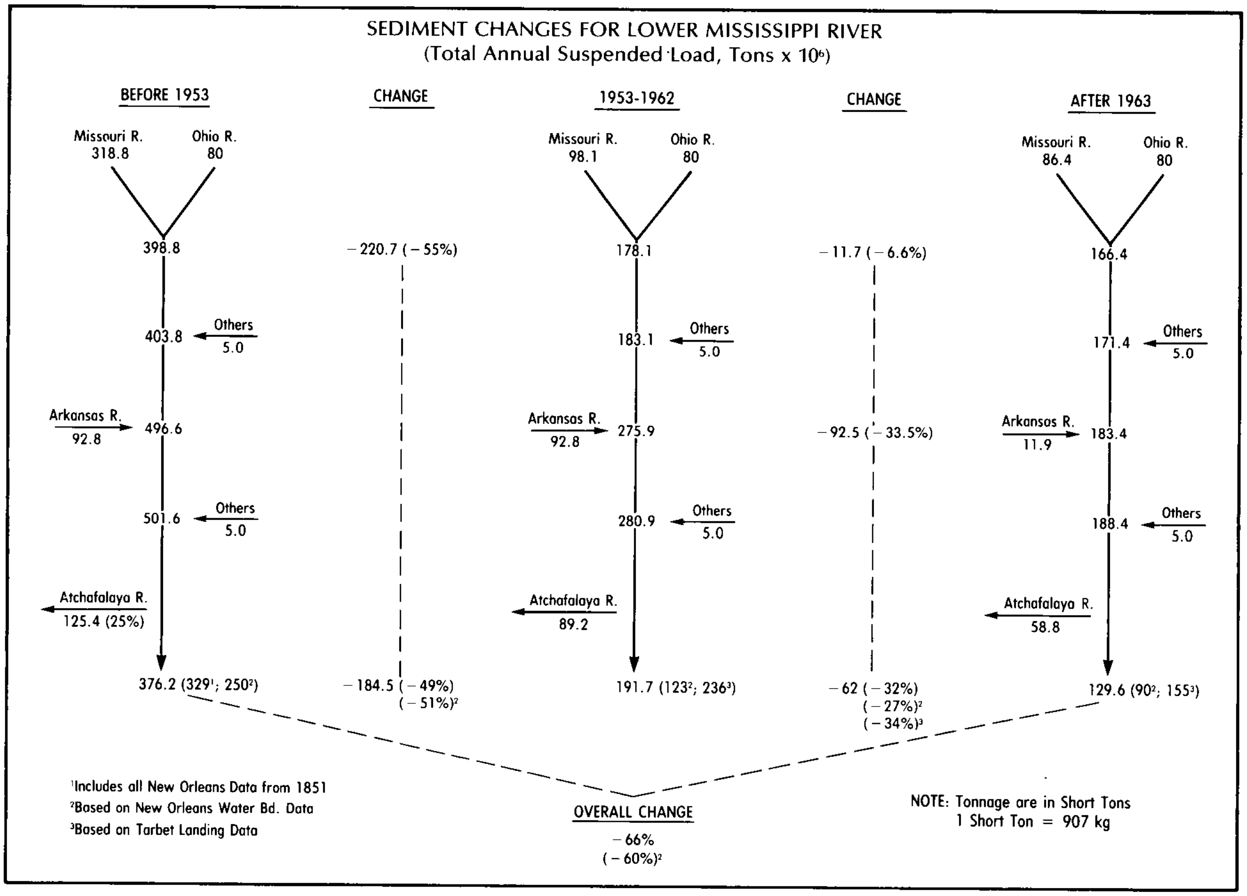

Figure 12-7. Mississippi River suspended sediment flow regime before and after major dam closures. Values on main stem and tributaries from Keown, et al., 1981.

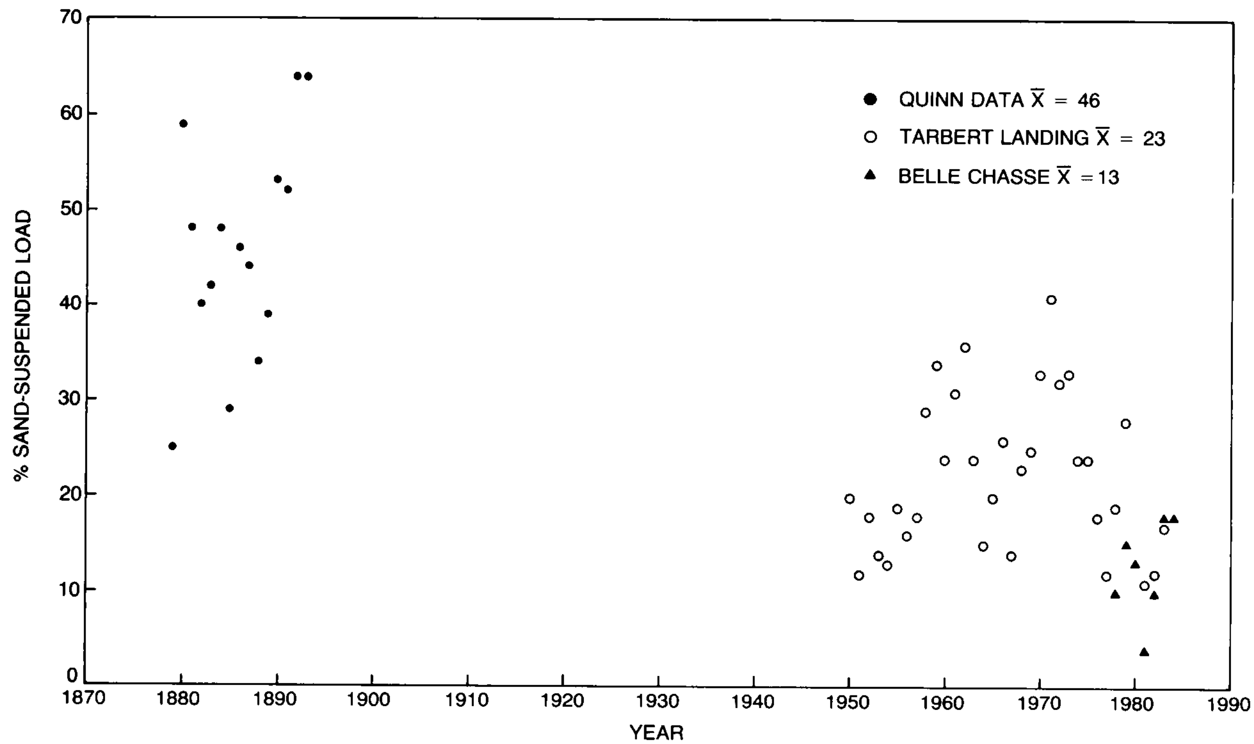

Figure 12-8. Changes in the percentage of sand carried as suspended load.

Changes in Suspended Sediment Delivered to Wetlands

Overbank Contributions

Since 1927, overbank flow to areas adjacent to the Mississippi River has been all but eliminated with construction of artificial levees. Estimates of the magnitude of this sediment loss to these areas can be made using measurement data, as well as using a modeling approach. The sediment and water discharge data from Tarbert Landing from 1949 to 1983 were analyzed (Figure 12-9) to determine the amount of sediment available for overbank flow if artificial levees were not present. Two assumptions were made in this analysis: (1) a bankfull discharge at a rate of 25.5 m3/s (900,000 cfs) was assigned, based on natural levee heights; (2) sediment concentration was considered uniform throughout the entire water column. The proportion of water discharge above bankfull was computed from daily records, and that proportion was used to determine the suspended sediment load carried in the above bankfull flow. The amount during this period that would have been available by overbank flow was estimated to be 163,428 x 10^3 tons (180,145 x 10^3 short tons). This value amounted to 14% of sediment carried during the flood flows, but only 2.6% of the sediment load carried during the entire 34-year period. Using a density conversion of 1.01 m^3 equals one ton (1.2 yd^3 = one short ton); this amount of sediment translates into an accumulation of 6.37 cm (2.5 in) over a 2,600 km^2 (1000 mi^2) area.

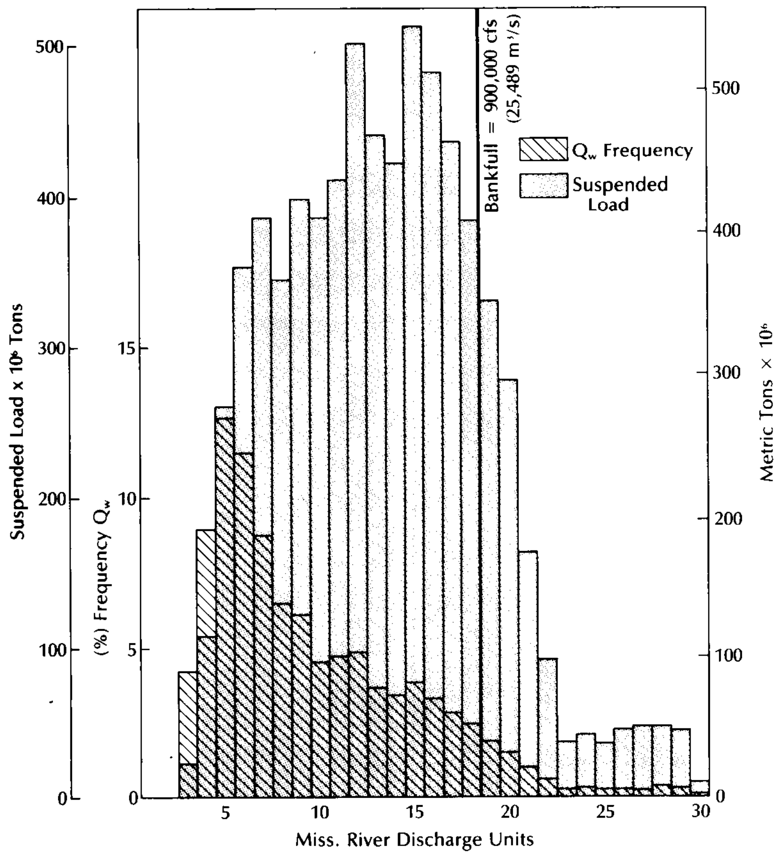

Figure 12-9. Frequency of sediment and water discharge at Tarbert Landing, from 1950 to 1983.

An estimate was made to determine the magnitude of change in the amount of suspended sediment available for overbank flow between the pre- and post-dam construction periods. For the purpose of this model, the sediment and hydrograph characteristics of the 1973 flood were used (Chin et.al., 1975). Similar assumptions were used in deriving this model as those in the previous calculations. Table 12-2 outlines the main overbank characteristics of the 1973 flood. Suspended sediment available for overbank flow prior to dam construction was estimated by substituting a revised value for sediment concentration (mg/l) into the 1973 calculations. An average monthly sediment concentration value for each flood month (March through June) was determined from measurements made between 1851 and 1930. Based on these values, the 1973 concentrations were multiplied by a correction factor (Table 12-3), and the daily suspended load was recalculated. These results (Table 12-2) suggest that for a major flood event, suspended sediment available for overbank flow has been reduced by 60% during the last century. This estimate compares well with that obtained in the previous section. These data also provide an estimate for the amount of sediment that would be deposited in areas adjacent to the River if the flow were not confined by artificial levees. In 1973 the sediment available would have covered an area of 750 km^2 (290 mi^2) with a 2.5 cm (1.0 in) accumulation, whereas during the pre-dam period the same flood would have deposited the same thickness over 1,950 km^2 (750 mi^2).

Table 12-2. Comparison of overbank sediment before and after dam building. Qw = water discharge; Qs = suspended sediment.

	1973			
	March	April	May	June
No. days > bankfull	8	30	31	22
% Qw > 25.5 m^3/s	13	30	35	15
Qs above bankfull (tons x 10^3)	1085	8471	7402	1767
Total sediment available for overbank= 18,720,979 tons				
	Pre-dam construction			
Qs above bankfull (tons x 10^3)	2262	20624	20779	5419
Total sediment available for overbank = 49,084,056 tons				

Table 12-3. Correction factor for suspended sediment calculation.

Month	Pre-dam (mg/l)	1973 (mg/l)	Correction
March	1033	421	2.45
April	746	301	2.48
May	572	202	2.83
June	797	213	3.74

Crevasse Splay Contributions

Crevasse splays occur during flood stage when overbank flow becomes concentrated and a well-defined channel develops with enough scour capacity to create permanent or semi-permanent breaks in the levee. Crevasses generally occur along the concave (outside) bank of meander bends. Along the lower Mississippi River, crevasse openings in levees have ranged in width from 100 m to over 1500 m and can scour to depths in excess of 15 m. Prior to the construction of artificial levees, crevasse splays were a common occurrence along the lower Mississippi River during bankfull periods (Elliot, 1932). Crevasses provide avenues for flood waters and sediments to enter into adjacent backswamp basins (Figure 12-10). Intermittent or short-duration crevasses result in a branching pattern of small levee ridges and abandoned channels extending for a short distance into the backswamp area. Crevasse splays that persist for longer periods may extend many km into backwater basins. The life span of these crevasses is not well documented, but, based on archaeological evidence, they may function and remain as topographic features for several hundred years (Gagliano and van Beek, 1970).

Subdeltas at the mouth of the River are also formed by crevasses (Figure 12-11). Comparative map studies (Gagliano et.al., 1971) indicate that subdeltas account for more than 80% of the new land built in the active delta during historic time. Subdeltas progress through a sequence of stages that are dependent on the interaction of such factors as sediment supply, coastal erosion, and subsidence. This sequence, which generally covers an interval of 50 to 150 years, is initiated by a crevasse and is followed by subaqueous growth, then a period of rapid subaerial development, and finally deterioration (Figure 12-12). If the subdelta crevasse is scoured deeply enough to maintain flow during succeeding low-water stages, it will be perpetuated and prograde rapidly; if not, it will deteriorate because of a loss of hydraulic efficiency through numerous channel bifurcations and aggradation of the receiving basin by associated sedimentation processes.

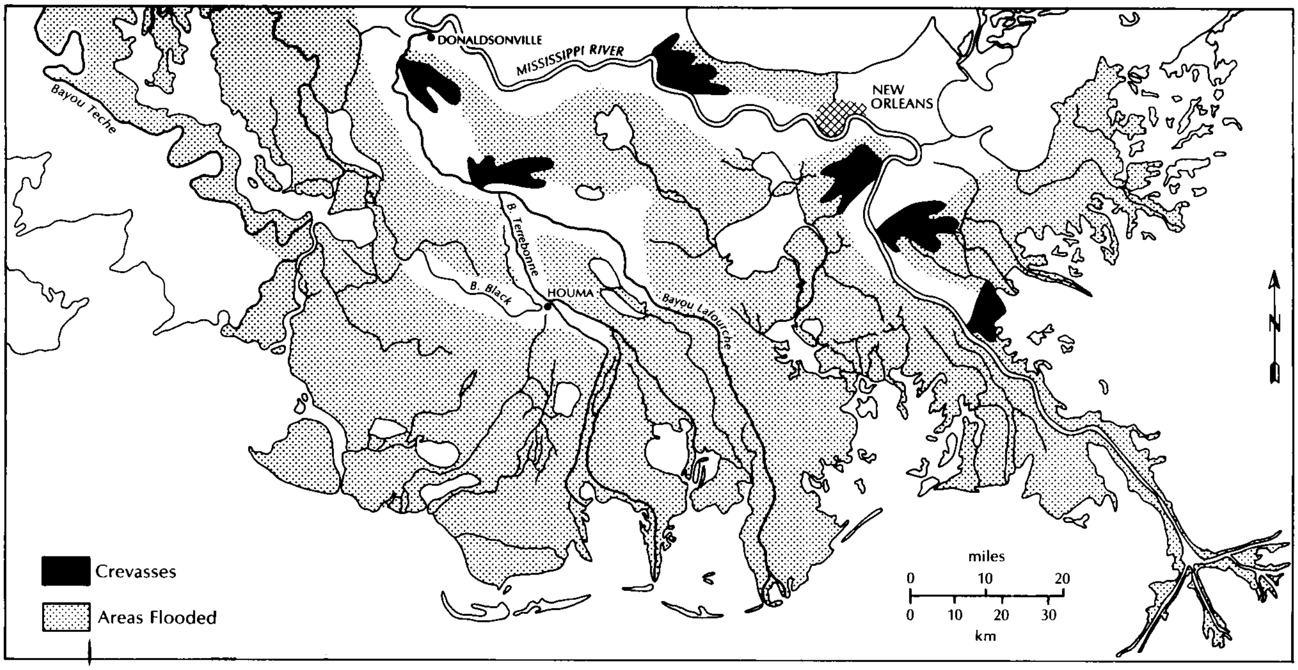

Figure 12-10. Major crevasses and areas of overbank flooding prior to 1874 (from map compiled by T. S. Hardee, map files COE, Vicksburg, Mississippi, 1874).

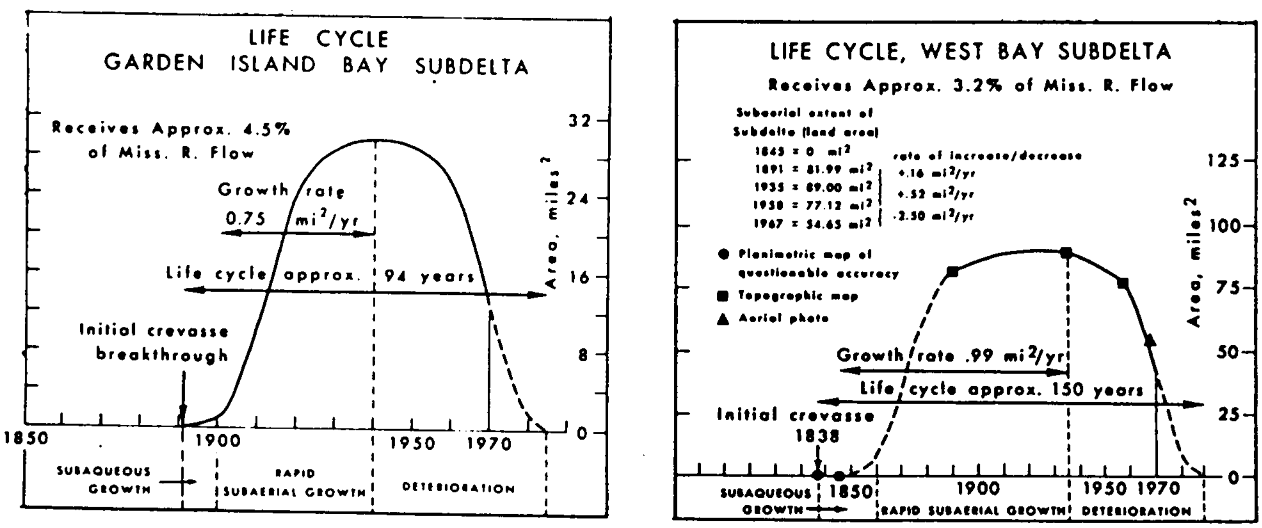

Figure 12-11. Subdeltas of the modern Mississippi River (after Coleman and Gagliano, 1964).

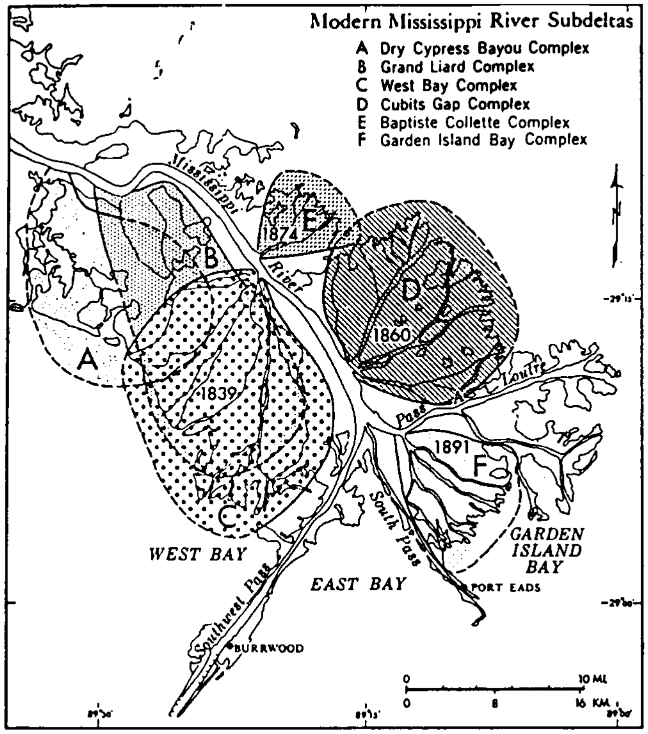

Figure 12-12. Life cycle of subdelta type crevasse (from Gagliano et al.,1971).

In contrast, a crevasse upriver from the deltaic location occurs and progrades when bankfull stage is reached but is inactive during low water periods. Scour during high-stage flows over several years or decades may cause crevasse channels to become more or less permanent distributaries. Data on the sedimentation characteristics of crevasses are rare, particularly for main channel types. Data compiled by Gagliano et al., (1971) on the major historic subdeltas of the Mississippi River are summarized in Table 12-4. The Bonnet Carre crevasse, located on the north bank of the River about 32 km upriver from New Orleans (Figure 12-10), is a main channel type that was initiated in 1849. From 1849 to 1874, the crevasse overflowed five times during bankfull flows.

A survey was conducted to estimate the amount and distribution of sediment contributed by these crevasse events to Lake Pontchartrain (Hardee, 1876). Data from this survey provide the basis for sediment volumes in Tables 12-4 and 12-5. An estimate was included in the sediment volume for the 91 km^2 subaerial portion of the crevasse assuming an average thickness of 0.9 m. The crevasse was replaced by the construction of the Bonnet Carre Spillway in 1931. The spillway, in effect, is an artificial and controllable crevasse used to protect New Orleans by diverting portions of major Mississippi River floods into Lake Pontchartrain. The spillway has been operated seven times since 1937, providing valuable information on the effects of crevasses. Discharge and sedimentation characteristics for these events are given in Table 12-5.

The data in these two tables indicate the importance of crevasses to the sediment regime of the adjacent wetlands, as well as suggest some differences between crevasse environments (location). The Bonnet Carre crevasse received sediment during flood flows, while sediment input to subdeltas occurred throughout the year. However, the sediment input to Bonnet Carre appears to have been much greater per flood event than it was to the subdeltas on an annual basis (Table 12-4). It is possible that most of the sediment input to both crevasse environments occurred during flood flow. Sediment storage is also related to the sediment trapping efficiency within the system and to the amount of loss by erosion processes. These data (Table 12-4) suggest that the subdelta environments have a lower sediment retention rate that, in part, reflects the greater degree of exposure to marine erosion processes.

Table 12-4. Sediment characteristics of major crevasses.

Crevasse	Area (km^2)	Thickness (m)	Time	Sediment Rate ($m^3 \times 10^6$)	Volume ($m^3 x10^6$)[a]	%Miss.R. Flow
1a.	319	0.9	1849-74	46[b]-61[c]/flood	229-306	N.D.
1b.			1937-83	10/flood	70	3.5
2.	171	2-13	1862-1952	19-29/yr	1835	10.7
3.	78	N.D.	1891-1959	5-8/yr	372	4.5
4.	231	2-7	1845-1935	24/yr	2156	3.2
5.	44	N.D.	1891-1960	5.3/yr	365	2.7
Average	131			15/yr		5.3

1a. Bonnet Carre crevasse (Hardee,1876; Saucier,1963)
1b. Bonnet Carre spillway (Gunter,1953; Saucier,1963; COE files, New Orleans
2. Cubits Gap (Gagliano et.al.,1971)
3. Garden Island Bay (Gagliano et.al.,1971)
4. West Bay (Gagliano et.al.,1971)
5. Baptiste Collette (Gagliano et.al.,1971).
[a] conversion 1 ton = 1.01 m3
[b] based on original estimate of Hardee (1876) for L. Pontchartrain
[c] includes estimate for 91 km^2 subaerial portion of crevasse.

Table 12-5. Bonnet Carre crevasse and spillway discharge and sediment characteristics.

Year	Sediment Vol. ($m^3 x10^6$)	Sediment Rate ($m^3 x10^6$)	% Miss. River Flow
Crevasse			
1849-74	153-229	46-61/flood	N.D.
Spillway			
1937	9.5		3
	23.0		
1950	9.5		2
1973	14.1		4
1975	1.5		-
1979	6.3		-
1983	6.3		
Total 1937-83	70.2	ave. 10.0/flood	ave. 3.5

Data for Bonnet Carre (Table 12-5) show that the total effects of the spillway openings were appreciably less than those during the crevasse phase. The primary reasons for the difference in volume of sediment were most probably the duration of discharge and the change in sediment concentration (Figure 12-2) because the volume of discharge and the size of the area through which the discharge flowed were comparable. Data on the 1973 spillway opening illustrate the relative importance of sediment influx to adjacent wetlands by crevasses compared to unconfined overbank flow. Approximately 14.1 x 10^6 m^3 of sediment (Table 12-5) was discharged through the spillway into an area probably less than 259 km^2, while 18.9 x 10^6 m^3 of sediment (Table 12-2 ;1.01 m^3 = 1 ton) was available for overbank flow but would have been distributed over a 450 km length of the River. The average sediment volume that was passed through the spillway during the seven openings was about 10 x 10^6 m^3 (Table 12-5). This volume estimate may provide a gross estimate of the sediment available for diversion by spillway-type structures into wetlands during flood events.

Changes In Bedload Regime

Because there are no direct methods to measure the quantity of bed material transported by the Mississippi River, several indirect methods were used to search for possible changes in the amount transported. These approaches included computing the volume of sediment stored on active point bars and determining changes in the elevation of the channel thalweg from Red River Landing to the Gulf. In the latter, an increase in elevation over time represents aggradation, while a decrease indicates erosion of the channel floor. The point bar volume data are summarized in Table 12-6 and are subdivided into an upper section from Cairo to Red River Landing and a lower one from Red River Landing to the Gulf. During the period from 1880 to 1915, point bars along the entire River accumulated sediment and increased their size by about 20%. The River during this period was still an actively meandering and aggrading channel. The volume of sediment carried by the River may have been more than normal because of enormous volumes of bank caving associated with the New Madrid earthquake (1810-11) and increased soil erosion resulting from the deforestation and agricultural practices of the 19th and early 20th centuries. Point bars on the upper River section during subsequent periods begin to decrease in volume and, after 1935, show a 15% decline to 1975. A decrease in point bar volume below Red River Landing began after 1935 and increased to 24% after 1948. The overall decline in point bar volumes must be recognized as not necessarily representing an absolute change in the amount of bed load transported by the River; however, it reflects a shift in the location of stored bed material from the point bar to the channel floor. The major decrease in point bar volume during the 1935-48 period was preceded by major levee construction. By 1928 levee construction had eliminated crevasse breaks and had increased the volume of water confined between the levees to the extent that the 1939 flood level was 2.7 m above the bankfull stage (Winkley, 1977; Figure 12-5). Sediment removal from point bars continued to 1975, probably because of the construction of revetments from 1945 to 1970. Concrete revetments fix in place the river bank opposite a point bar, constricting the channel during flood flow, and causing the river to expend a greater portion of its energy toward the point bar side. The increased flow between levees also increases the potential for eroding and transporting sediment from the channel floor farther downstream.

A fundamental question is whether the amount of bed material that reaches the lower portion of the River, below the Old River diversion (below Red River Landing), has changed. One measure of any such change is the depth of the channel thalweg below mean sea level. Figure 12-13 compares the thalweg elevation (below msl) in 1880 with the most recent 1975 survey and shows an accumulative loss of depths between Old River and the Gulf. The loss of depth represents an accumulation of bed material on the channel floor, but, as Figure 12-13 indicates, this accumulation is not distributed equally along the River course. The thalweg from Old River to New Orleans, from km 0 to 306 (mile 0 to 190) has aggraded approximately 1.8 m/km (9.6 ft/mi), while below New Orleans the accumulation has been 0.15 m/km (0.8 ft/mi). The accumulation of sediment between Old River and New Orleans may represent bed material eroded from point bars or the channel floor in the upper River that is in transit downriver. Below New Orleans it appears that sediment is being carried downstream directly to the delta without intervening channel storage.

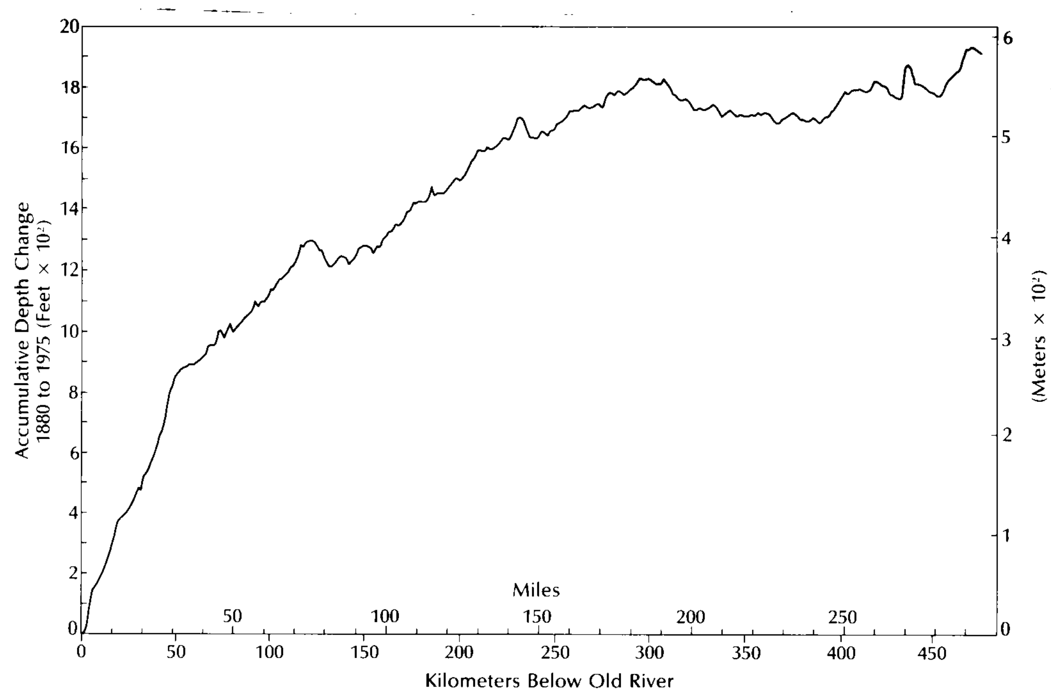

Figure 12-13. Accumulative thalweg depth (below msl) change from 1880 to 1975, Red River Landing to Gulf.

Table 12-6. Point bar volume changes.

	1880	1915	change	1935	change	1948	change	1975	change	1880-1975 change
Cairo to Red River Landing										
Total ($m^3 \times 10^6$)	38.09	45.67	7.56 (20%)	43.15	-2.52(-5.6%)	31.38	-11.77(-27%)	26.25	-5.14(-16%)	-12.04(-30%)
Ave/mile ($m^3 \times 10^6$)	0.050	0.060	0.010 (19%)	0.056	-0.003 (-5.7%)	0.048	-0.009 (-16%)	0.041	-0.007 (-15%)	-0.009 (-19%)
Red River Landing to Gulf										
Total ($m^3 \times 10^6$)	1.53	1.86	0.33 (22%)	1.95	0.09 (5%)	1.78	-0.17 (-9%)	1.35	-0.43	-0.18 (-12%)
Ave/mile ($m^3 \times 10^6$)	0.005	0.006	0.001 (22%)	0.007	0.003 (4%)	0.006	-0.0006 (-9%)	-0.002 (-24%)		-0.0007 (-13%)

In addition to this apparent shift of bed material into the lower River, there appears to have been a concomitant change in the median grain size of bed sediments (Table 12-7). The data suggest that the median grain size of bed sediments has increased from 1900 until as recently as 1950 and has since decreased.

Table 12-7. Changes in grain size of bed sediment.

Date	D_{50}(mm)	No.Samples	Source
1. Red River/ Tarbert Landing			
1932/34	0.398	7	WES Paper 17
1950	0.58	6	Latimer and Schweizer, 1951
1971-77	0.28	283	Keown, 1981
2. Donaldsonville			
1932-34	0.25	4	WES Paper 17
1975	0.10	55	Keown, 1981
3. Head of Passes			
1912	0.028	235	Shaw, 1914
1932-34	0.072	11	WES Paper 17
1938	0.11	3	WES Tech. Mem. 158-1
1975	0.05	86	Keown, 1981

Summary

The data compiled from several sources indicate that the suspended sediment load transported by the Mississippi River to the Gulf has decreased since 1950 by approximately 60%. This decrease appears to coincide with dam closures on the Missouri and Arkansas Rivers. The present volume of sediment that is available for unconfined overbank flow into areas adjacent to the River, if not restricted by artificial levees, represents approximately 3% of the total annual suspended load. Since 1950, this would have amounted to 163 x 10^6 tons of sediment. The volume of sediment available at flood stage by confined flow through crevasses may almost equal the quantity available by unconfined overbank. During the 1973 flood, the sediment volume passing through the Bonnet Carre spillway was 90% of that available by unconfined overbank flow.

There has been a decided shift in the volume of bed sediment storage from point bars to the channel floor, and this shift may allow this material to be more readily transported. Although it is difficult to quantify, one suspects that the major decrease in suspended sediment load would also be reflected by a similar decrease in bed material. Whatever changes have occurred in the upper section of the River, evidence indicates that bed sediments have been accumulating in the lower section of the River between Old River and New Orleans. This has resulted in aggradation within the thalweg of over 1.8 m/km of channel since 1880. Only minor aggradation is evident below New Orleans. Thus, the sediment accumulation above New Orleans appears to represent a wedge of sediment that is or will migrate downstream to the delta possibly without any tendency for sediment storage in this lower segment.

Grain size analyses indicate that there has been a fining of sediments within both the suspended and bedload fractions of the lower River. The loss of this coarse material may play an important role in maintaining subaerial land surrounding the delta front.

Chapter 13

MARSH ACCRETION, MINERAL SEDIMENT DEPOSITION, AND ORGANIC MATTER ACCUMULATION ALONG MAN-MADE CANALS AND NATURAL WATERWAYS: INTRODUCTION

by

Donald R. Cahoon
Ronald D. DeLaune
Ronald M. Knaus

In the subsiding environments of coastal Louisiana, continued existence of marsh habitat is particularly dependent on the ability of the marshland to maintain its elevation within the tidal range through the process of vertical accretion (DeLaune et al., 1978; Hatton et al., 1983; DeLaune, 1986). Accretion is accomplished through a combination of mineral sediment accumulation and peat formation (Figure 13-1). The two are interrelated because the influx of sediments also supplies nutrients for plant growth (DeLaune et al., 1979). Increased plant growth results in greater peat formation and increased stem density, allowing further entrapment and stabilization of sediment. There is increasing evidence of man's role in marsh deterioration. Subsidence, saltwater intrusion, and the alteration of hydrological regimes by man all affect vertical marsh accretion rates. Discussion in Chapters 14, 15, and 16 present the rates of vertical marsh accretion in natural (control) marshes and in the vicinity of OCS-impacted marshes.

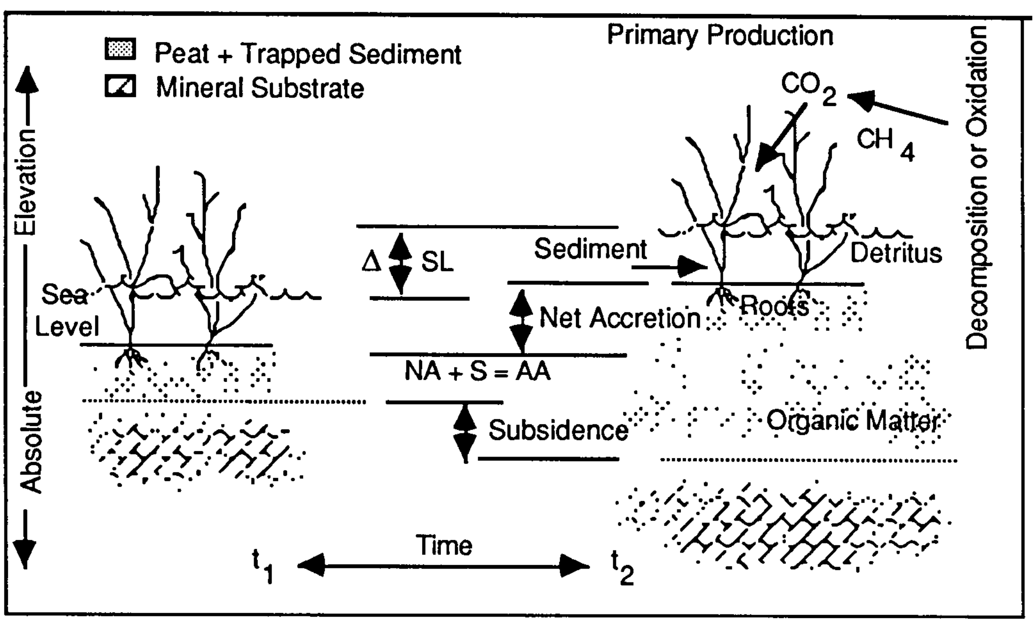

Figure 13-1. Schematic model of processes governing marsh accretion (ΔSL = sea level, NA = net accretion, S = subsidence).

It is obvious that many Louisiana Gulf Coast marshes are not accreting or aggrading rapidly enough to keep the marshes intertidal. The net result of decreased aggradation is the loss of viable sediment-created land area. A seven-year study documents a loss of 102 km^2/ yr (39 mi^2/yr) of coastal marshes in the Mississippi Deltaic Plain (Gagliano, 1981), and another study documents an additional loss of 26 km^2 per yr in the Chenier Plain (DeLaune et al., 1983). Approximately 75% of this total loss occurs as a change from emergent marsh to open water environments (bays, canals, ponds). The economic and esthetic consequences of wetland loss have been reasonably projected, and management agencies and other groups appear ready to commit resources toward resolution. However, until the hows and whys of marsh aggradation are understood, we will not know which are the most appropriate mitigating procedures.

From a coastwide view, it is evident that vertical marsh accretion rates on the order of 0.6 to 0.8 cm/year are not sufficient to maintain the elevation of the marsh, which is submerging at rates as great as 1 to 3 cm/yr (DeLaune, 1986). A continual decrease in relative elevation of the marsh with respect to water level results in the eventual conversion of marsh to an open water body. Wetland loss is a direct and rapid response to the "aggradation deficit" in many areas of coastal Louisiana.

Marsh deterioration is a complex problem, seemingly the result of numerous factors that cumulatively have a spiralling effect. Human activities, such as canal construction and leveeing, are often cited as having an indirect effect on land loss (Craig et al., 1979) by accelerating subsidence rates and reducing sediment availability and storage. Marsh aggradation in the past apparently has kept pace with subsidence and sea level rise. Through time, peat deposits 15 feet thick have accumulated in freshwater environments, thereby accreting at a rate sufficient to offset regional and local subsidence.

Several factors influencing the rate of formation and depth of Louisiana coastal marsh soils include changes in hydrology, local subsidence, eustatic sea level rise, oxidation, and the compaction of surface peats. The oxidation and decomposition process in surface peats can annually remove several hundred grams of organic carbon through CO_2 and methane emission to the atmosphere (Smith et al., 1983; DeLaune et al., 1983).

The mineral-organic matter component for individual marsh types also varies (DeLaune et al., 1987; Figure 13-2). Mean bulk density integrated over a depth of 45 cm is greatest in salt marsh with progressively lower values in inland marshes (fresh and brackish). Mineral sediment constitutes a progressively greater fraction of marsh soil solids in the salt marshes. This is a consequence of the hydrological regime. High energy marine processes and tidal action provide an abundance of reworked mineral sediments to the salt marshes. The mineral sediment content of the salt marsh is directly related to productivity (Figure 13-3). When bulk density of the root zone falls below 0.20 g cm^{-3}, salt marshes will not support appreciable growth of *Spartina alterniflora.*

Organic matter (dry weight %) is greater in fresh and brackish marshes (Figure 13-2). Organic matter on a dry weight basis constitutes an increasing fraction of soil solids as its marine influence diminishes inland from the coast. Organic matter is of greatest structural significance in low density, fresh, and brackish marsh environments. However, on a unit volume basis, the organic matter occupies the same volumes in fresh, brackish, and salt marshes (Hatton et al., 1983).

Influx of salt water into previously fresh, intermediate, and brackish marshes can also influence the rate of vertical marsh accretion by restricting plant photosynthesis, which, in turn, can reduce the organic source necessary for vertical accretion. Recent studies have

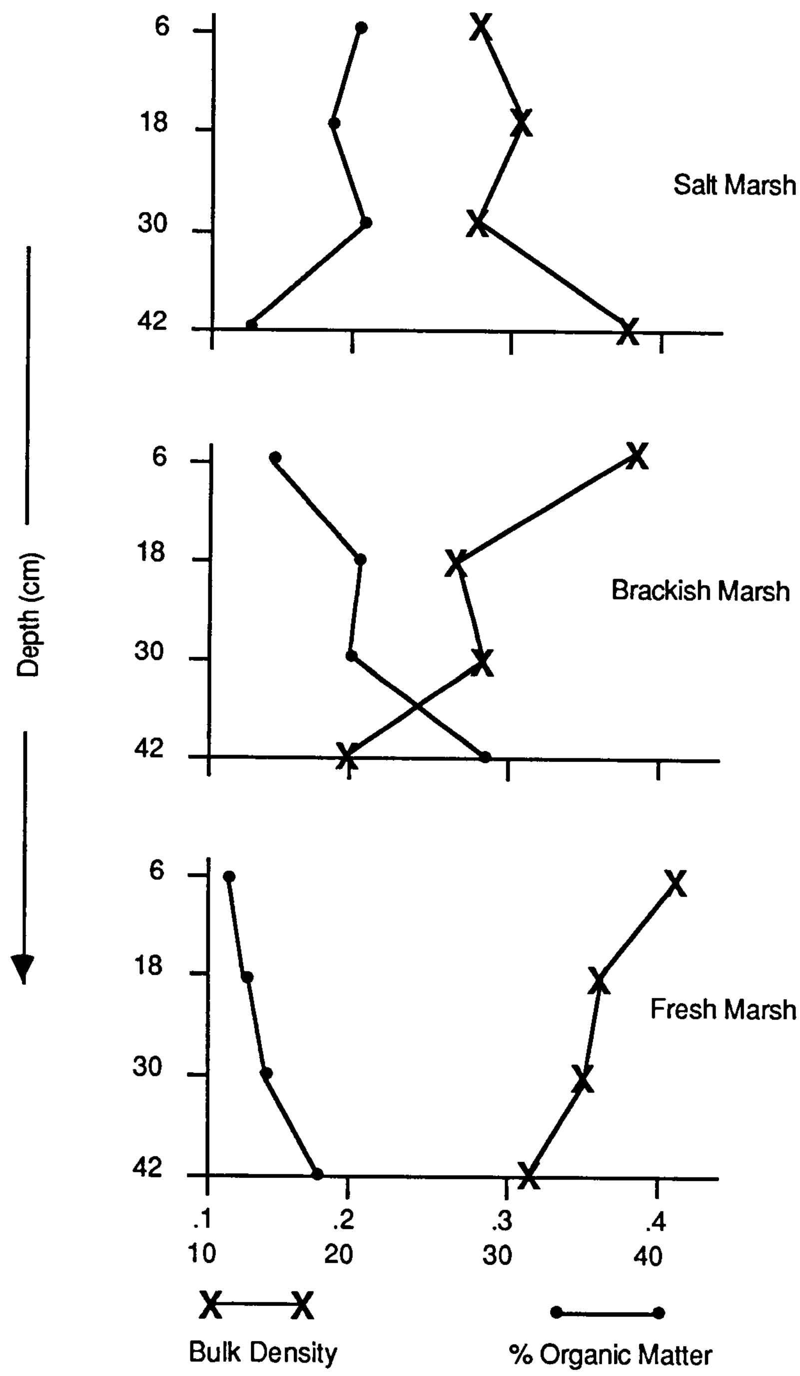

Figure 13-2. Typical bulk density and organic matter relationships.

shown that sublethal increases in salinity can reduce the normal gas exchange characteristics of Louisiana marsh vegetation (Pezeshki et al., 1987a, 1987b).

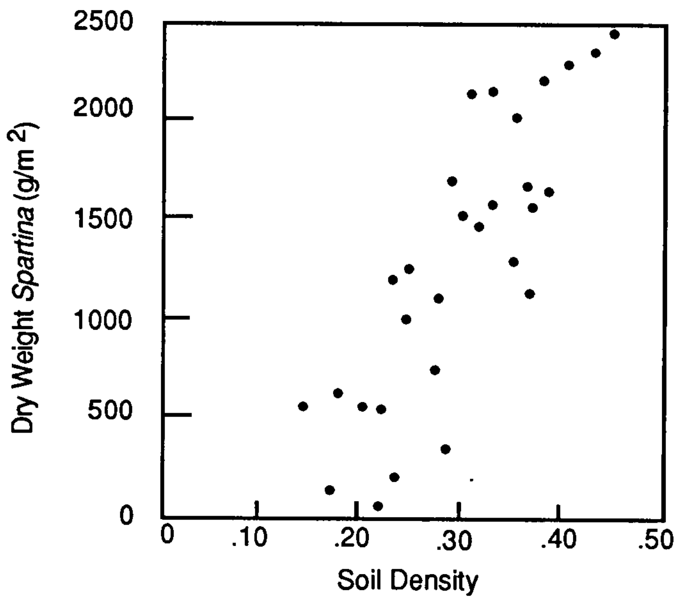

Figure 13-3. Soil bulk density and *Spartina alterniflora* relationship in salt marsh.

In the studies presented in the following sections, marsh accretion including mineral and organic matter accumulation rates were determined in the vicinity of canal and control sites using ^{137}Cs and ^{210}Pb dating techniques, clay layering techniques, and activable, stable tracer techniques. The use of these three techniques in concert enabled us to estimate historical marsh rates of vertical accretion (^{137}Cs and ^{210}Pb 25 to 100 years), recent accretion rates (stable isotope marker, clay marker), and accumulation rates in shallow water bottoms (stable isotope marker) within the same marsh region. Our studies were designed to help us answer the following questions:

(1) do man-made canals/channels influence vertical accretion and distribution of sediment across the marsh?

(2) to what extent does continuity of the spoil banks influence accretion rates?

(3) what is the relative importance of mineral and organic matter accumulation to the land building process and what influence do canals have on this ratio?

Field Site Descriptions - General Experimental Design

Field sites were established in the three major study regions of Lafourche (L), Terrebonne (T), and Cameron (C) Parishes described in Chapter 1 (see Figure 1-1) to evaluate regional influences on marsh aggradation. These sedimentological provinces represent a recently abandoned delta, an older abandoned delta and a relatively stable chenier plain environment, respectively. Areas of study were selected within the regions to test the influence of OCS pipeline canals (C), and natural waterways (N) on sedimentation processes in fresh (F), brackish (B), and saline (S) marshes.

Selection of field sites was closely coordinated between the three techniques used to measure vertical accretion. Marsh accretion rates were measured behind natural streamside levees and man-made canal spoil bank levees. Measurements in the Lafourche region were made behind continuous and discontinuous spoil bank levees and in impoundments with continuous and discontinuous levees in the Terrebonne region. (Site labels on the following figures represent the abbreviations of the region, marsh type, and levee type, respectively. For example, a canal with a continuous spoil levee in the salt marsh of Lafourche Parish is designated LSC and one with a discontinuous levee as LSCd.)

Two types of field plots were established in each region: (1) sampling stations 50 m inland from the landward edge of the natural or artificial levee; and, (2) transects running perpendicular to the waterway for at least 50 m inland. The distance of 50 m was selected because Hatton et al. (1983) demonstrated that the lateral influence of the streamside levee on vertical accretion was less than 50 m in Barataria Bay. Thus, we were ensured of encountering the full effect of the levee and its influence on marsh areas behind it.

Lafourche Region

We established field sites near Leeville, Louisiana, in the saline *Spartina alterniflora* marshes along Bayou Ferblanc, Southwestern Louisiana Canal (a navigation channel), an unnamed bayou, two OCS pipeline canals, and two oil and gas drilling site access canals (Figure 13-4). Both types of field plots were employed.

Transects were set up perpendicular to the water way at a natural (Bayou Ferblanc) and pipeline (OCS) site (LSN Transect and LSC Transect sites, respectively, in Figure 13-4), to analyze the influence of natural streamside and spoil bank levees on the sediment distribution gradient. At the transects sites, fifty meter boardwalks were constructed without disturbing the marsh surface, beginning at the marsh-side edge of the natural or man-made levee, to facilitate access and minimize disturbance of the marsh surface during stable isotope and clay marker plot preparation and sampling. Plots were established at 10 m intervals (0, 10, 20, 30, 40, and 50 m; Figure 13-5). Cores for ^{137}Cs and ^{210}Pb were sampled along a transect parallel to the boardwalk approximately every 30 m, beginning 30 m inland and extending for 100 m.

At all other sites, stable isotope and clay marker field plots were laid out 50 m behind the natural or man-made levee. A wooden sampling platform consisting of 4 poles, 2 cross pieces, and a single 8-ft plank (identical to those in Figure 13-5) was constructed parallel to the waterway at each site to minimize disturbance to the marsh surface during plot preparation and sampling. Field site selection is designed to analyze sediment and organic matter deposition rates behind natural streams (LSN), canals with continuous spoil banks (LSC), and canals with discontinuous spoil banks (LSCd). Field sites LSC2, LSC3, and LSCd2 had replicate sampling platforms (i.e., two).

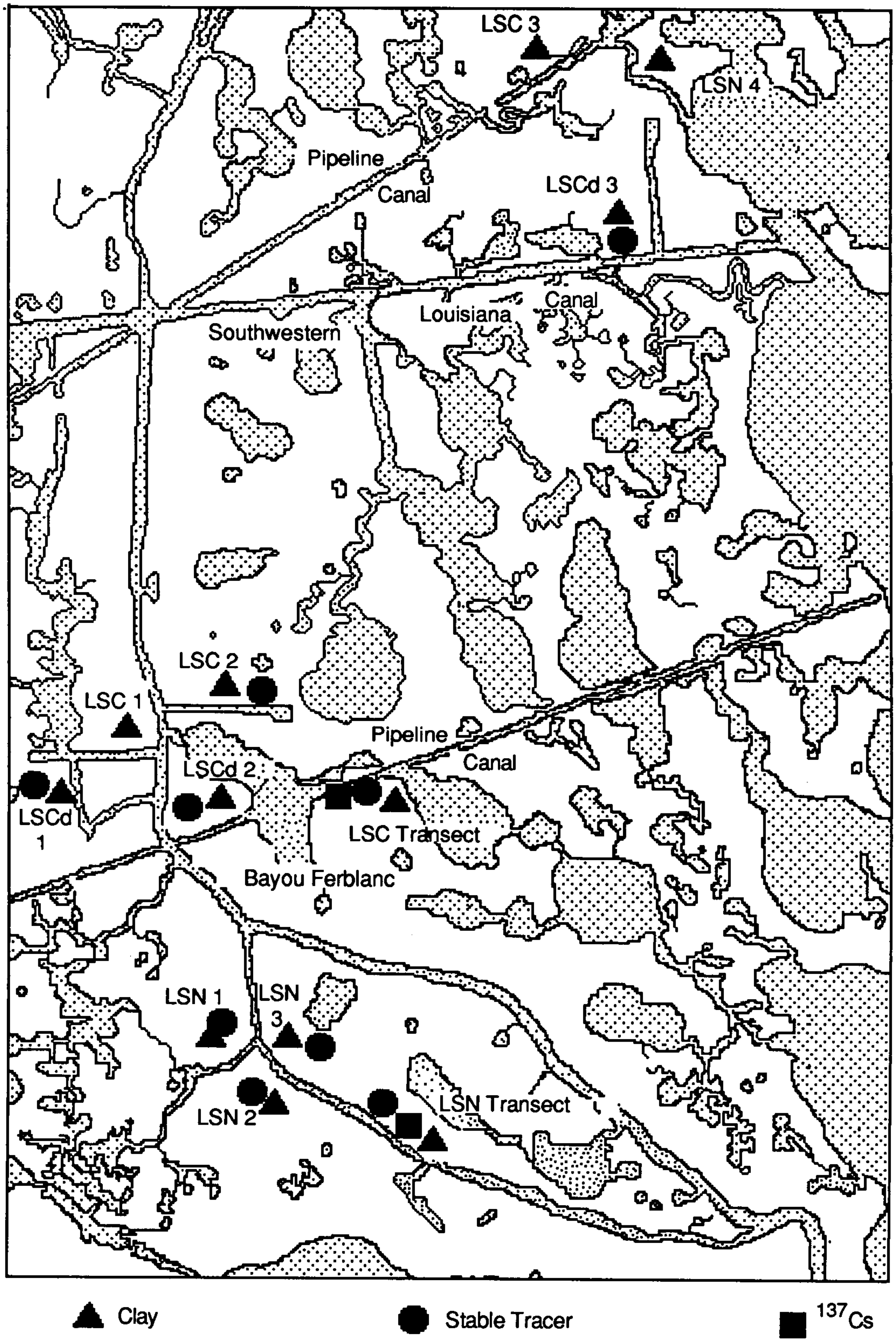

Figure 13-4. Lafourche Parish sites for the marsh aggradation investigations.

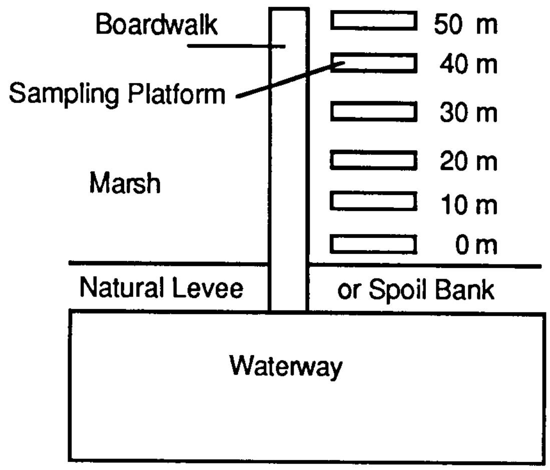

Figure 13-5. Illustration of boardwalk design employed for clay marker and stable isotope techniques in Lafourche Parish salt marsh.

Terrebonne Region

Field sites were established in the fresh marshes along the tributaries of Palmetto Bayou (Figure 13-6). The clay marker technique was not used because the highly organic, often floating, substrate was too unstable and irregular to support the clay in a distinct layer. Sampling platforms and boardwalks were not constructed for the same reason. Radioactive isotope and stable isotope transects were established along the same OCS pipeline (TFC Transect) that runs through the Lafourche region and a nearby natural waterway (TFN Transect). Samples were collected along a transect at 0, 25, and 50 m inland from the natural or man-made levee. In addition, stable isotope plots were established 50 m inland along natural waterways (TFN), pipeline canals (TFC), impounded marsh areas (TFI), and impounded marsh areas with discontinuous levees (TFIr).

Cameron Region

Field plots were established in the brackish marshes along the eastern shore of Lake Calcasieu (Figures 13-7and 13-8). This region has recently (within the last 10 years) undergone a major hydrologic alteration with the construction of a levee and associated continuous borrow canal along the eastern and southern shores of the lake. All natural and man-made waterways entering the lake in this region, except for Grand Bayou and Lambert Bayou in the Sabine National Wildlife National Refuge (NWR), have been blocked by this levee. Consequently, hydrologic exchange between the marshes and the lake has been severely altered for the area east of the lake, particularly that portion north of Sabine NWR. The only OCS pipeline in this region intersects the natural waterway, Bayou Connine Bois, on the eastern shore of the lake, north of Sabine NWR (Figure 13-7). Both the pipeline canal and the bayou have been blocked off where they join the lake. We considered selecting another region for our study, but this meant losing our aerial imagery control. Besides, there are few, if any, places on the Chenier Plain that have not been impacted similarly by levees. Therefore, field sites were established at three locations (Figures 13-7 and 13-8): (1) along the pipeline canal (CBC); (2) along Bayou Connine Bois, a natural waterway with restricted flow (CBNr); and, (3) Grand Bayou, a natural waterway without restricted flow (CBN).

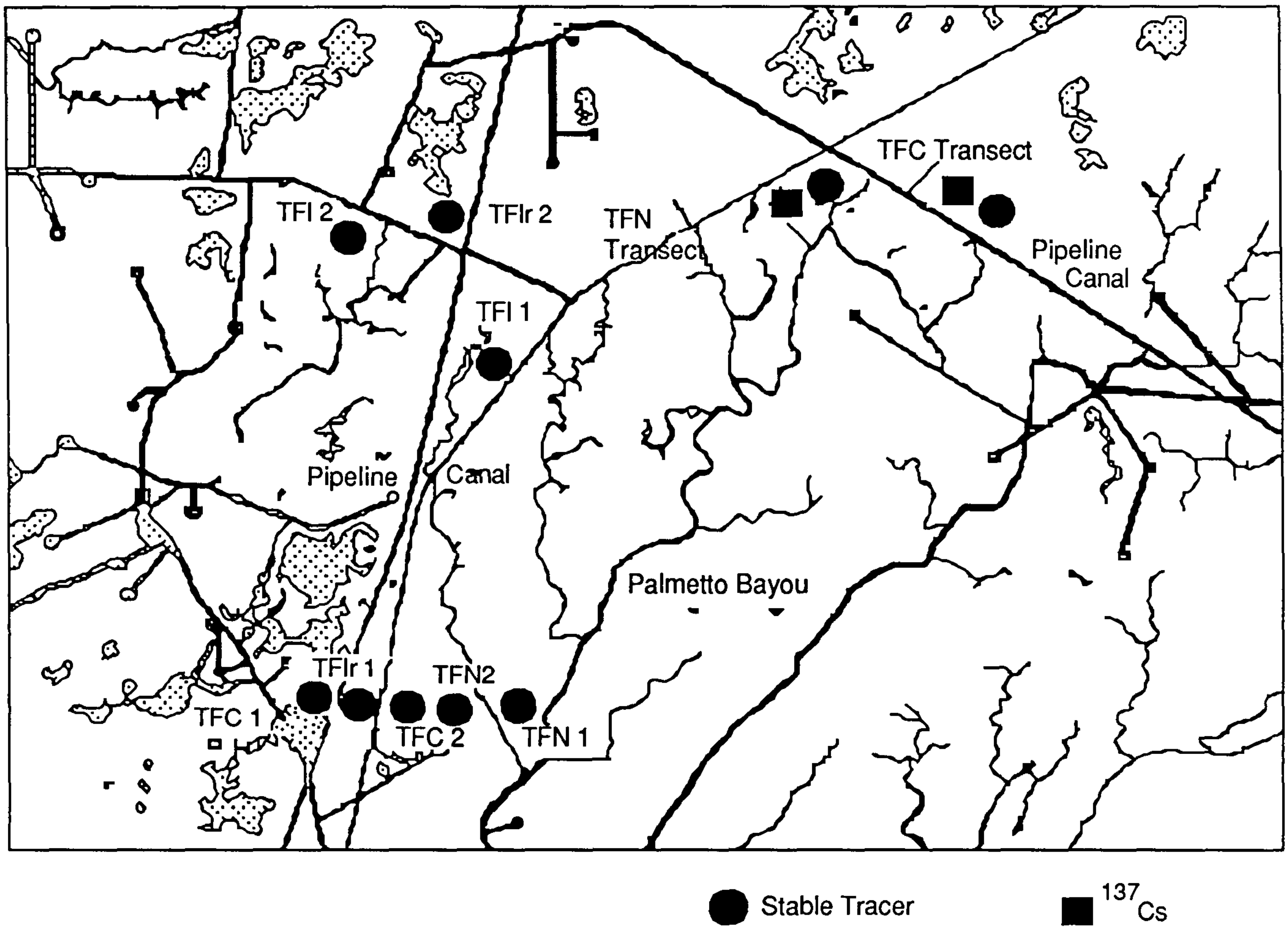

Figure 13-6. Terrebonne Parish sites for the marsh aggradation investigations.

The backmarsh comparisons performed for this study are summarized in Table 13-1 by marsh type and sediment marker technique. A total of 12, 11, and 6 field sites were established in the Lafourche, Terrebonne, and Cameron regions, respectively. Within these sites, a total of 112 clay and 51 stable isotope plots were marked, and 22 radioactive isotope plots cored (44 ^{137}Cs and 13 ^{210}Pb cores in total).

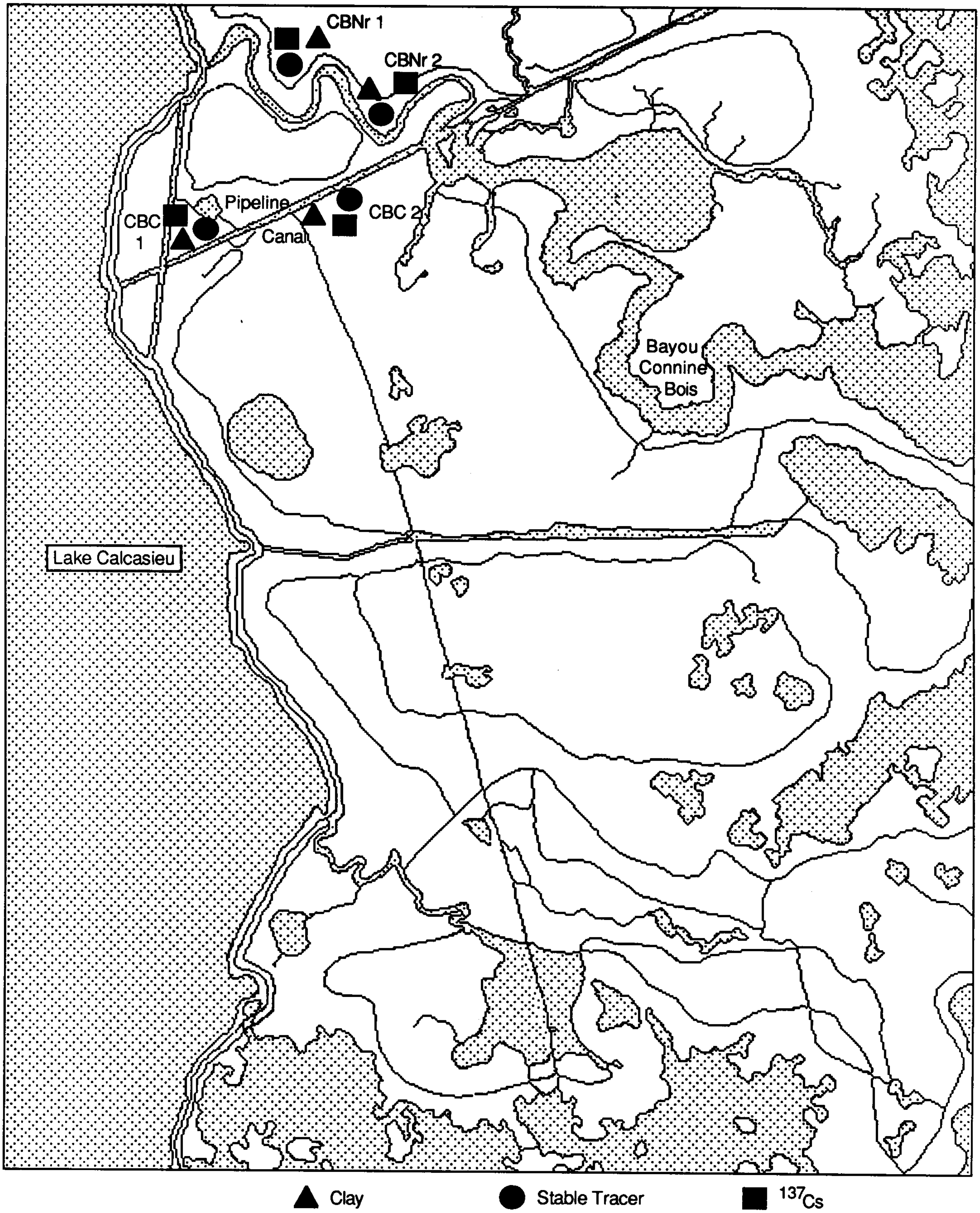

Figure 13-7. Northern Lake Calcasieu sites for the marsh aggradation investigations.

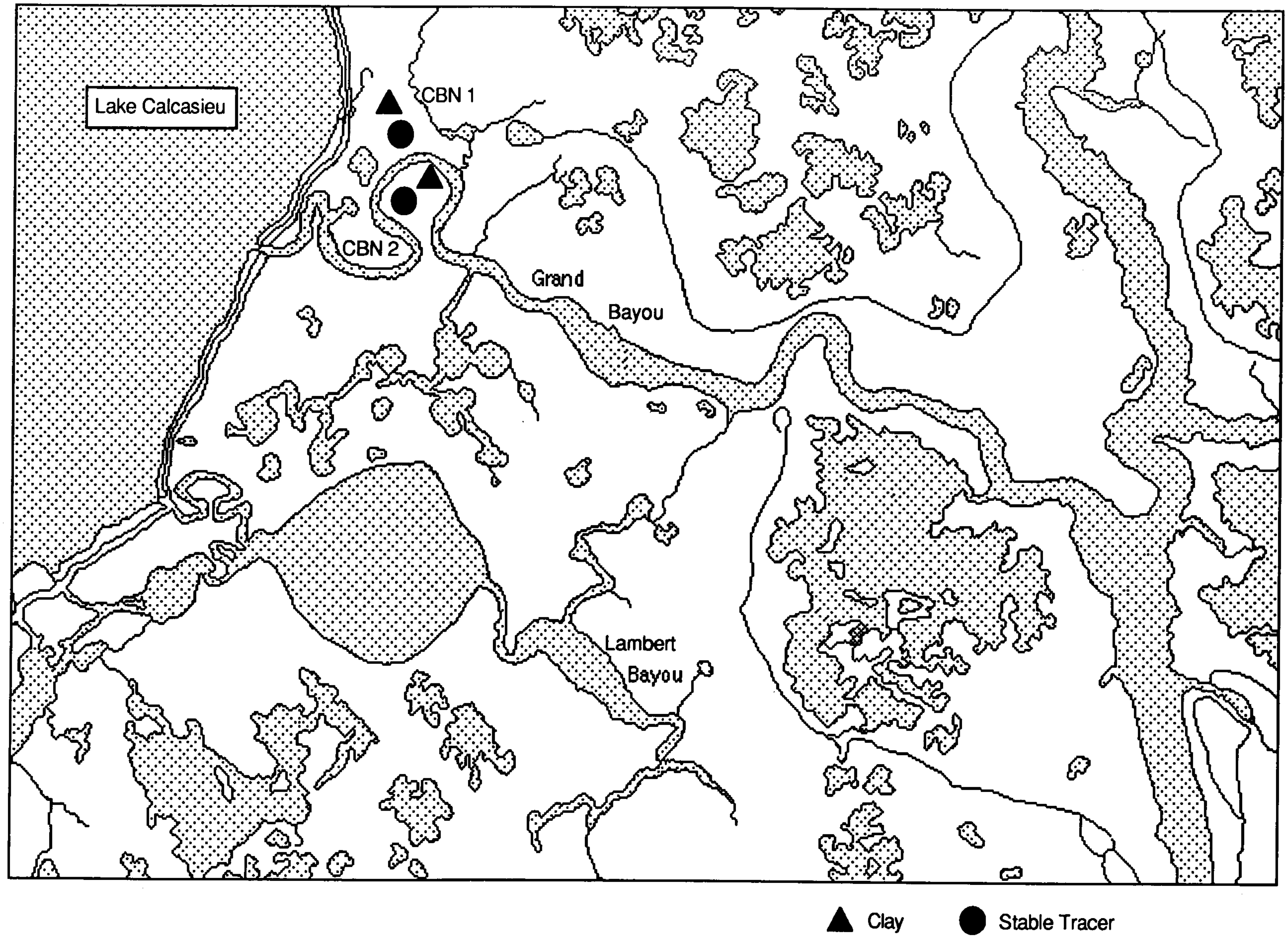

Figure 13-8. Southern Lake Calcasieu sites for the marsh aggradation investigations.

Table 13-1. Summary of the backmarsh sediment distribution comparisons, by marsh type and sediment marker technique.

Comparison	Marsh Type	Sediment Marker
Behind Natural Levee vs Continuous Spoil Levee	Salt (n=3)	Clay, Chemical
	Brackish (n=2)	Clay, Chemical, ^{137}Cs
	Fresh (n=3)	Chemical
Transect Analysis Behind Natural Levee vs Pipeline Spoil Levee (0-10-20-30-40-50m)	Salt (n=1)	Clay, Chemical , ^{137}Cs
	Fresh (n=1)	Chemical, ^{137}Cs
Behind Natural Levee vs Discontinuous Spoil Levee	Salt (n=4)	Clay, Chemical
Behind Continuous Spoil vs Discontinuous Spoil Levee	Salt (n=3)	Clay, Chemical
Within Impoundment with Flow vs Impoundment Without Flow	Fresh (n=2)	Chemical
Behind Natural Levee vs Within Impoundment with Flow	Fresh (n=2)	Chemical
Behind Natural Levee vs within Impoundment without Flow	Fresh (n=2)	Chemical

Chapter 14

MARSH ACCRETION, MINERAL SEDIMENT DEPOSITION, AND ORGANIC MATTER ACCUMULATION: ^{137}Cs AND ^{210}Pb TECHNIQUES

by

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The sites for ^{137}Cs and ^{210}Pb dating in this study are located within three major study areas in Lafourche, Terrebonne, and Cameron Parishes, South Louisiana (Figure 1-1). A control and a canal site were chosen from each study area. Sampling sites were established along the transect as representative of the marsh. Cores 15 cm in diameter and not less than 50 cm in length were taken along each transect with a thin wall cylinder for ^{137}Cs dating. Cores 100 cm in length were taken for ^{210}Pb dating. The transects started approximately 10 m from the shoreline or canal and extended not less than 50 meters into the marsh interior. Sediment accumulation and marsh aggradation rates were determined on all cores using ^{137}Cs and ^{210}Pb dating techniques. A total of 57 cores was obtained from the 3 study areas: 44 cores for ^{137}Cs dating and 13 cores for ^{210}Pb dating. Bulk density, percent carbon, and percent organic matter were determined with depth in each core, and aboveground biomass was obtained for each site.

The Lafourche Parish study area located in Southwestern Barataria Basin (Figure 13-4) included two sites: LSC transect and LSN transect. The LSC transect was taken perpendicular to an OCS pipeline canal that dissects Bayou Ferblanc. The transect consists of 8 sampling sites starting 10-12 m from the observed spoil bank with core # 1 and continuing 300 m into the marsh to core # 8. Because of the variation in the surface topography, each core pair (1 and 2, 3 and 4, 5 and 6) was taken 10-15 m apart and separated from the next pair by 20-30 m. The two additional cores (7 and 8) were taken well into the marsh interior, 300-350 m from the spoil bank. The LSN transect was taken between and flanked by two Bayou Ferblanc streams. The transect was 75 m in length and ran north starting with core #1, 10 meters from a small lake between the streams.

The two Terrebonne sites, TFC transect and TFN transect, were located near Palmetto Bayou (Figure 13-6). The TFC transect ran perpendicular to an OCS pipeline canal, was 50 m long, and consisted of six cores. The TFN transect running north from a small stream at the top of the Palmetto Bayou catchment area was 60 m long.

The two transects CBC2 and CBNr1 were taken in Cameron Parish and consisted of six cores each. Two cores where taken from each of the three additional study sites, CBC1, CBC3, and CBNr2. The CBNr1 transect ran north 70 m from Bayou Connine Bois near Lake Calcasieu (Figures 13-7 and 13-8) and transect CBC2 ran south 60 m from an OCS pipeline canal. The additional sites, CBC1 and CBC3, consisted of 2 cores each and were 50 meters from the OCS canal. The CBNr2 consisted of two cores 50 m from Bayou Connine Bois.

Materials And Methods

The isotopes ^{210}Pb and ^{137}Cs were used to determine vertical accretion rates in the vicinity of canals and control sites. Each method covers a different time scale. Accurate measurement of the radioactivity of succeeding samples taken from sediment cores provide estimates of the date at which a particular section of the core was deposited. ^{137}Cs can be used to date sediments as far back as 30 years. A variety of information can also be generated with the ^{210}Pb dating techniques that describe the sedimentary regime during the last 80-100-year period.

^{137}Cs is a product of nuclear weapons testing and does not occur naturally. Significant levels first appeared in the atmosphere in the early 1950s, with peak quantities detected in 1963/1964 (Pennington et al., 1973). ^{137}Cesium accumulates in the sediments where it decays exponentially to ^{137}Ba, with a half-life of 30 years.

The profile of ^{137}Cs with depth shows a maximum activity at depth corresponding to 1963 which then trails off to a depth equivalent to 1954 when ^{137}Cs was first detectable. Although there appears to be some migration of ^{137}Cs in the sediment, peak concentrations do not shift substantially since ^{137}Cs is rapidly absorbed by suspended particles and the clay components of sediment and soil (Jenne and Wahlberg, 1968; Robbins and Edington, 1975).

The measurement of ^{137}Cs is straightforward. Sediment cores were taken with care to minimize any compaction. The core is then sectioned (3-cm increments), dried, and ^{137}Cs activity counted with a Lithium Drifted Germanium detector and multi-channel analyzer. The rapidly accreting Louisiana coastal marshes tend to lessen the bioturbation of ^{137}Cs. Vertical accretion rates determined from ^{137}Cs dating are comparable to rates determined from marker horizons (DeLaune et al., 1983). The 1963 peak provides the best signal for determining accretion rates since there is some distortion of the 1954 marker or time ^{137}Cs first entered the system.

The isotope ^{210}Pb occurs naturally in the atmosphere caused by the radioactive decay of uranium. The longest lived immediate parent of ^{210}Pb is ^{226}Ra, with a half-life of 1,622 years. Some quantity of ^{226}Ra and a daughter, ^{222}Ra, diffuse out of the earth's crust and decay to ^{210}Pb. The ^{210}Pb is entrapped by rainfall and has a mean atmospheric residence time of 9.6 days (Francis and Chesters, 1970). Its concentration in rainwater is believed to have remained constant over a very long period, and theoretically its rate of accumulation at the earth's surface is relatively constant (Benninger, 1976). Some ^{210}Pb is produced by the decay of ^{226}Ra in the sediment column, and this fraction varies from relatively insignificant to very significant, depending on stratigraphy. A distinction must be made between the ^{210}Pb derived from the atmosphere (excess ^{210}Pb) and that derived from sediment (supported ^{210}Pb). It is the excess ^{210}Pb, unsupported, portion of total ^{210}Pb activity that we use for ^{210}Pb dating.

Unsupported ^{210}Pb is determined by subtracting the ingrown, ^{226}Ra-derived supported activity of ^{210}Pb from the total ^{210}Pb activity. Because the supported activity of ^{210}Pb is in secular equilibrium with 226Ra the following equation can be used:

$$A(^{210}Pb_{ex}) = A(^{210}Pb_{total}) - A(^{226}Ra) \qquad \text{(Eqn.14.1)}$$

^{210}Pb decays exponentially by alpha particle emission to ^{210}Bi, with a halflife of 22.3 years; its sediment concentration profile drops exponentially with depth. Thus a depth curve may be obtained given by:

$$A(^{210}Pb_{ex})_t = [A(^{210}Pb_{total})_{t=0} - A\ (^{226}Ra)]\ e^{\gamma t}. \qquad \text{(Eqn.14.2)}$$

Sections of sediment profiles were digested in acid. After the digestion the ^{210}Po was plated on a silver planchet and counted as it decays to ^{206}Pb, a stable isotope.

The method we used in this study determines the activity of ^{210}Po because ^{210}Po is a more vigorous emitter than ^{210}Pb and is in secular equilibrium with ^{210}Pb (Flynn, 1968; Robbins and Edington, 1975; Armentano and Woodwell, 1975).

Bulk density and total organic matter were determined for each core at 3-cm intervals with depth. Bulk density was determined as total dry weight per unit volume (3-cm sections, 15-cm diameter). Core sections were dried at 110 C for 48 hours. The sections were ground thoroughly and 1-2 g samples were oven-dried at 110 C overnight and then accurately weighed. Total organic matter was determined by weight loss on ignition (Hesse, 1971). The organic matter was heated (low temperature ignition) at 375 C for 16 hours (Hesse, 1971). Earlier studies report no significant loss of carbon when temperatures are kept below 450 C (Ball, 1964). Total organic carbon was determined from the regression

$$y = 0.458x - 0.4 \qquad \text{(Eqn.14.3)}$$

where y is organic carbon and x is the weight loss on ignition (Ball, 1964).

The mineral and organic contributions of total soil solids were used to calculate annual mineral and organic accumulation rates for two periods (1983-1986, and 1956-1986). Total mineral bulk density was determined for each section. These values were averaged for each of two time periods with depth. The integral mineral bulk density and organic bulk density for each period were multiplied by the accretion rate from ^{137}Cs dating to give a mineral and organic accumulation rate for each period.

Statistical analyses were performed, using the Statistical Analysis System (SAS), on the 1963-1986 sedimentation data set. Simple linear regressions were performed on each sedimentation parameter (vertical accretion rate, bulk density, mineral accumulation rate, and organic accumulation rate) with respect to distance from the canal or stream. Spearman correlation coefficients (r) from these regressions were used to express any change with distance. Comparisons between canal and control marsh were made at each location from the means of the four sedimentation parameters, using the Student's t-test with a=0.05. Corresponding comparisons were also made on the canal sites and the control marsh sites for the three study regions.

Results

Lafourche Site

Vertical marsh accretion rates at the Lafourche site determined by the ^{137}Cs dating techniques at both canal and control sites indicated a slightly faster accretion rate at the canal site than the control site, using the 1963 and 1954 marker for determining rates (Table 14-

1). Based on the 1963 marker horizon, the accretion rate was statistically greater for the canal site: 0.68 ± 0.17 cm yr-1 compared to 0.47 ± 0.09 cm yr-1 for the control site (Table 14-2). Even though the rate for the canal site was greater than that for the control marsh, it was similar to accretion rates of 0.75 cm yr-1 for a site just south of the canal site reported in an earlier study for an inland marsh (DeLaune et al., 1978).

Table 14-1. Vertical accretion rates determined using ^{137}Cs and ^{210}Pb for transects in Lafourche Parish, near a pipeline canal (LSC) and control marsh (LSN) near Bayou Ferblanc.

Core	1986-1963 ($cm\ yr^{-1}$)	1986-1954 ($cm\ yr^{-1}$)	210Pb ($cm\ yr^{-1}$)
Transect (LSC)			
1	0.63	0.92	
2	0.85	0.80	
3	0.59	0.75	0.43
4	0.59	0.66	
5	0.72	0.80	0.66
6	0.99	0.94	
7	0.46	0.66	
8	0.59	0.70	
Average	0.68±0.17	0.78±0.11	0.54±0.16
Transect (LSN)			
1	0.59	0.66	0.43
2	0.46	0.61	
3	0.46	0.61	
4	0.46	0.61	0.42
5	0.33	0.51	
6	0.52	0.70	
Average	0.47±0.09	0.62±0.06	0.42±0.01

Table 14-2. Student's *t*-test for sediment parameter between sites at each location. Means with the same letter at each location are not significantly different ($P \leq 0.05$).

Parish	Site	Mean Vertical Accretion Rate ($cm\ yr^{-1}$)	Mean Bulk Density ($g\ cm^{-3}$)	Accumulation Mean Mineral Rate ($g\ cm^{-2}\ yr^{-1}$)	Accumulation Mean Organic Rate ($g\ cm^{-2}\ yr^{-1}$)
Lafourche	LSC	0.68a	0.21a	0.14a	0.04a
	LSN	0.47b	0.16b	0.05b	0.03b
Terrebonne	TFC	0.99c	0.13c	0.07c	0.04c
	TFN	0.90c	0.17c	0.10c	0.04c
Cameron	CBC2	0.57d	0.16d	0.03d	0.03d
	CBNr1	0.56d	0.21e	0.03d	0.04d

Regression analyses did not show a change in accretion rate with distance from canal or stream (Table 14-3). Rates determined from ^{210}Pb dating were slightly less than rates determined from ^{137}Cs dating. These results indicate that vertical marsh accretion was less in the past, or that oxidation and compaction of surface peat reduces the measured accretion rate, if estimated, over a greater portion of the marsh soil profile.

Table 14-3. Correlation coefficients (Spearman) for sediment parameters regressed against distance from canal or stream.

Parish	Site	Mean Vertical Accretion Rate ($cm\ yr^{-1}$)	Mean Bulk Density ($g\ cm^{-3}$)	Accumulation Mean Mineral Rate ($g\ cm^{-2}\ yr^{-1}$)	Accumulation Mean Organic Rate ($g\ cm^{-2}\ yr^{-1}$)
Lafourche	LSC	-0.16	-0.44	-0.24	0.01
	LSN	-0.47	0.45	-0.75	-0.49
Terrebonne	TFC	0.12	-0.33	-0.61	-0.55
	TFN	-0.23	-0.98a	-0.91a	-0.40
Cameron	CBC2	-0.13	0.64	0.14	0.19
	CBNr1	-0.94a	-0.38	-0.26	-0.84[a]

[a] $P \leq 0.05$, $n = 6$

Bulk density of the marsh soil profile was greater at the canal site as compared to the control site: 0.21 g m^{-3} vs 0.16 g cm^{-3}. The control site, however, contained a greater portion of organic matter in the marsh soil (Table 14-4). Correspondingly, there were significantly greater amounts of mineral sediment deposited at the canal site, compared with the control (Table 14-2). Based on the 1963 marker horizon, an average of 0.14 g cm^{-2} yr^{-1} of mineral matter is being deposited at the canal site when compared to 0.05 g cm^{-2} yr^{-1} at the control site (Table 14-4). Annual organic matter accumulation was statistically greater at the canal site due primarily to a faster accretion rate (Table 14-2). There was 0.04 g cm^{-2} yr^{-1} accumulation at the canal site versus 0.03 g cm^{-2} yr^{-1} at the control marsh (Table 14-5). Profile analysis of sediment distribution at the canal site indicated that sediment deposition has increased in recent years. We attribute this to a change in hydrology which causes a greater flooding frequency. The short-term result for a salt marsh is a greater amount of mineral sediment deposition. The canal site under study is rapidly opening up and apparently is flooded with greater frequency than the control site.

Table 14-4. Average bulk density (BD) and organic matter for the two locations in Lafourche Parish.

Station	Transect (LSC) Marsh Cut by Tennessee Gas Pipeline Canal: Avg. BD(g cm-3) (0-42 cm)	Transect (LSC): % organic matter	Transect (LSN) Control Marsh Adjacent to Bayou Ferblanc: Avg. BD(g cm-3) (0-42 cm)	Transect (LSN): % organic matter
1	0.22	26.7	0.15	31.2
2	0.22	21.5	0.15	31.6
3	0.23	23.3	0.15	32.6
4	0.19	31.8	0.17	32.6
5	0.19	30.7	0.18	32.7
6	0.25	22.9	0.15	31.2
7	0.19	31.4	-	-
8	0.17	36.5	-	-
Average	0.21±0.03	28.1±5.3	0.16±0.01	32.0±0.73

Table 14-5. Mineral sediment and organic matter accumulation (g cm^{-2} yr^{-1}) for transects in Lafourche Parish, near pipeline canal (LSC) and control marsh (LSN) near Bayou Ferblanc.

	Mineral Sediment		Organic Matter	
Core	1986-1963 (g cm^{-2} yr^{-1})	1986-1954 (g cm^{-2} yr^{-1})	1986-1963 (g cm^{-2} yr^{-1})	1986-1954 (g cm^{-2} yr^{-1})
Transect (LSC)				
1	0.16	0.21	0.04	0.06
2	0.17	0.15	0.05	0.04
3	0.13	0.14	0.03	0.04
4	0.12	0.11	0.03	0.04
5	0.13	0.12	0.04	0.05
6	0.24	0.19	0.05	0.05
7	0.11	0.15	0.03	0.05
8	0.09	0.09	0.04	0.05
Average	0.14±0.05	0.14±0.04	0.04±0.01	0.05±0.01
Transect (LSN)				
1	0.06	0.07	0.03	0.03
2	0.06	0.07	0.03	0.03
3	0.05	0.06	0.03	0.03
4	0.06	0.07	0.03	0.04
5	0.04	0.06	0.02	0.03
6	0.05	0.06	0.03	0.04
Average	0.05±0.01	0.06±0.00	0.03±0.00	0.03±0.00

Terrebonne Site

Vertical rate of marsh accretion determined by ^{137}Cs dating at the Terrebonne site indicated no statistical difference between the canal site and control marsh site (Table 14-3). Based on the 1963 marker horizon, vertical accretion was 0.99± 0.19 cm y^{-1} in the canal site compared with 0.90 ± 0.1 cm y^{-1} for the control (Table 14-6). Similar rates showing no statistical difference were observed for each site based on rates using the 1954 marker. The sedimentation rate determined from ^{210}Pb dating was less than rates obtained from ^{137}Cs dating. The measured accretion rate using ^{137}Cs dating was similar to rates of 0.92 ± 0.15 cm yr^{-1} reported for nearby Palmetto Bayou, Creole Bayou, and Plumb Bayou (DeLaune et al., 1987).

Bulk density was greater at the control site compared with the canal site (Table 14-7). Regression analyses (Table 14-3) showed that bulk density and annual rate of mineral sediment accumulation decreased with distance from the stream for the control site. This was apparently caused by the placement of the transect at the control site which included two sample locations where the so-called "streamside effect" is evident. Streamside marsh locations typically have greater bulk densities because of greater mineral sediment deposition near the stream. When these two sampling locations were eliminated, there was no statistical difference in bulk density between canal and control sites. Organic matter percentage was higher at canal sites. Again, this trend was caused by the higher mineral sediment inputs near the stream of the control site, which, in effect, reduces the percentage composition of organic matter in the marsh soil profile.

Table 14-6. Vertical accretion rates determined using ^{137}Cs and ^{210}Pb for transects in Terrebonne Parish, near pipeline canal (TFC)and control marsh (TFN) near Palmetto Bayou.

Core	1986-1963 ($cm\ yr^{-1}$)	1986-1954 ($cm\ yr^{-1}$)	^{210}Pb ($cm\ yr^{-1}$)
Transect (TFC)			
1	0.85	0.89	0.72
2	1.11	1.03	
3	1.11	0.98	
4	0.76	0.80	
5	1.17	1.17	
6	0.91	0.89	0.78
Average	0.99±0.17	0.96±0.13	0.75±0.04
Transect (TFN)			
1	0.98	0.98	0.89
2	0.98	1.08	
3	0.85	0.84	
4	0.72	0.80	0.58
5	0.98	0.94	
6	0.89	0.89	
Average	0.90±0.10	0.92±0.10	0.73±0.22

Table 14-7. Average bulk density (BD) and organic matter for the two locations in Terrebonne Parish.

	Transect (TFC) Marsh Cut by Tennessee Gas Pipeline Canal		Transect (TFN) Control Marsh Adjacent to Palmetto Bayou	
Station	Avg. BD(g cm-3) (0-42 cm)	% organic matter	Avg. BD(g cm-3) (0-42 cm)	% organic matter
1	0.14	32.5	0.26	24.6
2	0.12	36.5	0.23	18.2
3	0.10	39.2	0.17	24.7
4	0.19	40.4	0.17	30.7
5	0.10	40.2	0.10	39.1
6	0.10	41.9	0.11	40.2
Average	0.13±0.04	38.4±3.4	0.17±0.06	29.6±8.7

Mineral sediment accumulation based on 1963 marker horizons was accumulated at the rate of 0.07 ± 0.02 g cm-2 yr-1 for the canal site and 0.10 ± 0.05 g cm^{-2} yr^{-1} for the control site. Annual organic matter accumulation rates were 0.04 ± 0.01 g m^{-2} yr^{-1} for canal site and 0.04 ± 0.01 g cm^{-2} yr^{-1} for the control (Table 14-8). If the two streamside cores were removed, there would be little difference in mineral or organic matter accumulation between the canal and control marsh soils.

Table 14-8. Mineral sediment and organic matter accumulation (g cm^{-2} yr^{-1}) for transects in Terrebonne Parish, near pipeline canal, (TFC) and control marsh (TFN)near Palmetto Bayou.

	Mineral Sediment		Organic Matter	
	1986-1963	1986-1954	1986-1963	1986-1954
Core	(g cm^{-2} yr^{-1})	(g cm^{-2} yr^{-1})	(g cm^{-2} yr^{-1})	(g cm^{-2} yr^{-1})
Transect (TFC)				
1	0.08	0.08	0.04	0.04
2	0.08	0.07	0.05	0.04
3	0.07	0.06	0.05	0.04
4	0.04	0.05	0.03	0.03
5	0.07	0.07	0.04	0.04
6	0.05	0.05	0.03	0.03
Average	0.07±0.02	0.06±0.01	0.04±0.01	0.04±0.00
Transect (TFN)				
1	0.16	0.18	0.07	0.07
2	0.16	0.20	0.03	0.04
3	0.08	0.08	0.04	0.03
4	0.06	0.07	0.04	0.04
5	0.05	0.05	0.04	0.04
6	0.06	0.06	0.04	0.04
Average	0.10±0.05	0.11±0.07	0.04±0.10	0.04±0.01

Table 14-9. Vertical accretion rates determined using ^{137}Cs and ^{210}Pb for transects in Cameron Parish, near pipeline canal (CBC2) and control marsh (CBNr1) near Bayou Connine Bois.

	1986-1963	1986-1954	^{210}Pb
Core	(cm yr^{-1})	(cm yr^{-1})	(cm yr^{-1})
Transect (CBC2			
1	0.70	0.74	
2	0.51	0.55	
3	0.57	0.60	
4	0.45	0.51	
5	0.70	0.79	0.52
6	0.51	0.60	
Average	0.57±0.10	0.63±0.11	0.52±0.0
Transect (CBNr1)			
1	0.70	0.65	
2	0.70	0.83	0.55
3	0.57	0.60	
4	0.51	0.65	
5	0.45	0.60	
6	0.45	0.59	
Average	0.56±0.11	0.65±0.09	0.55±0.0

Cameron Site

Accretion rates determined from ^{137}Cs and ^{210}Pb dating of cores from the Cameron site indicated that vertical marsh accretion rates were essentially the same at the canal transect compared with the control marsh transect (Table 14-9). Based on the 1963 ^{137}Cs marker horizon, vertical accretion rates of 0.57 ± 0.10 cm yr^{-1} and 0.56 ± 0.11 cm yr^{-1} were obtained for the canal and control sites, respectively. Rates from ^{210}Pb dating supported ^{137}Cs results. The rates from ^{210}Pb dating, however, were less than ^{137}Cs, particularly when compared to the 30-year history based on the 1954 ^{137}Cs marker horizons.

Bulk density of the marsh soil profile was greater at the control site as compared with the canal site (Table 14-10). The bulk densities presented in Table 14-10 represent average values to a depth of 42 cm. The surface bulk densities for both canal and control marsh actually were less than 0.10 g cm^{-3}. Such soil bulk density values will not support brackish marsh vegetation. If this accretion trend continues, both canal and control marsh will likely disappear.

Table 14-10. Average bulk density (BD) and organic matter for sites in Cameron Parish.

	Transect (CBC2) Marsh Cut by Tennessee Gas Pipeline Canal		Transect (CBNr1) Control Marsh Adjacent to Bayou Connine Bois	
Station	Avg. BD(g cm^{-3}) (0-42 cm)	% organic matter	Avg. BD(g cm^{-3}) (0-42 cm)	% organic matter
1	0.14	38.7	0.24	27.1
2	0.14	38.0	0.25	31.6
3	0.16	36.9	0.20	35.5
4	0.17	35.3	0.15	35.0
5	0.17	36.1	0.21	29.6
6	0.15	39.5	0.21	27.0
Average	31.0±3.7	0.16±0.01	37.4±1.6	0.21±0.04

Because of higher sediment content, as reflected in higher bulk densities at the control site, percent organic matter content was less. Actual rates of organic matter accumulation, however, were the same for canal and control sites (Table 14-11). Annual mineral sediment deposition based on 1963 marker horizons was the same for each site based on 1963 ^{137}Cs marker horizons with an average of 0.03 ± 0.01 g cm^{-2} yr^{-1} being deposited at the canal and control marsh sites. Based on the 1954 marker horizons, the amount deposited was 0.05 g cm^{-2} yr^{-1} and 0.07 g cm^{-2} yr^{-1} at the canal and control site, respectively, indicating that mineral sediment deposition has decreased in recent years or the mineral content of the marsh soil at depth has increased with time through oxidation and decomposition of plants. However, there was no statistical difference in annual rate of mineral or organic matter deposition between the two transects during the past 20-25 years (Table 14-2).

Table 14-11. Mineral sediment and organic matter accumulation (g cm^{-2} yr^{-1}) for transects in Cameron Parish, near pipeline canal (CBC2) and control marsh (CBNr1) near Bayou Connine Bois.

	Mineral Sediment		Organic Matter	
	1987-1963	1987-1954	1987-1963	1987-1954
Core	(g cm^{-2} yr^{-1})	(g cm^{-2} yr^{-1})	(g cm^{-2} yr^{-1})	(g cm^{-2} yr^{-1})
Transect (CBC2)				
1	0.04	0.05	0.04	0.03
2	0.02	0.04	0.02	0.03
3	0.04	0.05	0.03	0.04
4	0.03	0.04	0.03	0.03
5	0.04	0.06	0.04	0.05
6	0.03	0.05	0.03	0.03
Average	0.03±0.01	0.05±0.01	0.03±0.01	0.03±0.01
Transect (CBNr1)				
1	0.04	0.06	0.04	0.04
2	0.04	0.10	0.04	0.07
3	0.04	0.05	0.04	0.04
4	0.01	0.05	0.03	0.04
5	0.02	0.07	0.03	0.04
6	0.04	0.07	0.03	0.04
Average	0.07±0.02	0.04±0.00	0.04±0.01	0.03±0.01

At the three additional sites (CBC1, CBC3, and CBNr2), in the Cameron study area where two parallel cores were taken, vertical accretion rates were essentially the same for the duplicate cores (Table 14-12). Based on the 1963 ^{137}Cs marker horizons, vertical accretion rates were 0.83 ± 0.0 cm yr-1, 0.57 ± 0.0 cm yr^{-1}, and 0.48 ± 0.04 cm yr^{-1} for sites CBC1, CBC3, and CBNr2, respectively. Rates based on 1954 marker horizons were similar to those based on 1963 marker horizons. Sedimentation rates obtained using the ^{210}Pb technique were 0.44 cm yr^{-1} and 0.40 cm yr^{-1}, which is somewhat less than results obtained from using ^{137}Cs dating, which, again, indicate either oxidation or compaction of the peat layer and a more rapid rate of accretion caused by increased submergence.

Table 14-12. Vertical accretion rates determined using ^{137}Cs and ^{210}Pb for canal sites (CBC1, CBC2), and control (CBNr2) Bayou Connine Bois site.

	1987-1963	1987-1954	^{210}Pb
Core	(cm yr^{-1})	(cm yr^{-1})	(cm yr^{-1})
(CBC1)			
1	0.83	0.88	0.44
2	0.83	0.83	
Average	0.83±0.0	0.85±0.03	0.44±0.0
(CBC3)			
1	0.57	0.67	0.40
2	0.57	0.67	
Average	0.57±0.0	0.67±0.0	0.40±0.0
(CBNr2)			
1	0.51	0.60	
2	0.45	0.55	
Average	0.48±0.04	0.57±0.03	

Soil bulk densities range between 0.12 g cm-3 and 0.27 g cm^{-3} for the three sites (Table 14-13). Soil bulk densities were lower at the surface as reflected in the CBNr2 sampling site, which is currently receiving less than 0.03 g cm^{-2} yr^{-1} of mineral sediment. Such levels will not, in our opinion, support growth of brackish marsh vegetation. Soil organic matter percentage ranged between 23.5% and 39.4%. Mineral sediment input and organic matter deposition were similar to processes at the transect sites (Table 14-4).

Table 14-13. Average bulk density (BD) and organic matter for sites in Cameron Parish.

	Marsh Cut by Tennessee Gas Pipeline Canal			Control Marsh Adjacent to Bayou Connine Bois	
Site	Avg. BD(g cm-3) (0-42 cm)	% organic matter	Site	Avg. BD(g cm-3) (0-42 cm)	% organic matter
CBC1-1	0.27	23.5	CBNr2-1	0.21	29.8
CBC1-2	0.18	30.1	CBNr2-2	0.20	28.7
CBC3-1	0.15	39.4			
CBC3-2	0.12	28.7			
Average	0.18±0.06	30.4±6.62		0.20±0.01	29.2±0.8

Table 14-14. Mineral sediment and organic matter accumulation (g cm^{-2} yr^{-1}) for two sites (CBC1), and (CBC3) near pipeline canal, and (CBNr2) control marsh near Bayou Connine Bois.

	Mineral Sediment		Organic Matter	
Core	1987-1963 (g cm^{-2} yr^{-1})	1987-1954 (g cm^{-2} yr^{-1})	1987-1963 (g cm^{-2} yr^{-1})	1987-1954 (g cm^{-2} yr^{-1})
(CBC1)				
1	0.10	0.14	0.05	0.05
2	0.06	0.07	0.05	0.05
Average	0.08±0.03	0.10±0.05	0.05±0.00	0.05±0.00
(CBC3)				
1	0.03	0.04	0.03	0.03
2	0.03	0.04	0.02	0.02
Average	0.03±0.0	0.04±0.0	0.02±0.01	0.02±0.01
(CBNr2)				
1	0.02	0.05	0.03	0.04
2	0.03	0.04	0.02	0.02
Average	0.02±0.01	0.04±0.01	0.02±0.01	0.03±0.01

Summary

Marshes can maintain themselves during periods of increasing water level through organic matter accumulation and by capturing mineral sediments. Using ^{137}Cs and ^{210}Pb dating techniques in this study we have determined the comparative role of vertical marsh accretion, including organic matter and mineral sediment accumulation, for maintaining natural (control) marshes and marshes in the vicinity of the OCS-impacted marshes. Rates determined from ^{210}Pb dating were less than those obtained from ^{137}Cs dating, suggesting either oxidation and compaction of surface peats or more rapid accretion during the last 25 years.

For the Lafourche site, vertical marsh accretion rates, annual mineral deposition, and organic matter accumulation were greater at the canal site compared with control marsh location. Analyses of maps and photographs indicate that the canal site was deteriorating faster than the control marsh site. It appears that in a rapidly deteriorating salt marsh with increasing inundation, greater amounts of organic matter and mineral sediment may be deposited, even though there is a net loss of marsh through lateral erosion by widening interior marsh ponds and tidal channels. From visual observation it appears that the canal site is flooded more frequently and to a greater depth than the control marsh site. At all of the marsh locations studied, it would have been desirable to have had actual recorded data on the frequency and depth of flooding for both canal and control site for each of the three locations studied.

Considering rapid subsidence at the Lafourche site and oxidation and compaction of surface peats, neither the control marsh nor canal site will likely remain stable for any appreciable length of time. Bulk density values are low for maintenance of a viable salt marsh at each location. The canal site will likely deteriorate first caused by increased flooding.

For the Terrebonne sites (canal and control) located near the Atchafalaya delta, marsh was vertically accreting faster than the two Lafourche locations. However, there was no statistical difference in vertical accretion, mineral sediment deposition, and organic matter accumulation between the canal and control marsh. The Terrebonne marshes are accreting rapidly and receive adequate sediment for maintenance of a freshwater marsh. If salt water can be prevented from entering these areas, both canal and control site will likely remain stable.

Even though vertical accretion was essentially the same for canal and control marsh in the Cameron study area, bulk density values with depth were lower, indicating that there may not be enough mineral sediment deposited to maintain a brackish marsh at either site. The transect with distance from the canal and control site at this location also showed no difference in the rate of vertical accretion between the canal and control site. Since canal construction, there also was not any statistical difference in rates of mineral sediment deposition and organic matter accumulation.

In general, a greater variation in the marsh formation process was observed between regions than between canal and natural marsh within a region. Vertical marsh accretion was greater at the Terrebonne site, which is located near the emerging Atchafalaya delta (Table 14-15). Mineral sediment deposition since 1963 was greater for the Lafourche canal site and for the control marsh at the Terrebonne sampling site. Organic matter accumulation was essentially the same at all study areas.

In summary, based on degree of variability we could not find any major effect of canals on marsh accretionary processes, including changes in mineral and organic matter deposition patterns at the three canal sites studied. Based on this study and information obtained from previous studies of nearby marshes, any difference in vertical accretion rates in canals studied would likely be less than 0.10 cm/yr. This is supported by the fact that we also did not find any difference in marsh accretion with distance from individual canals. It is difficult to compare the rates of accretion obtained for each study area with submergence rates determined from water level increases obtained from analysis of existing tide gage data of the region for determining if individual marshes are keeping up with increases in water level. Submergence rates obtained from gage data give an indication of subsidence of the area but do not provide specific hydrological information for individual marshes (canal or natural). Based on vertical accretion rates reported, these could be aggradation deficits. Unfortunately, there are no data on water level differences between sites to compare vertical accretion rates. One effect of canals could be alteration in hydrology and subsequent water level increases, which could cause aggradation deficits, even though vertical accretion may be the same for canal and control marsh. Marsh building is a complex process with many variables. Any region or location must be studied in greater detail to truly identify controlling factors.

Table 14-15. Student's *t*-test for sediment parameter means among canal sites and control marsh sites. Means with the same letter are not significantly different ($P \leq 0.05$).

			Accumulation	
Site	Mean Vertical Accretion Rate ($cm\ yr^{-1}$)	Mean Bulk Density ($g\ cm^{-3}$)	Mean Mineral Rate ($g\ cm^{-2}\ yr^{-1}$)	Mean Organic Rate ($g\ cm^{-2}\ yr^{-1}$)
Canal				
LSC	0.68a	0.21a	0.14a	0.04a
TFC	0.99b	0.13b	0.07b	0.04a
CBC2	0.57a	0.16b	0.03b	0.03a
Control Marsh				
LSN	0.47c	0.16c	0.05c	0.03c
TFN	0.90d	0.17c	0.10d	0.04c
CBNr1	0.56c	0.21c	0.03c	0.04c

Chapter 15

MARSH ACCRETION, MINERAL SEDIMENT DEPOSITION, AND ORGANIC MATTER ACCUMULATION: CLAY MARKER HORIZON TECHNIQUE

by

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The use of white-clay artificial marker horizons is an effective method for measuring recent sediment accumulation in coastal Louisiana wetlands (Baumann et al., 1984). A major advantage over isotope marker techniques is that it is comparatively inexpensive because it requires less complex equipment for collecting and processing samples. For these reasons and because of the highly variable nature of coastal wetland habitats, more clay marker horizon plots were established and sampled than stable and radioactive isotope plots. This greater flexibility in sample size was used in those regions where the technique is best suited: the salt marsh in Lafourche Parish and the brackish marsh in Cameron Parish. In all, 88 white-clay marker horizons (inland plus transect stations combined) were established at the Lafourche Parish site and 24 at the Cameron Parish site. The Lafourche plots were established during June and July, 1986, while the Cameron plots were laid out in November, 1986, and January, 1987.

Methods and Experimental Design

Both types of field plots (50-m inland stations used to compare accretion rates behind different types of waterways and transect stations at 0, 10, 20, 30, 40, and 50 m to analyze accretion rates over distance behind different types of waterways) were established in Lafourche Parish. In Cameron Parish, the accretion analysis is based only on comparison of inland (50-m) stations. Marker horizons were laid down in 50 x 50 cm plots at all sites. Four plots were established at all inland sites–one on the streamside (s), the non-streamside (ns), and off the end of the platform (e), plus one away from the platform (ab) (Figure 15-1). This arrangement was designed to test for any possible platform effects on the distribution and accumulation of matter. At the Lafourche transect sites, only the three plots around the platform were laid down so that all plots could be accessed without walking on the marsh surface. To facilitate finding the plots, all marker horizon plots were laid down at a uniform distance from the edge of the platform (20 cm) and the position of the plot was recorded. Also, a single stake made of plastic-coated aluminum clothesline wire (3 mm diameter, which is less than the diameter of a *Spartina alterniflora* stem) was placed near the plot to mark its location.

The Lafourche marker horizons were cored after 6 and 12 months and the Cameron plots after 6 (CBNr and CBC) or 8 and 12 (CBN) months. Thin-walled aluminum beverage cans (6 cm diameter, with the tops cut off and holes punched in the bottom to allow air to escape as the core is taken), were used to take 8- to 10-cm long cores through the white clay marker horizon. A single core was taken from a previously unsampled quadrant of every plot during each sampling event, and the hole made by the core tested for the presence of marker clay. If there was no sign of marker clay in the hole, a second or even third core was taken. However, in order to limit disturbance in the plot, a core was retained after three or four attempts. Once taken, the bottom of the core was sealed with parafilm, while still in the can and stored in a vertical position.

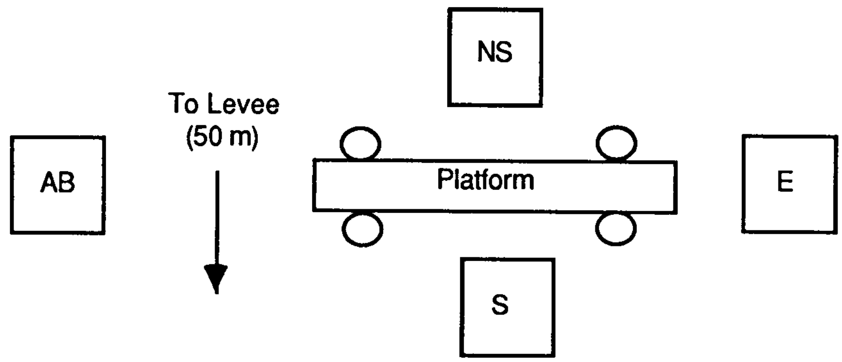


Figure 15-1. Illustration of arrangement of 0.25 m^2 (50 cm x 50 cm) clay marker horizon plots around the sampling platform at an inland sampling station. The platform is laid out parallel to the waterway 50 m inland from the landward edge of the natural or artificial levee. Drawing not to scale.

Cores were placed immediately in a freezer upon returning to the laboratory. The frozen cores were removed from the cans by slicing them in half longitudinally. The cut surface of the core was scraped smooth with a razor blade, and the depth of sediment accumulated on top of the marker clay was measured with a caliper at the center of the core and one cm to either side of center. Hence, only the center one-third of the core was measured, and the physically disrupted outer edges of the core were avoided.

The bulk density of the top 2 cm of marsh was estimated for each region, during the summer of 1987, from two cores at each site. A 60-cc syringe with the tip cut off was used to sample adjacent to the marker horizon plots. The percent mineral and organic matter were estimated for each bulk density core by standard laboratory procedures. The samples were ashed at 375°C for 16 hours according to the procedures outlined by Hesse (1971). Percent mineral and organic matter of the material deposited atop the marker horizon was not estimated because of the possibility of contamination from the clay marker.

Data were analyzed with SAS ANOVA and NPAR1WAY (SAS, 1985c) statistical programs . All 6-month data sets and average transect values were analyzed by ANOVA. Because of smaller sample sizes all bulk density (including percent organic and mineral data), Cameron, and 12-month Lafourche data sets (50-m and transect estimates) were analyzed by Wilcoxon (Rank Sums) non-parametric methods.

Results

Marker Horizon–Coring Technique

A core was considered successful if the marker clay was present in a clearly distinct layer. A core was not successful for one of three reasons: (1) clay was in the core but was badly disrupted by biological activity (i.e., bioturbation by crabs, bivalves, birds, and other organisms); (2) the core tube missed the plot; or (3) the marker horizon eroded. If there was no clay present in the core and marsh vegetation was present at the site, it was difficult to determine whether the horizon eroded or the plot was missed. However, because we took great care marking the plot location and did not use a core until marker clay was found in the hole (or three or four attempts were made to find the clay marker), it was probable that the clay marker horizon was no longer present.

The success rate varied regionally, locally, and over time. In the salt marsh of Lafourche Parish, coring success during the first sampling event (6-month) averaged 75% for all waterway comparisons, but only 55% in Cameron Parish. In Lafourche Parish, the technique was twice as successful at the sites located south of Southwestern Louisiana Canal (a navigation channel that bisects the salt marsh on an east-west line, Figure 13-4) compared to sites located north of the navigation channel (Table 15-1). After 12 months, 3 to 4 cores were taken at every plot north of the channel because we found little or no sign of the marker clay. At the Canal-d site, there was no sign of marker clay at any of the four plots. South of the navigation channel, coring at the Canal-c sites was two-thirds the success rate at the Natural and Canal-d sites. However, the success rate at Canal-c waterway remained constant after 12 months while the Natural and Canal-d sites decreased to a comparable level.

Table 15-1. The success rate (%) of single cores taken from each replicate clay marker horizon plot behind streamside levees (Natural) and continuous (Canal-c) and discontinuous (Canal-d) canal spoil levees in Lafourche Parish and hydrologically restricted natural (Natural-r) and continuous canal spoil levees in Cameron Parish. (A core is considered successful if a distinct layer of marker clay is evident.)

Parish	Waterway	Number of Plots	Time Interval 6 months	Time Interval 12 months
Lafourche	Non-transect			
	North of SW Louisiana Canal			
	Natural	4	25	25
	Canal-c	4	50	50
	Canal-d	4	50	0
	Average		42	25
	South of SW Louisiana Canal			
	Natural	12	92	67
	Canal-c	8	63	63
	Canal-d	8	100	50
	Average		86	61
	Transect			
	Natural	18	83	72
	Canal-d	8	78	61
	Average		81	67
Lafourche Parish: Grand Mean			76	58
Cameron				
	Natural	6	33	100
	Natural-r	8	63	ns
	Canal-c	8	63	ns
	Average		55	–

The coring success rate after 12 months in Lafourche Parish decreased at all waterways compared to the 6-month estimates. At one site (LSCd2) south of the navigation channel, a section of marsh disappeared along with one of the plots (plot e) after the six-month sampling. Plot e was on a small point of marsh surrounded on three sides by an open water pond. After six months, 21.9 mm of material had accumulated on this plot. After twelve months, the pond had expanded in area such that the point of marsh was then open water, with no remaining sign of the plot or stakes.

The vast majority of unsuccessful cores failed because marker clay was not present in the core. In Lafourche Parish, 16% of the six-month unsuccessful cores had marker clay present that was too disrupted to measure accurately. The remaining 84% had no marker present at all. After twelve months, only 7% of the failed cores were unmeasurable because of bioturbation of the marker clay, while 93% of the failed cores had no visible marker horizon present. In Cameron Parish, marker clay was present in seven out of eight failed cores, but it could not be measured accurately.

Lafourche Parish

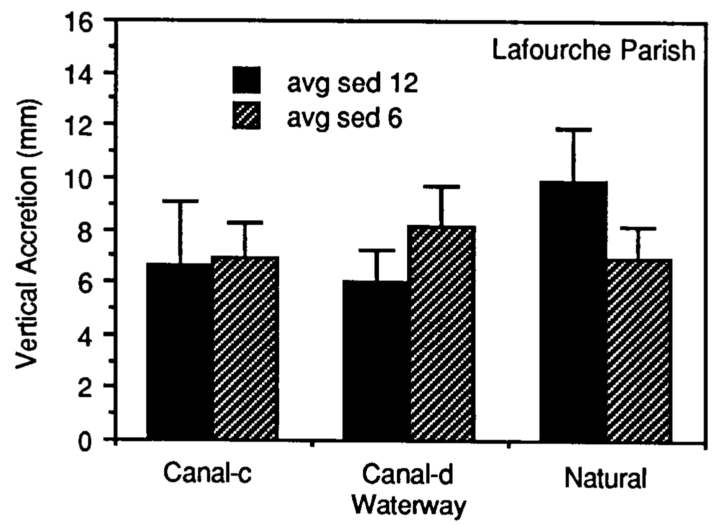
Waterway Analysis. The vertical accretion rate of recently deposited marsh sediments did not differ between the natural bayou, canal with continuous spoil levee (canal-c), or canal with discontinuous spoil levee (canal-d) for either the 6- or 12- month sampling intervals ($P = 0.75$ and $P = 0.54$, respectively; Figure 15-2). The mean annual accretion rate estimated from 12-month cores ranged from 0.60-0.99 cm yr-1 and was essentially the same as the estimates for six months. The presence of a sampling platform appears to have had no effect on vertical accretion rates or accumulation of mineral and organic matter because there was no significant difference between replicate plots around the platforms after twelve months along the three waterway types (natural, $P = 0.35$, 0.62, and 0.66, respectively; canal-c, $P = 0.44$, 0.42, 0.42, respectively; canal-d, $P = 0.41$, 0.63, and 0.47, respectively; Figure 15-2).

The bulk density of the upper 2 cm of marsh soil ranged between 0.10-0.18 g cm^{-3} and was significantly lower behind the discontinuous spoil banks than behind the natural and Canal-c sites ($P = 0.03$; Figure 15-3). However, the percent organic matter was the same for all 3 waterway types after 12 months ($P = 0.14$ and 0.21, respectively). Consequently, there was no significant difference between waterways in mineral or organic matter accumulation ($P = 0.18$ and 0.41, respectively; Table 15-2).

Table 15-2. Mineral sediment and organic matter accumulation behind streamside levees (Natural), and continuous (Canal-c) and discontinuous (Canal-d) canal spoil levees in a Lafourche Parish salt marsh (means ± 1 SE, n = number of sites). The vertical accretion estimates are based on twelve month cores.

Waterway	n	Vertical accretion cm yr^{-1}	Bulk density g cm^{-3}	%min/%org	Accumulation Mineral g $cm^{-2}yr^{-1}$	Accumulation Organic g cm^{-2} yr^{-1}
Canal-c	3	0.66±0.25	0.14±.008	70/30	0.06 ± 0.02	0.03 ± 0.01
Canal-d	3	0.60±0.12	0.10±.007	64/36	0.04 ± 0.01	0.02 ± 0.01
Natural	4	0.99±0.20	0.18±.038	74/26	0.13 ± 0.04	0.05 ± 0.01

Even though the three waterway types did not differ statistically in vertical accretion or mineral accumulation, the natural sites had consistently higher values for both variables than the Canal-d sites. The small 12-month sample size (18 cores with 45% success rate,



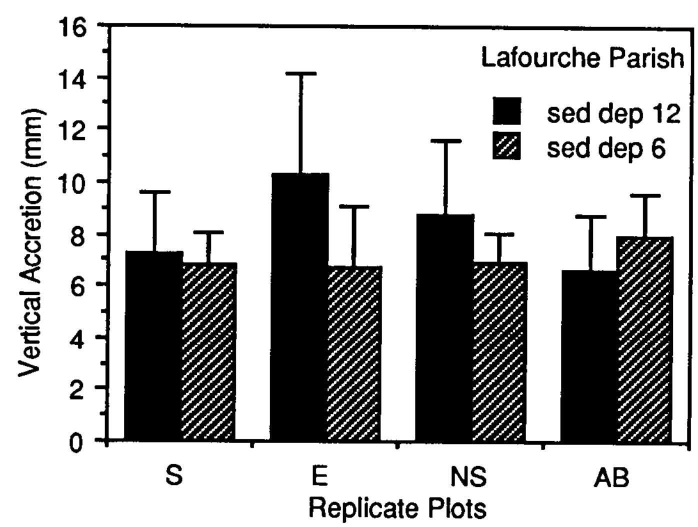
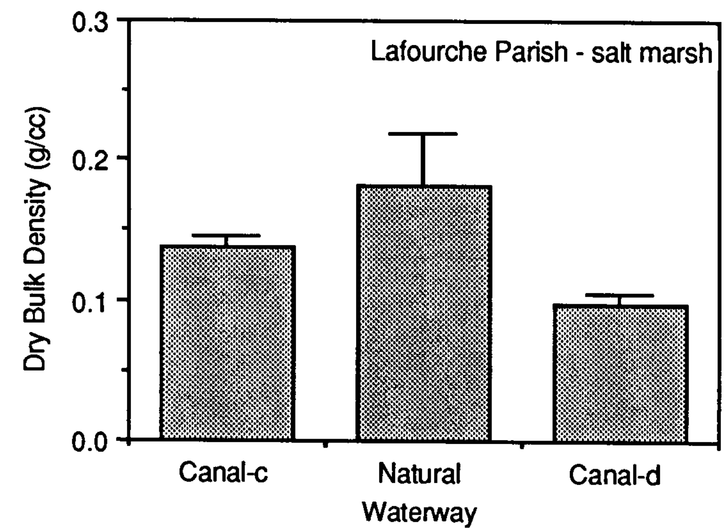


Figure 15-2. Sediment deposition rates behind the streamside levee (natural), continuous canal spoil bank levee (Canal-c) and discontinuous spoil bank levee (canal-d) (upper graph) and in the replicate plots around the sampling platforms after 6 and 12 months in the Lafourche Parish salt marsh (lower graph). Values presented are means ± 1 SE.



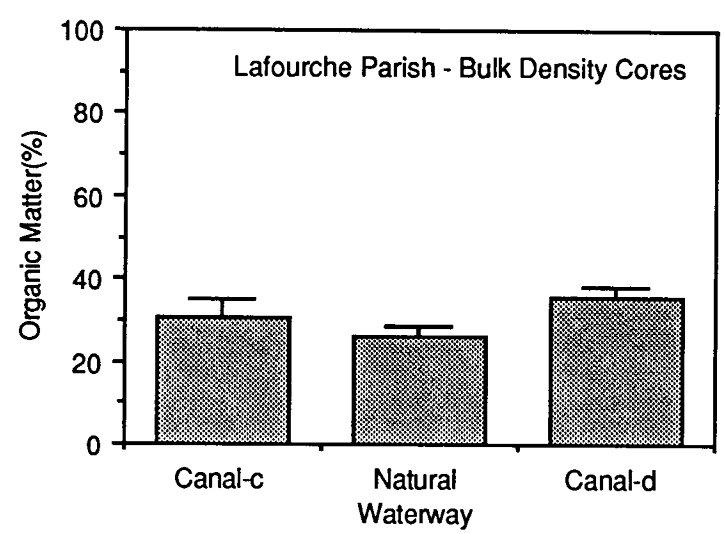


Figure 15-3. Dry bulk density (upper graph) and percent organic matter (lower graph) of the top 2 cm of marsh substrate behind the streamside levee (natural), continuous canal spoil bank levee (Canal-c) and discontinuous spoil bank levee (Canal-d) in the Lafourche Parish salt marsh. Values are means ± 1 SE.

over 10 waterway sites) and large variances made it impossible statistically to detect differences ≤ 4 mm in vertical accretion and ≤ 0.09 g cm^{-2} yr^{-1} in mineral accumulation at the 95 % confidence level.

Distance Analysis. Unlike the waterway comparison above, the transect average (i.e., average over all distances) for vertical accretion and bulk density was significantly higher for the Canal-d transect site than the Natural transect site ($P = 0.021$ and 0.018, respectively; see Figure 15-4). The 6-month vertical accretion average was also significantly higher at the Canal-d site (1.59 ± 0.22 versus 0.56 ± 0.09 cm yr^{-1}; $P = 0.001$). The transect average of percent organic matter differed significantly between waterways ($P = 0.0001$; Figure 15-5) as did the transect average for mineral accumulation (0.18 ± 0.02 vs 0.08 ± 0.03 g cm-2 yr-1, respectively, p = 0.0151, Table 15-3).

Table 15-3. Annual rates of mineral sediment and organic matter accumulation behind streamside levee (Natural waterway) and discontinuous man-made canal spoil levee (Canal-d) in a Lafourche Parish salt marsh (means ± 1 SE; n = number of successful cores).

Waterway	Distance	n	Vertical accretion[a] cm yr^{-1}	Bulk density[b] g cm^{-3}	%min/%org	Accumulation Mineral g $cm^{-2}yr^{-1}$	Accumulation Organic g cm^{-2} yr^{-1}
Natural							
	0	2	1.69	0.235	72/28	0.29 ± 0.14	0.12 ± 0.06
	10	2	0.54	0.12	67/33	0.05 ± 0.02	0.02 ± 0.01
	20	2	0.46	0.13	62/38	0.04 ± 0.01	0.03 ± 0.01
	30	3	0.73	0.11	54/46	0.05 ± 0.01	0.03 ± 0.01
	40	1	0.96	0.11	58/42	0.06	0.04
	50	3	0.28	0.125	58/42	0.02 ± 0.01	0.02 ± 0.01
Mean			0.72±0.17	0.14±0.13		0.08 ± 0.03	0.04 ± 0.01
Canal-d							
	0	3	0.90	0.15	79/21	0.11 ± 0.04	0.03 ± 0.01
	10	1	1.51	0.20	82/18	0.25	0.05
	20	2	1.34	0.18	80/20	0.19 ± 0.10	0.05 ± 0.03
	30	3	1.73	0.155	79/21	0.21 ± 0.01	0.06 ± 0.01
	40	1	1.20	0.20	81/19	0.19	0.05
	50	1	1.17	0.205	81/19	0.19	0.19
Mean			1.31±0.16	0.18±0.008		0.18 ± 0.02	0.05 ± 0.01

[a] Means from Figure 15-6
[b] Means from Figure 15-7

There was no significant difference in vertical accretion between replicate plots in the vicinity of the platforms (canal: s = 1.00 ± 0.3, e = 1.47 ± 0.17, ns = 1.50 ± 0.26, P = 0.4925; bayou: s = 0.49 ± 0.14, e = 1.04 ± 0.38, ns = 0.62 ± 0.29, P = 0.33). Mineral and organic accumulation replicate measurements also were non-significantly different along the canal and bayou transects (P = 0.5606 and 0.2013; and P = 0.4039 and 0.3453, respectively).

Within each transect, there was no significant difference with distance in vertical accretion and percent organic matter for both the 6- (P = 0.06 and 0.19, respectively) and 12-month cores (P = 0.42 and 0.24, respectively; Figures 15-6 and 15-7). At the canal site, bulk density did not differ with distance (P = 0.07), although the differences were

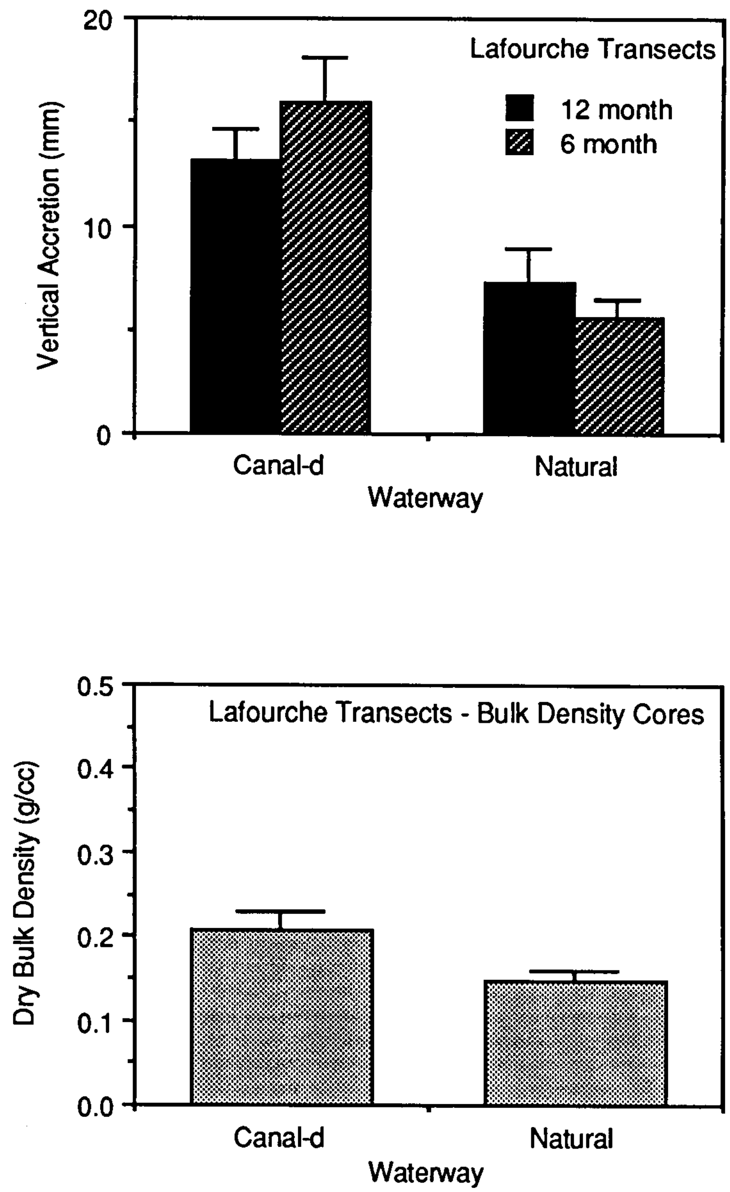


Figure 15-4. Sediment deposition rates (upper graph) and bulk density estimates (lower graph) (average for the entire transect) for the transect sites in the Lafourche Parish salt marsh. The Canal-d means of sediment deposition (both the 6- and 12-month) and bulk density are significantly higher at the 95% level. Values presented are means ± 1 SE.

significant at the 10% level. The largest mean difference (0.22 g cm^{-3}) exists between the spoil bank site, and 0 and 30 m behind it. It is highly likely that spoil bank bulk density could be nearly double the values in the marsh because the spoil levee is constructed of earthen material removed from as deep as 2 m below the live root zone of marsh vegetation. At the bayou site, bulk density also did not differ with distance (P = 0.16). Consequently, mineral and organic matter accumulation did not differ with distance for both the bayou (P = 0.14 and 0.21, respectively) and canal (P = 0.37 and 0.57, respectively) transects (Table 15-3) because vertical accretion and percent organic matter (P = 0.22, canal; P = 0.16, natural) did not change significantly with distance (Figure 15-7).

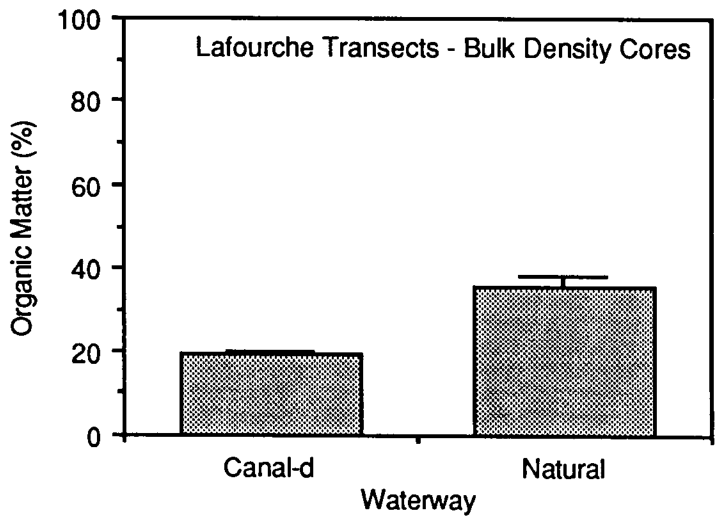


Figure 15-5. Organic matter content of the top 2 cm of the marsh substrate (average for the entire transect) for the transect sites in the Lafourche Parish salt marsh. Values presented are means ± 1 SE.

Cameron Parish

The vertical accretion rate of recently deposited marsh sediments in the brackish marsh at Cameron Parish (0.0-0.25 cm) was lower than the rate for the Lafourche Parish salt marsh after six months. Vertical accretion rates behind the hydrologically restricted waterways (Canal-c and Natural-r) in the northern part of the region were not significantly different from one another but were significantly lower than vertical accretion rates behind the Natural (control) waterway in Sabine NWR (P = 0.0419; Figure 15-8). The vertical accretion estimates from the single coring effort (six to eight months) in Cameron Parish are presented in Figure 15-8 as mm of accretion per sampling period. The six- and eight-month estimates were not adjusted to an annual basis because of the potential error involved in sampling only a portion of the annual hydrologic-climatologic cycle. The Natural waterway cores were taken eight months after the marker horizons were laid down, while the other sites were sampled six months after marking. Nine of the 12 successful cores had no measurable sediment accumulation. Also, the bulk density and organic matter content of the top 2 cm of marsh soil did not differ significantly between the 3 waterways (P = 0.1538 and P = 0.2921, respectively; Figure 15-9).

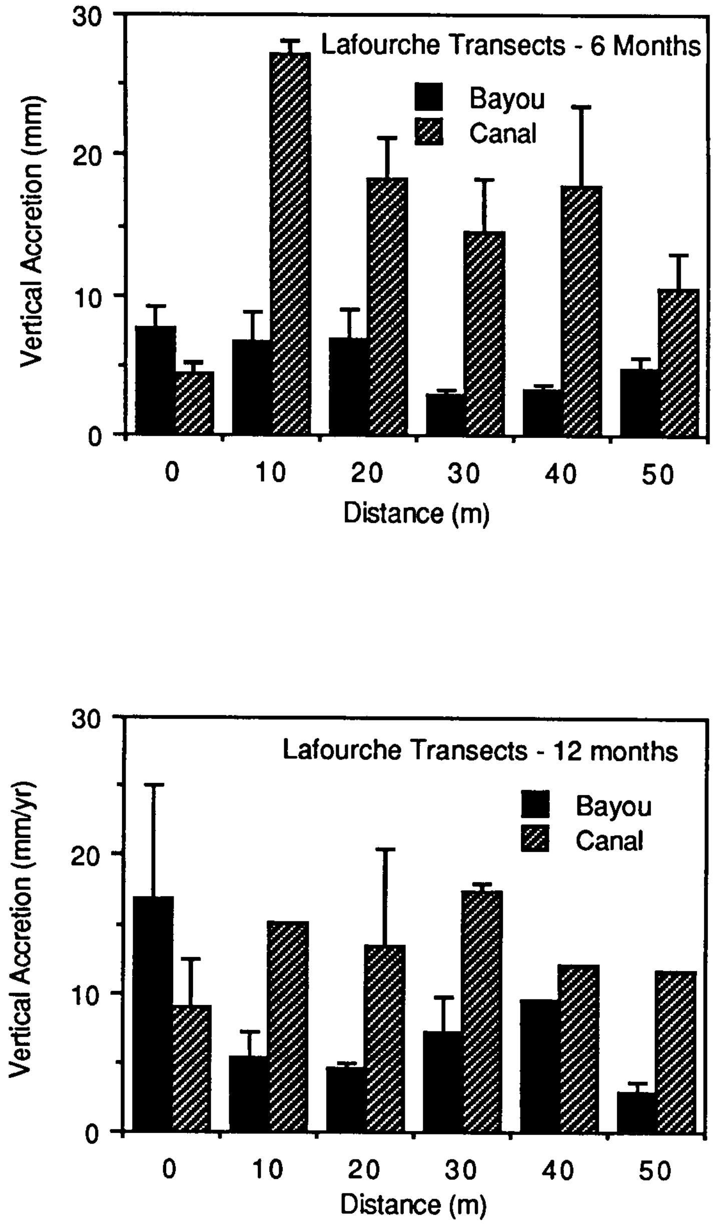


Figure 15-6. Sediment deposition along the canal and bayou transects after 6 and 12 months in the Lafourche Parish salt marsh. Values presented are means ± 1 SE.

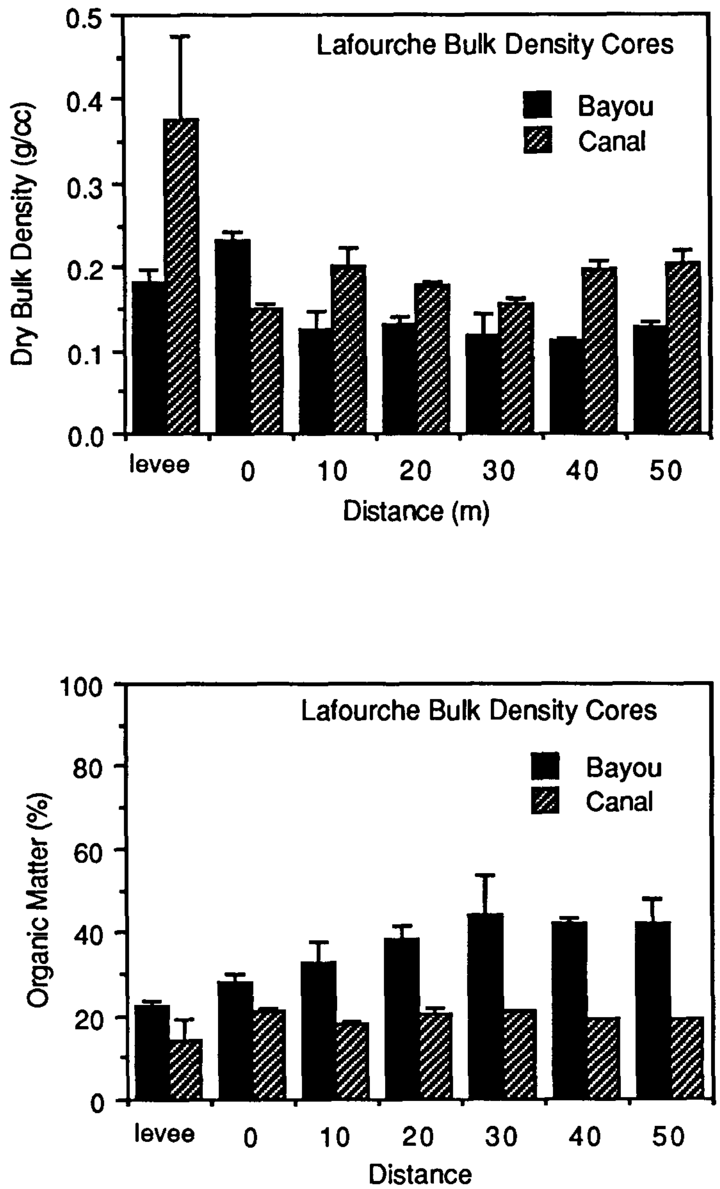
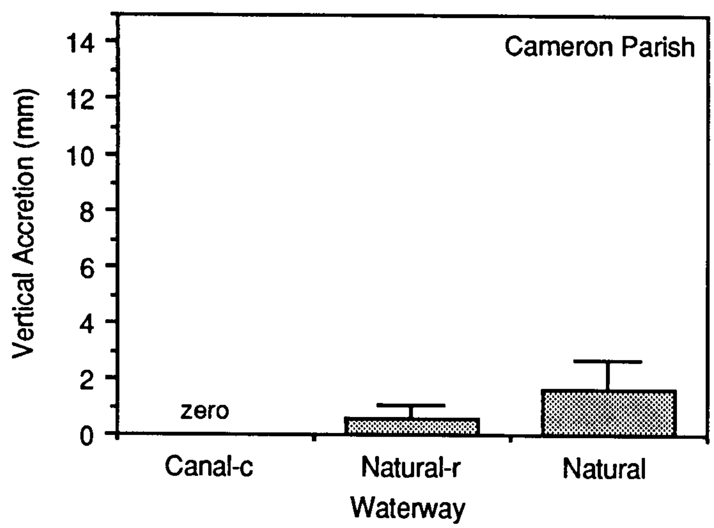


Figure 15-7. Dry bulk density (upper graph) and organic matter content (lower graph) of the marsh substrate along the canal and bayou transects in the Lafourche Parish salt marsh.



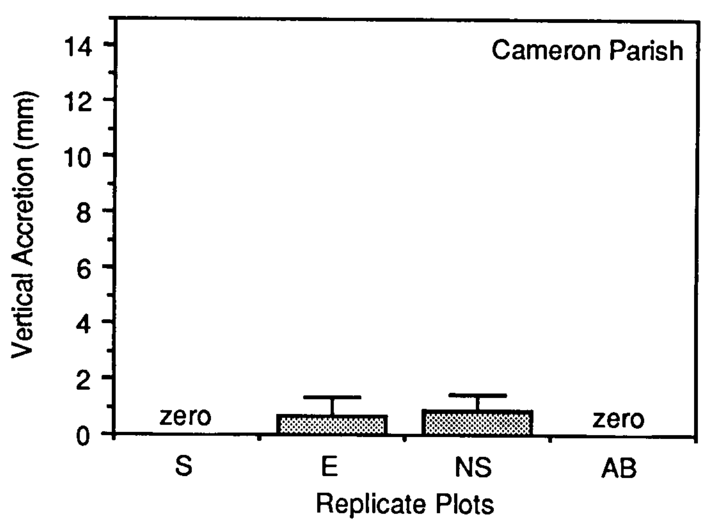
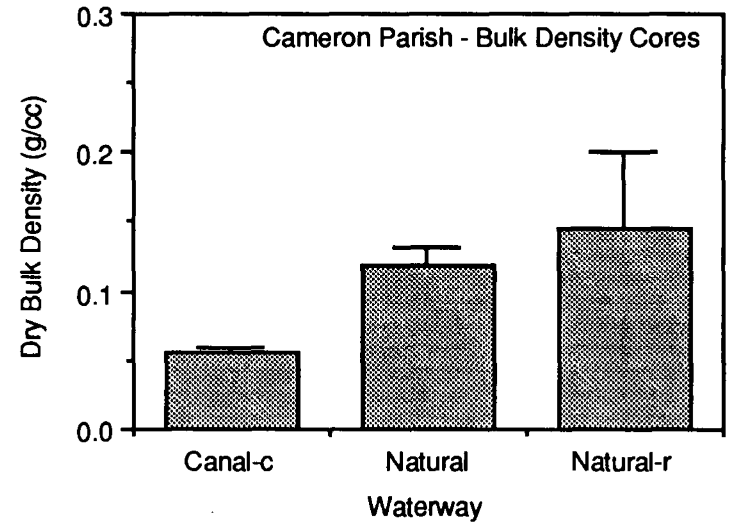


Figure 15-8. Sediment rates behind the continuous spoil bank levee in the hydrologically restricted area (Canal-c), the natural bayou in the hydrologically restricted area (natural-r), and the natural bayou in the non-hydrologically restricted area (natural) (upper graph) and in the replicate plots around the sampling platforms in the Cameron Parish brackish marsh (lower graph). Values presented are means ± 1 SE.



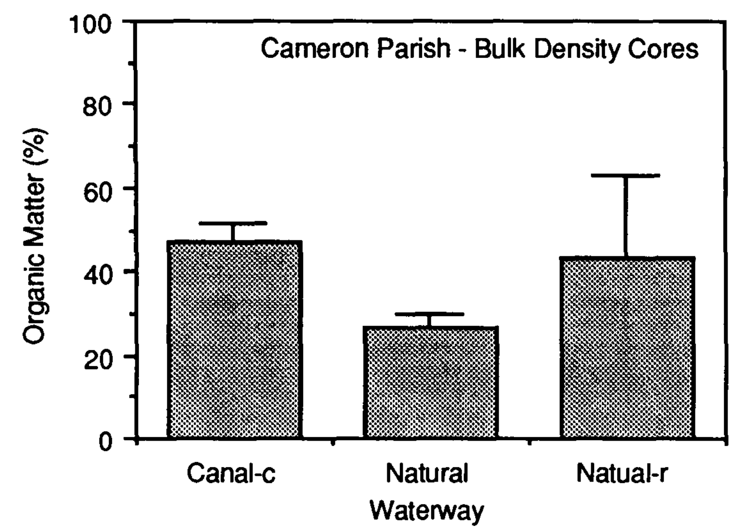


Figure 15-9. Bulk density estimates (upper graph) and organic matter content (lower graph) of the marsh substrate behind the continuous bank levee in the hydrologically restricted area (Canal-c), the natural bayou in the hydrologically restricted area (natural-r), and the natural bayou in the non-hydrologically restricted area (natural) in the Cameron Parish brackish marsh. Values presented are means ± 1 SE.

Twelve-month vertical accretion rates at the natural waterway in Sabine NWR cores collected in November were significantly greater than the 8-month estimates collected in July (1.1 versus 0.25 cm/yr, respectively). This suggests that the majority of marsh aggradation (mineral organic accumulation) occurred in the fall and, perhaps, winter on this coast. The November estimates of bulk density of the marsh surface and the percent of mineral and organic matter accumulation (0.15 g cm^{-3} and 77 to 23%, respectively) did not differ from July estimates.

Discussion

Recent (≤ 1 Year) Sedimentation Processes

The deposition and accumulation of waterborne mineral and organic matter on the surface of a marsh is strongly influenced by hydrology, microtopography, and biological agents acting in or on the substrate. All of these factors probably influenced sedimentation rates in the Lafourche and Cameron Parish study areas, although their separate effects were not partitioned and quantified.

Seasonal Influences. The fact that the 12-month (summer to summer) accretion rates in the Lafourche Parish salt marsh were more or less equal to the 6-month (summer to winter) rates indicates that sediment deposition on the marsh surface does not occur at a constant rate but varies seasonally. Baumann et al. (1984) estimated that approximately 75% of net accumulation at streamside and inland marshes occurred during fall and winter in Barataria Bay (Lafourche Parish), while in Fourleague Bay (Terrebonne Parish) 50% of net accumulation occurred during summer. Seasonal variability in sediment accumulation must be taken into account when analyzing recent sedimentation processes. For this reason, the 6- and 8-month results from Cameron Parish were not adjusted to an annual basis, and our interpretation of the 6- and 8-month data is based only on late winter, spring, and early summer deposition. Twelve-month samples from one of the three sites in Cameron Parish suggest a strong seasonal trend in marsh aggradation (fall and winter). It is noteworthy that such seasonal variability can only be detected by analysis of recent sedimentation events.

Influence of Locale. The longevity of marker horizons provides a means to estimate marsh surface stability at any given locale. Substrate surface stability is directly related to the intensity of hydrologic activity at the site. On a local scale, the disappearance of marker horizons in areas where vegetation is still present may indicate that the marker horizon was laid down in (1) a stable, but high energy, area of marsh where the marker clay was quickly mixed and redistributed throughout the soil column or (2) an unstable, eroding marsh area from which the marker clay was quickly removed through hydrologic action. There was no indication in any core without a distinct marker horizon that the marker clay was still present but mixed thoroughly from top to bottom throughout the core. Rather, it is more likely that the disappearance of the marker horizon is caused by very localized erosion of the marsh surface. Assuming a lack of physical disturbance by animals or climate, locally high rates of disappearance of the marker horizons may indicate areas of marsh that will soon begin to disintegrate. This idea can only be verified through long-term monitoring of these regions. A cursory review of aerial photographs of the marsh surface in the immediate vicinity of the marker horizon plots in Lafourche Parish reveals that all marsh areas are deteriorating, but no substantially higher degree of marsh break-up is occurring at those sites with low coring success.

Influence of Animals. The highly variable vertical accretion rates in the Lafourche salt marsh were the result of not only annual, seasonal, and diurnal variations in climatologic and hydrologic processes (Chapter 3) and man's influence on them but also animal activities. Mussels and fiddler crabs were present at all the salt marsh sites, streamside and inland. Crab burrows were within many of the plots, and the marker horizon was clearly visible inside the burrow entrance. Burrow construction results in relatively large amounts of substrate material being brought to the surface and deposited around the entrance (Katz, 1980). The presence of mussels in a salt marsh substrate can improve sediment trapping functions and therefore may have a direct positive effect on vertical accretion (Bertness, 1984). Also, birds (e.g., rails) were commonly observed using the marsh surface in the vicinity of the plots. Thus, a portion of the variability in recent aggradation processes observed at the Lafourche site is undoubtedly caused by biological agents acting on or in the substrate.

Influence of Canals on Sediment Distribution

Lafourche Parish - Waterway Analysis. Our evaluation of the dynamics of recent aggradation processes along the three waterway types did not reveal that canals significantly affect sediment accretion and accumulation (other than bulk density). However, even though vertical accretion and mineral accumulation behind the two canal types were not statistically significantly lower than that at the natural waterway, caution should be exercised in strictly interpreting the statistical analysis. The sample size was small and there was high sample variability which limited the statistical power of the test (i.e, the ability of the test to detect differences of 1.7-fold in vertical accretion and > 3 fold in mineral accumulation). Therefore, we reserve judgment on any statistical analysis with a probability of $P \leq 0.200$ because the magnitude of differences displayed in this analysis (differences in mean vertical accretion and mineral accumulation rates as great as 4 mm and 0.09 g cm^{-2} yr^{-1}, respectively (Table 15-2), with natural waterways always exhibiting higher rates) could be of long-term importance to the biological system, if, indeed, they are real.

Lafourche Parish - Distance Analysis. The analyses of recent aggradation along transects corroborate the findings of the ^{137}Cs and ^{210}Pb analysis (Chapter 14). Even though the rates of aggradation processes at the canal sites in the waterway analysis described above were not significantly different from those at the natural sites, the estimates of vertical accretion, bulk density, and organic matter content were all significantly higher at the pipeline compared with the bayou transect site. The reason for this different pattern may be found by examining the local hydrologic conditions at the pipeline transect site.

The transect was located south of the pipeline that bisects the marsh region in an east-west direction. Such alignment probably favors sediment deposition at this site during the passage of cold fronts. Southerly winds probably push the sediment-laden water up against the spoil bank, effectively retarding its movement and allowing suspended sediments to settle. In addition, the marsh at the canal-d transect has more ponds (open water) along it than the natural transect. The pipeline sampling sites at 10, 20, 30, 40, and 50 m were adjacent to open water ponds, most of which are connected to one another and eventually to the adjoining lakes on either side of the transect (see Figure 13-4). Each sampling site had a direct waterway connection to the larger waterbody and source of tidal waters. The only exception was the 0 m site, which was immediately adjacent to the spoil bank levee. This direct connection to flood waters, combined with the alignment of the canal, may account for the higher vertical accretion and mineral accumulation values at this site. (Note that plot LSCd2-e was located on the edge of a pond and received 21.9 mm of deposition before it disappeared.) Therefore, the canal-d transect site appears to be dominated by back-flooding from the lakes with only the 0 m station (low in mineral and high in organic matter) directly affected by the spoil bank levee. What role the canal may play in bringing sediment to the vicinity of the back marsh and how the canal influences aggradation processes in the marsh north of it is not known. On the other hand, the natural transect site appears to be dominated by typical overbank flooding processes because of its north-south alignment.

Cameron Parish. The hydrologically restricted northern end of the sampling site is characterized by low-velocity water movement and very shallow, silted-in water bodies. The closest direct hydrologic connection between the region and Lake Calcasieu is greater than 7 km. Therefore, the hydrology of this region is dominated by precipitation (C. Pettefer, Land Manager, Big Pasture, *personal communication*). In this hydrologically restricted setting, 9 of the 10 successful 6-month cores revealed no measurable sediment accumulation atop the marker clay. The marsh behind the pipeline spoil levee received no sediment accumulation (five successful cores all displayed no measurable accumulation), while the marsh behind the nearby natural streamside levee received some sediment (one out of five successful cores had measurable accumulation), although the amount was small. The natural waterway near the mouth of Grand Bayou at Sabine NWR (with a direct, unrestricted hydrologic connection to Lake Calcasieu) received significantly more material (two successful cores, both with measurable accumulation). It should be noted, however, that these plots experienced two more months of winter sedimentation events (November to July versus January to July).

The accretion rates estimated from the three successful cores were only 12 to 50% of the annual accretion rates estimated from 25 and 80 to 100 year cores (which equaled, at some sites, 0.8 cm yr^{-1}; Chapter 14). These zero and low accretion rates could be explained if sediment accumulation occurs episodically only a few times a year during the time period we did not sample, which appears to be the case for at least the natural waterway site at Sabine NWR. It still remains to be seen how much accumulation will occur in the hydrologically restricted sites. Even if the recent 6-month estimates were multiplied five-fold to account for seasonal variability and to adjust them to an annual basis, accretion at some sites would still not equal the rates of 25 or 100 years ago in the same locations. If the pattern displayed in the 6-month samples persists throughout an annual cycle, then the marsh in the recently hydrologically restricted region will eventually become flooded completely as water level rise continues to outpace sedimentation rates.

Conclusions

(1) Six to twelve month accretion rates (i.e, vertical increase) and density of recently accumulated mineral and organic matter vary seasonally and spatially across the surface of the marsh. Essentially all deposition in the saline marshes of southern Lafourche Parish is associated with the passage of winter cold fronts. On the other hand, most accumulation occurs in the late summer to fall in the brackish marshes along Grand Bayou on the southeastern shore of Lake Calcasieu in Cameron Parish. Spatially, accretion and accumulation vary widely over short distances across the marsh surface in both brackish and saline marsh systems.

(2) The influence of canals and their associated spoil banks on recent accretion and accumulation events varies and depends on the environmental setting. Vertical accretion rates 50 m behind canals were, in most cases, lower but rarely significantly different from rates behind natural waterways in the Lafourche Parish salt marsh. Bulk density was significantly lower behind canals with discontinuous spoil banks compared to natural

waterways. However, the continuity of the spoil bank did not influence vertical accretion. In addition, analysis of our coring success data and the distance analysis in Lafourche Parish suggests that the alignment of long OCS pipeline canals bisecting an entire marsh region perpendicular to the hydrologic gradient exerts a strong influence on accretionary processes.

(3) Mineral and organic matter accumulation (i.e., density of matter) differ in relative importance to the land-building process. Mineral sediment equalled or exceeded organic matter accumulation at nearly every site, regardless of the influences described above.

Chapter 16

MARSH ACCRETION, MINERAL SEDIMENT DEPOSITION AND ORGANIC MATTER ACCUMULATION: RARE EARTH STABLE TRACER TECHNIQUE

by

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This investigation uses a novel method of marking and measuring new marsh sediment layers in such a way that markers laid today will be unambiguous in future months and years. Stable tracers (rare earth elements) were placed both in the water and on the flooded soils of designated plots in the marshes of three parishes in south Louisiana (Figure 1-1). As the sediment in marker-treated water settles to the bottom of the marsh, the new sediment has the marker intimately incorporated into it, while the marker on the mud surface is adsorbed directly to the mud. Subsequent layers of sediment are free of the tracer. Since the elements picked for the tracer are not radioactive, play no role in plant or animal nutrition, and are insoluble at pH levels found in marshlands, the marker will be immobile and long-lasting in the sediment soil profile.

The analytical method used to detect the marker element in the sediment is instrumental neutron activation analysis (INAA). The INAA technique first gained notoriety in the 1960s when strands of Napoleon Bonaparte's hair were analyzed and found to contain levels of arsenic consistent with chronic poisoning. The technique is based on neutron bombardment of elements that readily absorb the neutrons. These elements then become activated (radioactive). From the quality and quantity of gamma radiation emitted from the radioactive atoms produced, the original bombarded element can be both identified and quantified at levels of less than one ppm. Marker elements in the marsh sediment layer have been carefully chosen so that they absorb neutrons and emit easily identifiable gamma rays, while existing at low tracer concentrations. Tracers that meet these criteria are found among the rare-earth elements.

Yearly accretion rates of marsh sediment have been estimated historically by well-established methods of determining depths of ^{210}Pb (Armentano and Woodwell 1975) and ^{137}Cs soil horizon markers (Chapter 14; Delaune et al. 1978, 1983; Simpson et al., 1976). Very recent sediment accumulation has been estimated by establishing visible layers of brick dust (Richard, 1978), clay (Chapter 15), or glitter in the sediment profile and measuring the sediment as it accumulates above the visible marker (Baumann, 1980).

Although the long-term markers of ^{210}Pb and ^{137}Cs are widespread and already in place ("free"), collection and analysis of samples are expensive and time consuming. The soil columns, too, become compacted as the sediment accumulates. Estimates from the data assume uniform sediment accumulation over 100 years for ^{210}Pb and nearly 25 years for ^{137}Cs analyses. Some of the problems associated with a visible marker technique include: 1) the marker may be more dense than the medium being measured; 2) a large quantity of the marker may be necessary for the marker to be discernible; 3) hydrology and life forms in the area may be changed; 4) the marker may be lost in freshwater systems; and 5) accurate location of marked areas may not be possible in the future. Incorporating a biologically nonessential radioactive element would be an ideal method to permanently mark a layer of sediment, but large-scale use of long-lived radiotracers in the environment is precluded from areas where human food and water supplies may become contaminated.

Since the recently accelerated industrial use of the marsh influences contemporary accretion rates, it is mandatory that decision makers and marshland managers have at their disposal data on the very recent sedimentation activity in marshes that are candidates for restoration programs. This research establishes a method by which current accurate data on yearly accretion rates can be obtained using INAA to detect artificial soil horizon markers incorporated in newly formed sediments.

Materials and Methods

Choice of Marker

The stable tracer marker must be biologically nonessential, in a relatively inert chemical form, low background levels, and relatively inexpensive. The isotope of the stable tracer must have a reasonable natural isotopic abundance and a high thermal neutron capture cross-section. The neutron-induced (activated) radionuclide must have an adequately intense and energetic gamma ray emission for ease of quantitative and qualitative analysis as well as possess a half-life sufficiently long for gamma ray analysis, post neutron bombardment (Knaus and Curry, 1979). Certain rare earth elements possess the above characteristics and are ideal for large-scale environmental studies involving field applications where radiotracers would be prohibited (Knaus and El-Fawaris, 1981). The rare earth elements selected for this research were dysprosium (Dy) and samarium (Sm).

Application of Marker

Kilogram quantities at 95% purity of individual rare earths were purchased in soluble nitrate form from Research Chemicals, NUCOR Corporation, Phoenix, Arizona. Measured amounts of slightly acidified Dy-Sm nitrate mixtures were diluted with natural water of the marsh, then applied to marsh vegetation (e.g., *Typha* spp. and *Spartina* spp.), mud surfaces, and water surfaces at the experimental sites by a CO_2-driven spray apparatus typically used for herbicide and insecticide applications. Knowing the area covered by the spray and the concentration of the spray, a minimum of 100 μg of the metal of each of the tracers was applied to each square centimeter of marsh area. The sensitivity of the INAA technique for Dy and Sm in environmental matrices was 0.10 μg per 0.1 g (wet wt) of sample. Upon contact of the water-diluted tracer spray with the natural waters of the marsh, an increase in pH caused a milky-white precipitate to form which extended through the water column and onto the top surface of the sediment.

Sampling of Marker

Recent sediment accumulations are often a loose, flocculent material that is nearly impossible to sample if the stratigraphy of the sample is to be preserved. To overcome this problem, a cryogenic coring device has been developed (Knaus,1986) that freezes the core *in situ*. In this study, the frozen core was extracted from the sediment, placed on dry ice in the field, and taken to the laboratory for sectioning. The sequence of procedures leading to numerical results from the frozen cores was a lengthy one (Figures 16-1 and 16-2). In the laboratory, the frozen core was prepared for sectioning by scraping off loose, angular debris projecting from the core to produce a smooth vertical surface of frozen mud and organic detritus. This scraping step assures that the samples taken from the core are not contaminated with rare earths from handling and packaging of the core and, especially, the scraping eliminates possible contamination from melt water that occasionally flows from the ice above the water-sediment interface onto the core.

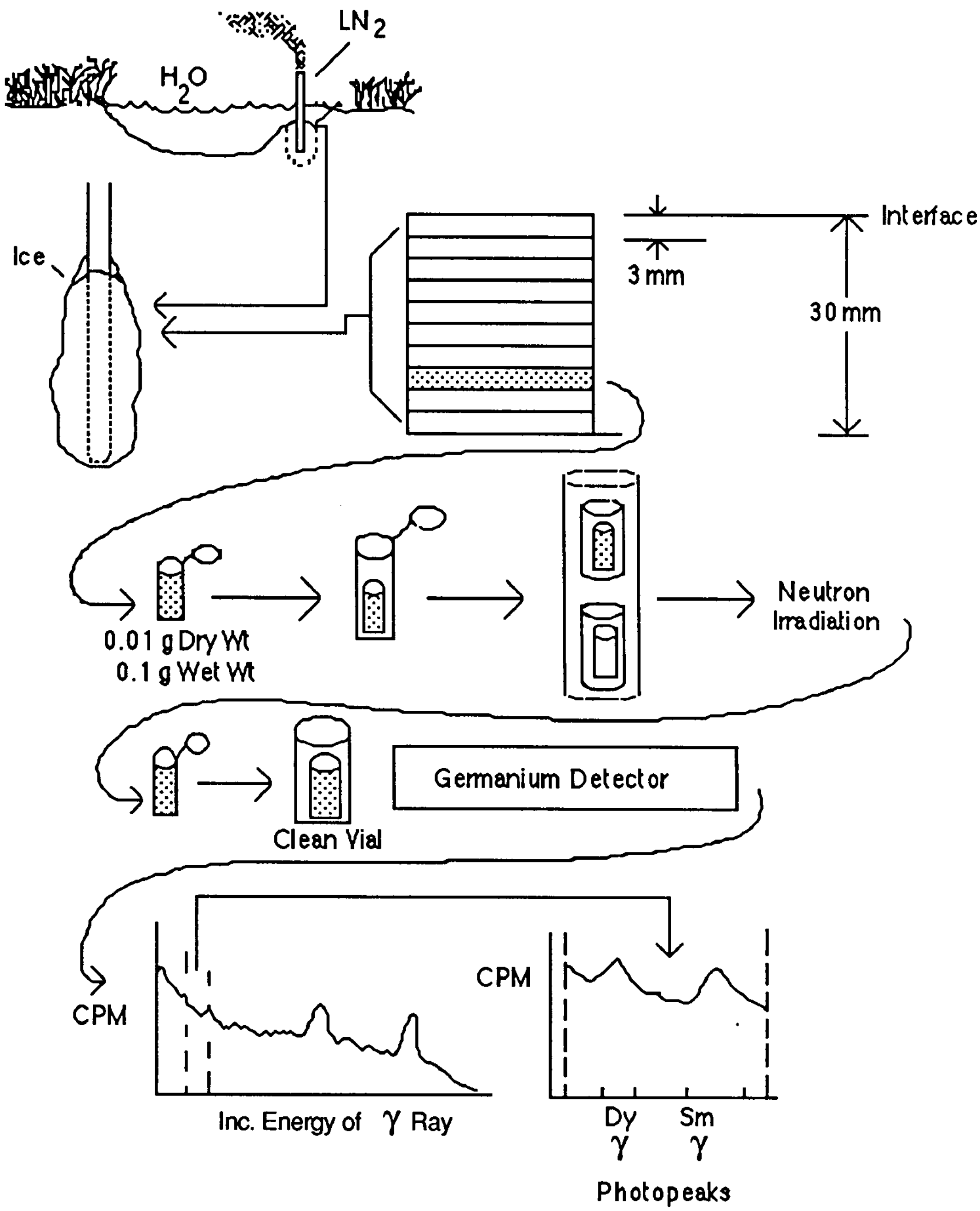

Figure 16-1. A schematic diagram of the procurement of samples from a marsh site and the preparation of samples for INAA. Part of the frozen core was sectioned into ten 3-mm samples. Each sample was triply encapsulated for neutron irradiation and placed in a clean vial for gamma ray analysis. Photopeaks for ^{165}Dy and ^{153}Sm were quantified in net cmp (corrected for decay and Compton background) and compared to standards.

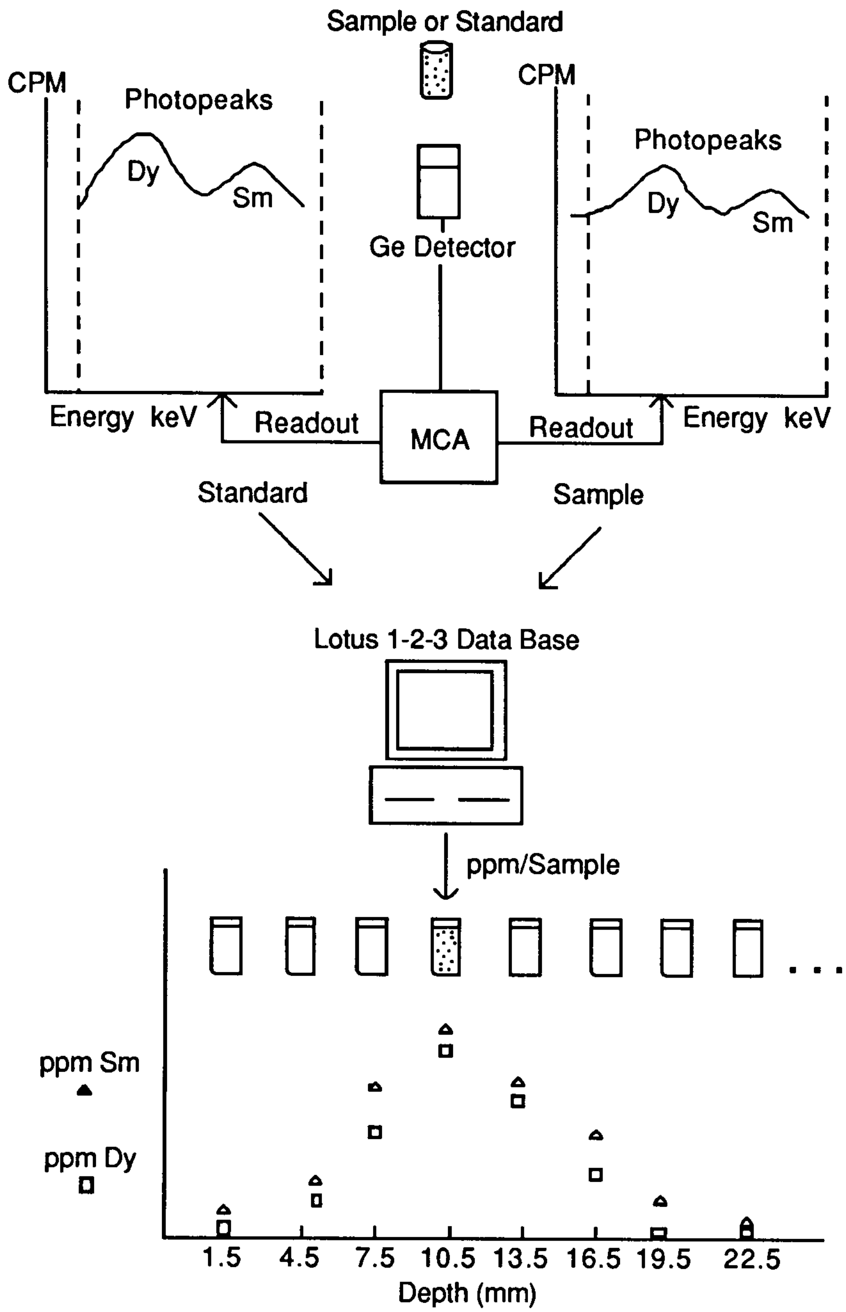

Figure 16-2. A typical read-out of one of a series of spectra, together with a standard, is shown. A graph of the entire spectral series (ppm vs depth) leads to a depth determination of the horizon marker of 10.5 mm in this example. The curve is classified as a Type II A (see Figure 16-3).

The prepared core was sectioned serially beginning at the water-sediment interface along its length. In this study, where only 6-month and 1-year accumulations were to be determined, the core was sectioned in 3-mm intervals (5-mm interval for one-year Terrebonne Parish cores) for 10 sections. After the 10 3-mm sections were collected, a careful measurement was made to assure that the sum of the 10 sections was, indeed, equal to a total of 30 mm. Because of the possibility of contamination from the copper coring device, no 3-mm sections were taken close to the surface of the coring device. The resolution of the sectioning can be no greater than 3 mm. Therefore, all data are reported for depths intermediate in each section, e.g., the 9- to 12-mm section datum is plotted at a depth of 10.5 mm (Figure 16-2).

For wet weights, the 3-mm serial sections were weighed in pre-weighed 2/27th-dram (0.2 cc) polyethylene vials (polyvials). The samples, typically 0.1 g (wet wt), were placed in a drying oven at 60 C overnight and re-weighed to obtain dry weights.

Sample Preparation and Neutron Irradiation

The dried samples were heat sealed in the polyvials and placed within 2/5th-dram (1.5 cc) polyvials. These, in turn, were heat sealed and two of the 2/5th-dram polyvials were encapsulated in one 2-dram (7 cc) polyvial for neutron irradiation (Figure 16-1). Approximately 10% of the samples, each, were taken or sent to the Oregon State University Radiation Center TRIGA Reactor Facility and the Los Alamos National Laboratory's Omega West Reactor Facility; 80% of the samples were either taken in person or mailed to the Texas A & M University Nuclear Science Reactor Facility for INAA.

At each reactor site, samples and standards were irradiated with a neutron flux equivalent to 10^{13} neutron cm^{-2} sec^{-1} for 2 minutes via the pneumatic transfer system. Analysis for the 0.0947 MeV gamma ray of the 2.33-h ^{165}Dy was performed within 5 hours, post neutron bombardment. Analysis of the same sample for the 0.103 MeV gamma ray of the 46.7-h ^{153}Sm was performed 48 to 100 hours, post bombardment. The combination of the two tracers is an internal verification of the data and also overcomes the possible interference with 165 Dy detection of the 15-h ^{24}Na in samples high in sodium content. If the Compton continuum of the 2.754 and 1.369 MeV ^{24}Na peaks masks the short-lived gamma ray peak of ^{165}Dy, the gamma ray peak of long-lived ^{153}Sm will grow relative to the ^{24}Na gamma ray peak after multiple half-lives of ^{24}Na pass. Each batch of irradiated marsh samples was accompanied by standards of known Sm, Dy, and Na content.

Preparation of Irradiation Standards

Standard preparation and interpretation are integral parts of core processing. A primary standard was prepared from nitrate salts of the rare earths, Dy $(NO_3)_3 \cdot 5H_20$ (Stock #-Nucor Corp.-Dy-N-3-010) and Sm $(NO_3)_3 \cdot 6H_20$ (Stock #-Nucor Corp.-Sm-N-026). Each of these salts in the crystalline form was ground with a mortar and pestle to a uniform particle size. Quantities of this preparation were placed in a $CaCl_2$ desiccator, and the system was allowed to come to equilibrium for seven days. After this, 0.2699 g of the Dy salt and 0.2956 g of the Sm salt were placed in the same 1000-ml volumetric flask. Five ml of technical grade 16N HCl was added and the flask was brought up to volume with tap water. The acidic environment overcomes the tendency of Dy and Sm to form hydroxide and carbonate precipitates. The resulting mixed primary standard is 100 ppm Dy and Sm, ± 10%. (At the time of standard preparation, four samples of 1.5 g of each of the Dy and Sm salts were placed in crucibles for ignition and subsequent gravimetric analysis is to be completed at a later date.) The primary standard was used throughout the present study. In this way, every separate data set is comparable to all others. Therefore, even though the absolute values of the standards have not been determined, the relative comparison of data sets was precise.

For the preparation of the secondary standards for INAA, 1-, 2-, 10-, and 20-ml aliquots of the primary standard were placed in 100-ml volumetric flasks. The secondary standards were brought to volume with tap water after adding 1 ml of 16N HCl. An aliquot of 0.1 ml of each of the four secondary standards was pipetted into four 2/27th-dram polyvials to make 0.1, 0.2, 1.0, and 2.0 μg standards. A reagent blank was prepared along with the standards in the same manner. Standards and reagent blanks were dried along with the core samples and encapsulated for neutron irradiation as described in the "Sample Preparation" section of this paper.

Data Reduction

Spectral data were received from the reactor facility on a computer printout. There were two separate printouts for each sample vial representing gamma ray emissions detected from ^{165}Dy and from ^{153}Sm. By visual inspection two separate 10- to 12-channel portions of the 4000-channel spectrum were partitioned for analysis for the Dy and Sm photopeaks. These sections of the data represent the characteristic photopeaks of the tracers, which are usually Gaussian in nature (Figure 16-2).

Once the photopeak has been chosen, the data were loaded on a Lotus 1-2-3 spreadsheet. The data base was then subjected to a number of calculations in which the net photopeak area for both Dy and Sm was determined (Covell, 1959), decay corrected, and reported as net cpm. This process was applied to both standards and samples. Once the cpm per μg of Dy and Sm in the series of standards had been determined, a least squares regression analysis was carried out, with μg Dy or Sm as the independent variable and net cpm as the dependent variable. The regression analysis on the standards included a linear, logarithmic, exponential, and power curve fit. In nearly every case, the power curve gave the best correlation coefficient and the best goodness of fit. The power curve fit was in the form

$$Y = BoX^{B1} \qquad \text{Eqn 16.1}$$

where Y is cpm and X is μg of Dy or Sm.

In this generalized form, if B_1 approaches 1, then the relationship between μg matter and cpm is a linear one. If B_1 departs significantly from 1, then a non-linear relation is implied (Steel and Torrie, 1980). Usually, the relationship is significantly non-linear. From this regression equation, cpm per sample was converted to μg of Dy or Sm, and this value was ultimately converted to ppm of Dy or Sm in the soil sample.

A 99% confidence interval for background levels of Dy and Sm in marsh soil samples had previously been determined using a Student's *t*-distribution (Steel and Torrie, 1980, p. 65). This background interval was subtracted from all values determined by the method described above. A background-corrected value in ppm of Dy and Sm was calculated and plotted on the Y-axis against depth on the X-axis (Figure 16-2).

Core Data Analysis

The distribution of the stable tracer horizon markers varied greatly among the three parish study sites and within each site. A classification scheme for the various types of distribution of the marker was developed for this study and is depicted in Figure 16-3. A Type I distribution is the ideal finding, where the location of the marker is located unequivocally in a single 3-mm section of the core; 18% of the cores sampled in this study were categorized as a Type I class. The mode and mean are identical, and the center of the 3-mm section containing the marker is reported for the depth.

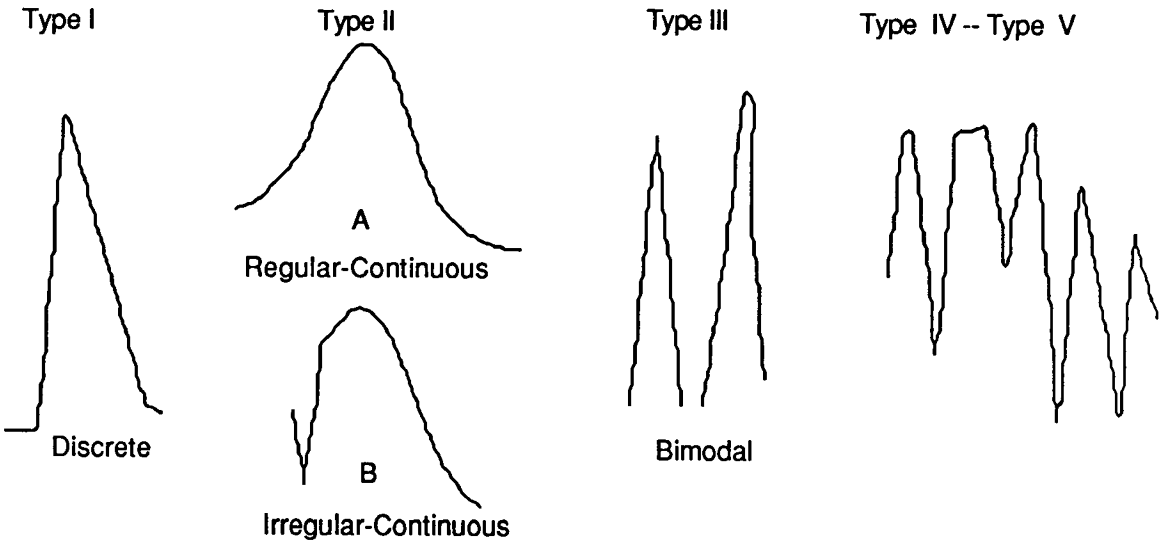

Figure 16-3. The distribution of stable tracer marker in the sediment is categorized in five classifications. The diagrams represent amount of tracer on imaginary Y-axes, plotted against depth of the tracer distribution along imaginary X-axes. Type I is a discrete distribution. Type II is divided into two subcategories: (A) regular-continuous, which approaches a Gaussian distribution; and, (B) irregular-continuous, which also follows a Gaussian distribution, but has one (or two) data point(s) that fall below the Gaussian curve. Type III displays two distinct distributions of nearly equal magnitude. Type IV classification is assigned to irregular distributions with values well above background values, indicating mixing of the sediment. Type V is as Type IV, but the values are comparable to background values indicating that no marker was found.

A Type II distribution is divided into two subclasses: Type II A and Type II B. Type II A approximates a Gaussian distribution and can be fitted to a binomial with the mean reported as the depth of the marker. Type II A distributions accounted for 42% of the cores sampled. Type II B is like Type II A but has one or two data points that do not easily fit the binomial. The depths of Type II B cores are reported as idealized binomial means of the distributions unless visual inspection determines an obviously skewed distribution of the marker away from the mathematically determined mean. Type II B distributions accounted for 9% of the cores. Type III accounted for 15% of the cores sampled, depicting two distinct peaks where two or more data points between the peaks registered at or near background levels for the stable tracers. Type IV accounted for 8% of the cores,depicting a random distribution of tracer above background levels. Type V is like Type IV, but the levels of tracer were within the background range; 9% of the cores were Type V.

All of the above core data were subjected to three separate reduction schemes reported in Table 16-1* . The first analysis was a binomial best fit model (Steel and Torrie, 1980, p. 524). A binomial distribution was fitted to the individual data points using a computer program which iterates to a best fit using a chi-square goodness of fit criterion. The mean depth was calculated from the idealized binomial curve generated from the data points. If the distribution was Type III, the binomial was fit to each distribution separately.

The second data reduction technique was a visual inspection and selection of the depth at which the "apparent mode" occurs. The apparent mode is that point on a graph (tracer abundance versus depth of tracer) that would be the highest data point on the graph if no "marker wash-in" occurred. Marker wash-in is a phenomenon that has been observed by others working with trace substances in the soil profile, particularly ^{137}Cs (Miller and Heit, 1986). It seems likely that additional marker sorbed to the sprayed vegetation may wash onto the surface of treated experimental plots well after the original marker had been placed. In 85% of the cores analyzed, the apparent mode and the real mode were the same.

The third reduction was visual inspection and selection of the "break-point." The break-point is the depth at which the sharpest drop-off of marker concentration occurs. In 50% of the cores analyzed, the depth at which the apparent mode and the break-point occur was identical.

Number and Location of Stable Tracer Corings

A total of 57 frozen cores was analyzed for this project; 33 cores were of 6-month accumulations, and 21 cores were of 1-year accumulations; 3 cores were analyzed for background values of Dy and Sm. These cores were divided among the three marsh locations as follows: 17 6-month cores and 10 1-year cores from the Lafourche Parish sites; 10 6-month cores and 12 1-year cores from the Terrebonne Parish sites; and 6 6-month cores from the Cameron Parish sites. Each core has 10 individual soil sections for a total of 570 samples undergoing INAA. Of the total, 3 background cores near the Lafourche site (30 samples) were analyzed. Standards accompanying the core samples accounted for 44 more irradiations, for a grand total of 644 samples analyzed.

Figure 13-4 shows the location in Lafourche Parish of sites sprayed with the tracers, Dy and Sm. The seven sites, LSC-0 m, 15 m, 35 m, LSN 1, 2, 3, and LSCd-2 were sampled 6 months after tracer application and the 14 sites LSN-0 m, 10 m, 30 m, 50 m, LSC d, LSC 2 W, E, LSC-0 m, 20 m, 30 m, 40 m, LSN-1, 2, and 3 were sampled 1 year after marker application (see Table 16-1).

Figure 13-6 shows the locations in Terrebonne Parish of the stable tracer sites. The 10 sites TFC-0 m through TFN-50 m were sampled at 6 months post-tracer application and the 12 sites TFN-1-0 m through TFN-50 m were sampled 1 year after tracer application (Table 16-1). Sites TFC-2 and TFIr-2 were sprayed but not sampled.

Figures 13-7 and 13-8 show the location in Cameron Parish of sites sprayed with the stable tracer. The 6 sites CBNr-1 through CBN-2-S (Table 16-1) were sampled 6 months post-marker application.

* All raw data from which Tables 16-1, 16-2, and 16-3 and Figures 16-4 through 16-8 were derived will appear in Appendix A, "Rare-earth Soil Horizon Markers to Determine the Short-term Accretion in Louisiana Marshes," master's thesis by Daniel L. Van Gent, Nuclear Science Center, Louisiana State University, May 1988.

Table 16-1. The results from 6-month and 1-year cores taken at the three sites of the study areas are listed. The text contains explanations of the headings used in the columns. Figures 1-1, 13-4, 13-6, 13-7, and 13-8 show locations of the sites.

		Binom. Depth (mm)	Mode	Break Pt.	Class	Avg. Modes
		Dy Sm	Dy Sm	Dy Sm	Dy Sm	Dy & Sm (mm)
Cameron						
7/8/87						(mm/6mo)
8-mo	CBN 1	11.8	7.5	7.5	IIA	5.6
		11.2	7.5	7.5	IIA	
8-mo	CBN-2-S	10.0	13.5	13.5	IIA	10.1
		9.7	13.5	13.5	IIA	
6-mo	CBNr 1	7.3	7.5	7.5	I	7.5
		7.3	7.5	7.5	I	
6-mo	CBNr 2	9.0	1.5	1.5	IIA	1.5
		8.7	1.5	1.5	IIA	
6-mo	CBC 1	9.1	1.5	1.5	IIA	1.5
		8.9	1.5	1.5	IIA	
6.mo	CBC 2	10.1	4.5	4.5	I	4.5
		9.8	4.5	4.5	I	
Lafourche						
12/15/86						(mm/6mo)
6-mo	LSC TRANS.(0-M)	N/A	1.5	1.5	I	1.5
		N/A	1.5	1.5	I	
6-mo	LSC TRAN (BRG) 15M	-	-	-	V	-
		-	-	-	V	
6-mo	LSC.TRANS (STRM) 35	-	-	-	V	-
		-	-	-	V	
6-mo	LSN 1	5.7	7.5	7.5	I	7.5
		6.2	7.5	7.5	I	
6-mo	LSN 2	8.5	10.5	10.5	IA	10.5
		8.6	10.5	10.5	IIB	
6-mo	LSN 3	8.2	4.5	13.5	IIA	4.5
		8.6	4.5	4.5	IIA	
6-mo	LSCd 2	11.0	7.5	10.5	IIA	7.5
		11.3	7.5	7.5	IIA	
5/13/87 & 6/11/87						
6-mo	LSN TRAN (0-M) JU	9.1	7.5	10.5	IIA	7.5
		9.1	7.5	10.5	IIA	
6-mo	LSN TRAN (10-M) JU	10.7	13.5	10.5	IIA	13.5
		10.7	13.5	10.5	IIA	
6-mo	LSN TRAN (30-M) JU	9.2	7.5	7.5	IIA	6.0
		10.3	4.5	4.5	IIB	
6-mo	LSN TRAN (50-M) JU	10.6	7.5	7.5	IIA	7.5
		7.0	7.5	7.5	IIA	
6-mo	LSCd 3 (50-M) JU	9.8	7.5	10.5	IIA	7.5
		9.8	7.5	10.5	IIA	
6-mo	LSC 2 WEST MAY	7.5	7.5	-	IIA	7.5
		7.5	7.5	-	IIA	
6-mo	LSC 2 EAST MAY	4.6	4.5	4.5	I	4.5
		6.2	4.5	4.5	IIA	
						(mm/yr)
1-yr	LSC TRN (0-M)M CAN	N/A	16.5	16.5	III	16.5
		N/A	16.5	16.5	III	
1-yr	LSC TRAN (20-M) JU	9.9	16.5	10.5	IIA	16.5
		9.9	16.5	10.5	IIA	
1-yr	LSC TRAN (30-M) JU	9.2	16.5	7.5	IIB	15.0
		9.2	13.5	7.5	IIB	

cont'd on next page

Table 16-1 cont'd.

		Binom. Depth (mm)	Mode	Break Pt.	Class	Avg. Modes
		Dy Sm	Dy Sm	Dy Sm	Dy Sm	Dy & Sm (mm)
1-yr	LSC TRAN (50-M) JU	10.3	16.5	10.5	I	16.5
		10.3	16.5	10.5	I	
1-yr	LSN 1 MAY	11.0	13.5	7.5	IIA	13.5
		10.4	13.5	7.5	IIA	
1-yr	LSN 2 MAY	5.9	7.5	7.5	IIA	7.5
		5.7	7.5	4.0	IIA	
1-yr	LSN 3 MAY	7.8	4.5	4.5	IIA	4.5
		7.9	4.5	4.5	IIA	
Terrebonne						
2/14/87						(mm/6mo)
6-mo	TFN TRANS (0-M)C	18.4	19.5	19.5	IIB	18.0
		16.5	16.5	16.5	IIA	
6-mo	TFN TRANS (25-M)	-	-	-	V	-
		-	-	-	V	
6-mo	TFN TRANS (50-M)	20.7	19.5	23.5	IIA	19.5
		-	-	-	V	
6-mo	TFC TRANS (0-M)	14.5	13.5	13.5	I	13.5
		17.2	13.5	13.5	I	
6-mo	TFC TRANS (25-M)	5.9	4.5	4.5	I	4.5
		-	-	-	V	
6-mo	TFC TRANS.(50-M)	11.1	10.5	17.5	I	9.0
		7.0	7.5	10.5	IIB	
6-mo	TFN-2-50M WEST	15.9	25.5	25.5	IIA	25.5
		15.9	25.5	25.5	IIA	
6-mo	TFN-1-50M EAST	11.5	13.5	13.5	IIA	13.5
		11.6	13.5	13.5	IIA	
6-mo	TFN-1-50M EST CO	-	-	-	N/A	-
		-	-	-	N/A	
6-mo	TFIr-1-50M TYPHA	-	19.5	-	III	19.5
		-	-	-	V	
6/24/87						(mm/yr)
1-yr	TFN TRANS (0-M)	N/A	42.5	7.5	III	42.5
		N/A	42.5	7.5	III	
1-yr	TFN TRANS (25-M)	-	-	-	V	32.5
		32.5	32.5	-	IIA	
1-yr	TFN TRANS (50-M)	N/A	47.5	37.5	I	47.5
		N/A	47.5	22.5	III	
1-yr	TFC TRANS (0-M)	23.4	22.5	22.5	IIA	19.2
		5.1	N/A	2.5	IIA	
1-yr	TFC TRANS (25-M)	-	-	-	V	32.5
		N/A	32.5	-	III	
1-yr	TFC TRANS (50-M)	27.2	27.5	27.5	IIA	24.1
		21.0	-	-	IV	
1-yr	TFN-1-50M EAST	N/A	32.5	22.5	I	35.0
		N/A	37.5	37.5	III	
1-yr	TFN-2-50M WEST	24.9	27.5	27.5	I	25.5
		26.5	N/A	-	IV	
Terrebonne						
6/24/87						
1-yr	TFI-1-50M	22.3	7.5	17.5	IV	22.1
		22.0	22.5	22.5	IV	
1-yr	TFI-2-50M	-	-	-	-	-
		-	-	-	-	-
1-yr	TFC 1-50M	21.1	7.5	27.5	IIA	20.0
		19.6	7.5	27.5	IIA	
1-yr	TFLr-1-50M TYPHA	27.8	12.5	12.5	IIA	26.0
		24.4	42.5	37.5	IIB	

Table 16-2. For comparative purposes, one-year cores taken from four sites that are not MMS study areas are listed. The text contains explanations of the headings used in the columns.

	Binom. Depth (mm)	Mode	Break Pt.	Class	Avg. Modes
	Dy Sm	Dy Sm	Dy Sm	Dy Sm	Dy & Sm (mm)
Cocodrie					
11/8/86					
COC1 (PLNK-7)	12.1	10.5	10.5	IIA	10.5
	11.1	4.5	4.5	IIA	
COC2 (NEXT PLNK-7)	14.6	13.5	16.5	IIA	10.5
	14.2	10.5	10.5	IIA	
CO3ELBPLK-BY-PLK	15.5	7.5	19.5	IIA	7.5
	15.8	4.5	13.5	IIA	
Fourchon					
2/20/86					
FOU REP1 (BKGD)	-	-	-	V	
	-	-	-	V	
FOU REP2. (BKGD	-	-	-	V	
	-	-	-	V	
FOU REP3 (BKGD)	-	-	-	V	
	-	-	-	V	
6/21/86					
FOU1(SAM 1)	-	-	-	IV	
	-	-	-	IV	
FOU2(SAM 2)	20.0	16.5	16.5	IIA	7.5
	13.1	7.5	7.5	IIA	
FOU3(SAM 3)	16.4	1.5	13.5	IIA	7.5
	12.8	7.5	7.5	IIA	
5/13/87					
FOU87EDG COL-2	13.7	10.5	13.5	IIA	9.0
	15.8	7.5	7.5	IIA	
FOU87H2O DEEP PR	15.3	7.5	7.5	IIA	7.5
	12.8	7.5	7.5	IIA	
FOU 87ST.PROBE	-	-	-	V	
	-	-	-	V	
Lac des Allemands					
8/8/86					
LAC1 S-3 (LS-1)	-	-	-	III	32.5
	27.3	32.5	32.5	III	
LAC1B S-3(LS-1)	-	-	-	III	
	60.1	52.5	72.5	III	
LAC N/F S-3(LS-2)	-	-	-	IV	57.5
	-	57.5	57.5	IIB	
LAC2 S-2(1)	22.3	17.5	17.5	III	62.5
	22.3	17.5	17.5	III	
LAC2B S-2(1)	56.7	62.5	62.5		
	56.7	62.5	62.5		
LAC 3A S-2(2)	28.7	37.5	37.5	III	57.5
	28.7	37.5	37.5	III	
LAC 3B S-2(2)	50.4	52.5	52.5		
	50.4	62.5	62.5		
LAC4A S-3(3)RS	6.5	32.5	32.5	III	42.5
	26.5	-	-	V	
LAC4B S-3(RS)	49.1	32.5	32.5	III	
	49.1	-	-	IV	

cont'd on next page

Table 16-2 cont'd.

	Binom. Depth (mm)	Mode	Break Pt.	Class	Avg. Modes
	Dy Sm	Dy Sm	Dy Sm	Dy Sm	Dy & Sm (mm)
Rockefeller					
6/17/86					
ROC1 SU. CANAL S-	28.5	22.5	22.5	IIA	22.5
	24.8	2.5	37.5	V	
ROC2 SU. CANAL S-	17.1	12.5	12.5	IIA	12.5
	27.8	-	-	IV	
ROC3 PRICE LAK S	30.1	-	-	IV	22.5
	22.6	22.5	22.5	IIA	
ROC4 PRICE LAK S	16.9	2.5	2.5	III	17.5
	20.7	17.5	17.5	IIA	
ROC5 PRICE LAK S	10.1	12.5	12.5	IIA	12.5
	26.8	2.5	2.5	IIB	

Results and Discussion

A generalized coherent and consistent data reduction scheme was applied to all of the data generated from the spreadsheet data base (Tables 16-1 and 16-2). This scheme consisted of three analyses: 1) a careful visual inspection of the graphs of marker versus depth for modes and break points; 2) the application of an objective mathematical model (binomial best fit); and 3) a final determination of the most probable position of the marker in the sediment (Table 16-1). It must be stated that the binomial best fit may not be the most appropriate mathematical model, but it provides a common basis for quantitative comparison of other, more sophisticated models as more cores are analyzed in future years.

Lafourche Parish Study Site

The two main purposes of taking and analyzing cores from the Lafourche sites (Figures 16-4 through 16-8) were to supplement data obtained by the established technique using clay for a horizon marker (Chapter 15; Baumann et al. 1984) and to establish novel techniques for collecting and measuring a chemical soil horizon marker from very recent accumulations of sediment. The experimental design did not include replicate cores to establish core variance because of the time and expense involved. [However, an estimate of within-plot variance has been determined using transect cores as replicates. This seems reasonable, since no trend of accretion with distance have been observed (also in Chapter 15)]

The data in Figure 16-4 show the 6-month accretion rates at the natural transect site. As described earlier in this section, 0 m, in these and all subsequent figures, is a point behind the natural levee or spoil bank (into the marsh, away from the waterway) at which the marsh is barely distinguishable from the major marshland habitat in the vicinity. Since no replicate cores were taken at any of the distances, it is not possible to determine any significant trend with distance from the bank into the marsh. However, visual inspection of Figure 16-4 reveals little difference with distance, so it is likely that within the 6-month time-frame of these data, there is no trend in the deposition rates with distance into the marsh. However, if the four data in Figure 16-4 are treated as replications within one plot, then it is possible to obtain a mean with a variance in that plot (within-plot-variance). The data in Figure 16-4 yield a mean of 8.6 mm per 6 months with a standard deviation of 3.3 mm.

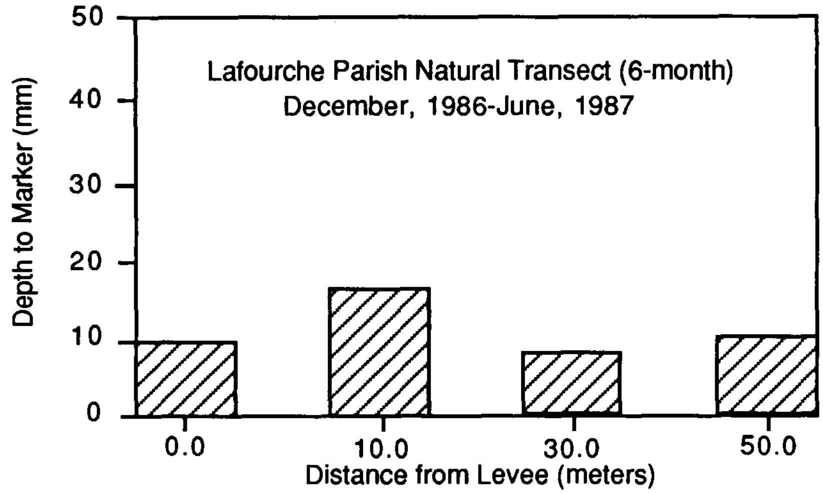

Figure 16-4. Six-month accretion (mm) was measured at four distances (m) along a boardwalk at the Lafourche saltwater transect, perpendicular to a natural waterway. The bars represent average values of the modes for the highest concentrations of the rare-earth markers with depth.

The data in Figure 16-5 show the 1-year accretion rates at the canal transect site. Since no replicate cores were taken at any of the distances, it is not possible to determine any significant trend with distance from the bank into the marsh. However, visual inspection of Figure 16-5 reveals little difference with distance, so it is likely that within the 1-year time-frame of these data, there is no trend in the deposition rates with distance into the marsh. When the four data in Figure 16-5 are treated as replications within one plot, a mean with a standard deviation in that plot is obtained: 16.1 ± 0.75 mm per year.

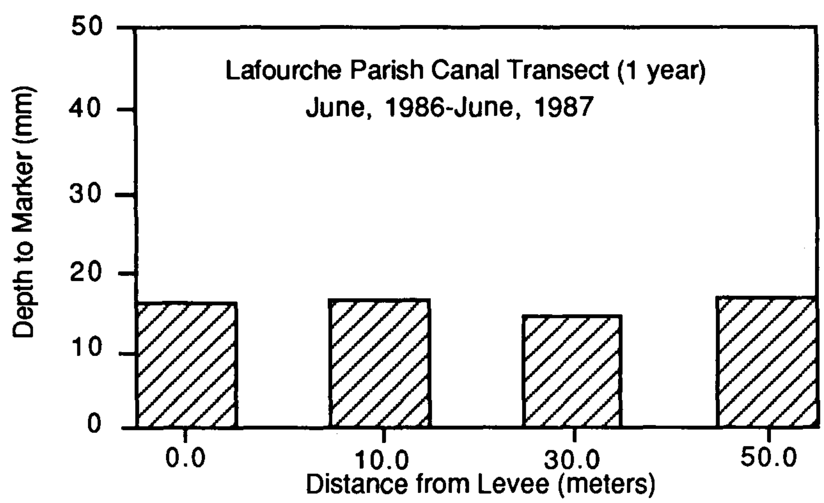

Figure 16-5. One-year accretion rates (mm) were measured at four distances along a boardwalk at the Lafourche saltwater transect, perpendicular to a canal waterway. The bars represent average values of the modes for the highest concentrations of the rare-earth markers with depth.

The alternative method of measuring short-term accretion rates using a clay marker (Chapter 15; Figures 16-6, 16-7, and 16-8) corroborates the findings in Figures 16-4 and 16-5. No significant differences with distance into the marsh were found along the same two (natural and canal) transects depicted in Figures 16-4 and 16-5 at the 6-month and 1-year accretion rate intervals. Because the natural and canal transect sites (Figures 16-4 and 16-5) were marked with rare-earth tracer at different times (6 months apart), the natural transect and the canal transect are not comparable at this point in time. However, this time difference between the two sites should become insignificant after sufficient time passes, and the sites may become comparable.

The data in Figure 16-6 show the 6-month accretion rates at 4 different sites located 50 m into the marsh. Three of the sites were off natural waterways, while the fourth site was off a canal. No replicates were made at any of the sites. Combining the 3 natural 50 m sites yields a mean value of 7.5 ± 2.5 mm during the 6-month, June to December, interval.

The data in Figure 16-7 show the 6-month accretion rates at 4 different sites located 50 m into the marsh. Three of the sites were off canal waterways, while the fourth site was off a natural waterway. No replicates were made at any of the sites. Combining the 3 canal 50-m sites yields a mean of 6.7 ± 1.5 mm during the 6-month, December to June, interval. Note that the LSN 50 m transect datum from Figure 16-4 is also recorded in Figure 16-7 for comparison. The data from Figures 16-6 and 16-7 cannot be compared directly because their respective 6-month accumulation time-frames are different. It is interesting to note that there appear to be no differences among the sites and no differences between treatments and dates.

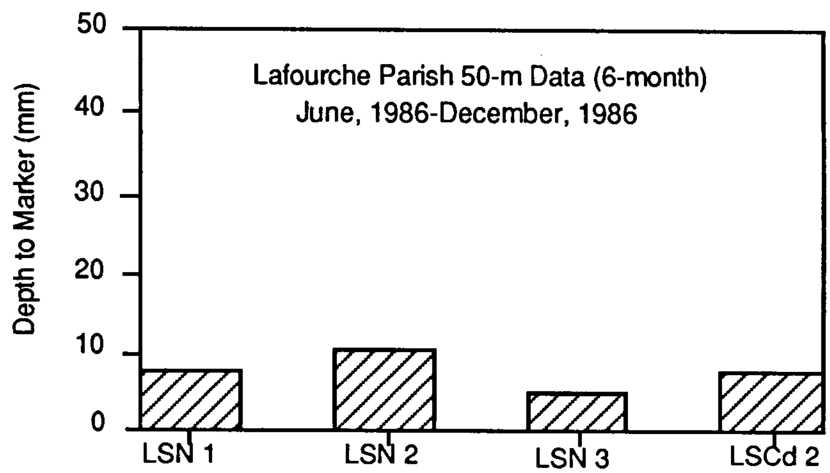

Figure 16-6. Six-month accretion from June, 1986 to December, 1987 was measured at four different locations, 50 m into the salt marsh, perpendicular to natural and canal waterways at the Lafourche Parish site. The bars represent average values of the modes for the highest concentrations of the rare earth markers with depth.

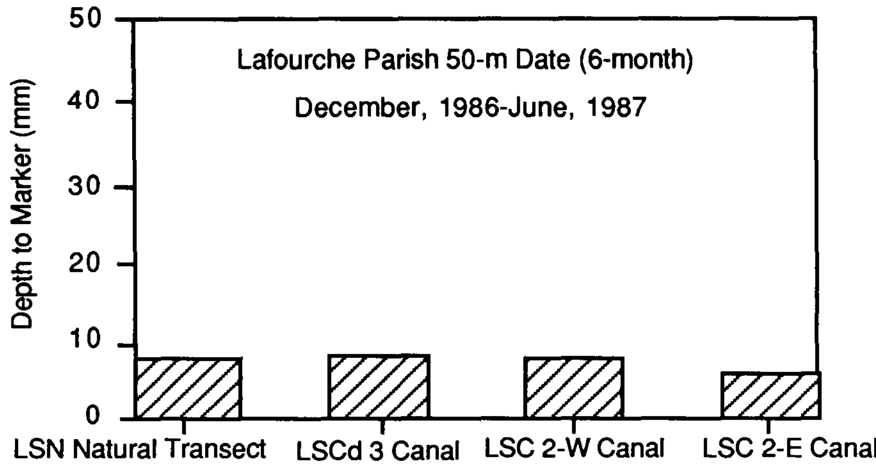

Figure 16-7. Six-month accretion from December, 1986 to June, 1987 was measured at four different locations, 50 m into the salt marsh, perpendicular to three canal and one natural waterway at the Lafourche Parish site. The bars represent average values of the modes for the highest concentrations of the rare earth markers with depth.

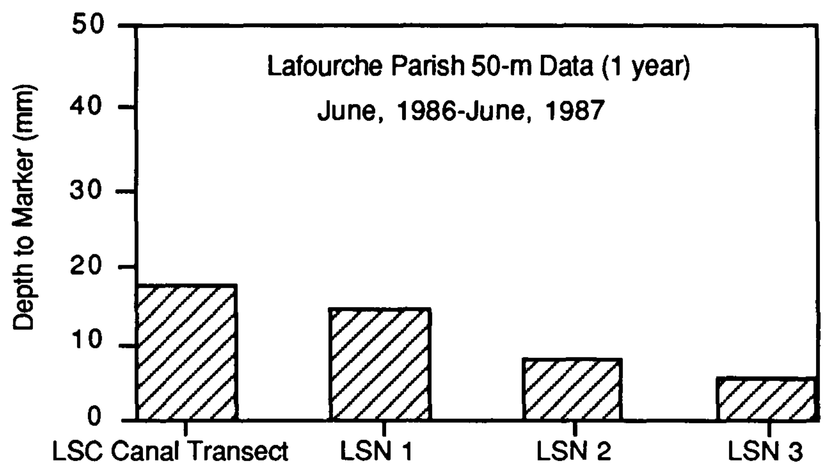

Figure 16-8. One-year accretion rates were measured at four different locations, 50 m into the salt marsh, perpendicular to three natural and one canal waterway at the Lafourche Parish site. The bars represent average values of the modes for the highest concentrations of the rare-earth markers with depth.

If the assumption is made that there is no difference of accretion with distance into the marsh, the data in Figures 16-4 and 16-7 (cores were collected at the same time) can be tested using an unpaired t-test. Testing for a significant difference between the natural transect treatment ($n = 4$) of Figure 16-4 and the canal treatment of Figure 16-7 reveals that there is no significant difference between treatments at 6 months ($P = 0.41$).

The data in Figure 16-8 show 1-year accretion rates at 50 m into the marsh at 4 sites. Note that the LSC 50 m transect datum from Figure 16-5 is also recorded in Figure 16-8 for comparison. Visual inspection suggests that there might be a difference between the one canal site and the three natural sites, but this cannot be stated definitely when there is no estimate of variance at each individual site. By comparing the 1-year accretion data in Figures 16-5 and 16-8, an unpaired *t*-test can be applied testing only for significant differences between the canal transect treatment ($n = 4$) of Figure 16-5 and the natural treatment ($n = 3$) of Figure 16-8. The canal treatment showed a significantly higher accretion rate versus the natural stream treatment at 1 year ($P = 0.02$). This is in agreement with the clay technique in the same study area and at the same time (Chapter 15).

The grand mean at the Lafourche study site (Table 16-3) for all 6-month data (11 cores, Table 16-1) is 0.76 cm with a range of 0.45-1.35 cm with a deviation of 0.26 cm. The grand mean for all 1-year data (seven cores, Table 16-1) is 1.29 cm with a range of 0.45-1.65 cm with a standard deviation of 0.49 cm. These means are probably representative of the entire Lafourche study area.

The Lafourche 6-month accretion data in Table 16-3 are subdivided into the two different six-month periods and are presented within parentheses. The four cores ($n = 4$) taken in December, 1986, were from different sites than the seven cores taken in June, 1987. The datum of 0.76 cm per year (datum not in parentheses) is obtained by averaging all 11 cores even though they are from two different seasons. The two 6-month data sets for two different seasons do, however, show similar accretion rates.

Terrebonne Parish Study Site

Clay soil horizon markers have not proved successful in freshwater habitats (Baumann et al., 1984), such as those found at the Terrebonne Parish sites examined in this report. Therefore, the sole method of determining 6-month and 1-year accretion rates was the rare earth marker technique.

The 6-month accretion rates at the natural and the canal transect sites in Terrebonne Parish are reported together in Figure 16-9 for ease of comparison. Because no replicate cores were taken at any of the distances, it is not possible to determine any significant trend of accretion with distance from the bank into the marsh. No value is reported for the 25-m natural transect core (TFN 25 m); this finding does not imply zero accretion but, rather, that the marker was not found. (Figure 16-9 reveals no differences of accretion with distance into the marsh after a 6-month or a 1-year interval for both natural and canal sites. It is likely that there is no trend of accretion at either of the sites within the 6-month time frame of these data.) The natural and canal transect sites were established and sampled at the same time, but because the TFN 25 m datum is missing, a 2-site statistical comparison cannot be done. A visual comparison suggests that slightly more accretion is occurring at the natural transect site. This suggestion of greater accretion at the natural sites over the canal site is given more credence when one observes that the 0 m and 50 m natural site data both increased by 60% during the second 6-month interval. Taking 40% of the 1-year accretion value of TFN 25 m (3.25 cm from Figure 16-10) yields a hypothetical 6-month accretion value of 1.35 cm. If one were to speculate that the missing 6-month TFN 25 m datum would be on the order of 1.35 cm, that value exceeds 2 of the 6-month canal transect values (TFC 25 and 50 m) and is equal to the third canal value (TFC 0 m).

Table 16-3. Location and recent sediment accretion rates for marshland habitats (natural and disturbed) in Louisiana as determined by rare earth soil horizon markers and instrumental neutron activation analysis. Rockefeller, Lac Des Allemands, Fourchon, and Cocodrie data are not MMS data; they are included for comparison. MMS data are below the line drawn above Leeville.

Geographic Location	Water Salinity[a] (collection interval)	Date, Marker placement (mo/yr)	6-mo samples (n)	6-mo accretion rate (cm)	6-mo range (cm)	Yearly samples (n)	Yearly accretion rate (cm)	Yearly range (cm)
Rockefeller Wildlife Refuge								
a) Price Lake	Fresh, perched above salt	2/84	—		—	3	0.75	0.54-0.96
b) Superior Canal[b]	Fresh to brackish	2/84	—	—	—	2	0.75	0.54-0.96
Lac Des Allemands	Fresh	5/84				11	2.67	1.20-4.70
	(5/84-3/85)		—	—	—	(6)	(2.8)	(1.20-4.70)
	(5/84-6/86)		—	—	—	(5)	(2.52)[c]	(1.45-2.80)
Cocodrie	Salt	4/86	3	0.95	0.75-1.05	–	–	–
Lafourche Parish								
a) Fourchon	Salt	2/86	2	1.12	1.12	2	0.66	0.60-0.72
b) Leeville	Salt	6/86	11	0.76	0.45-1.35	7	1.29	0.60-1.72
	(6/86-12/86)[d]		(4)	(0.75)	(0.45-1.05)	–	–	–
	(12/86-6/87)[d]		(7)	(0.77)	(0.45-1.35)	–	–	–
Terrebonne Parish	Fresh	6/86	8	1.42	0.45-2.60	11	2.97	1.95-4.75
Cameron Parish	Brackish	1/87	6	0.51	0.15-1.01	–	–	–
	(11/87-7/87)[e]		(4)	(0.38)	(0.15-0.75)	–	–	–
	(11/87-7/87)[e]		(2)	(0.79)	(0.56-1.01)	–	–	–

[a] Marsh type based on dominant vegetation type (Cowardin et al. 1979)

[b] Earthen barrier breached to salt water, February 1985

[c] The 2-yr accumulation was 5.04 cm; the 1-year accretion is obtained by dividing 5.04 cm by 2 years

[d] The11 6-month samples analyzed from Lafourche Parish were comprised of 4 samples, collected 12/86 and 7 samples collected 6/87, which were from 7 different sites than the first 4 sites

[e] Four 6-month samples from Plum Bayou; two 8-month samples from Grand Bayou; values are pro-rated to 6-month accumulation period.

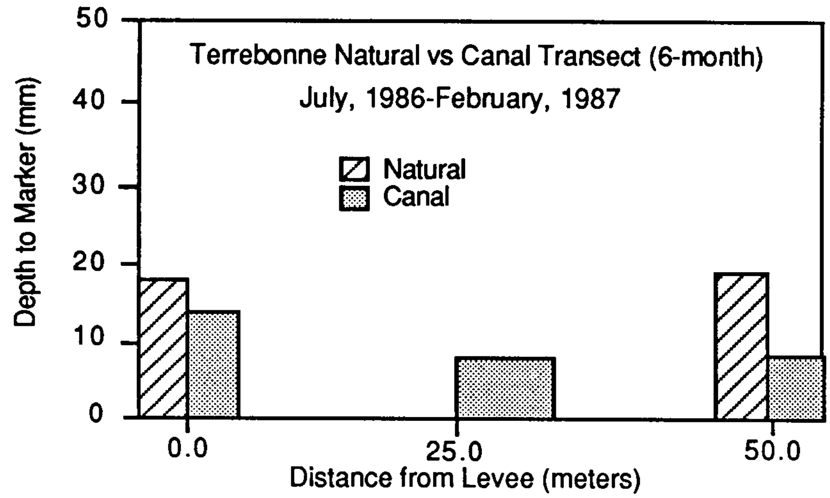

Figure 16-9. Six-month accretion (mm) was measured at three distances (m) along two transects in freshwater habitats in Terrebonne Parish. One transect was located off a natural waterway and the other off a canal. No marker was found in the natural 25-mm core.

The data in Figure 16-10 show 1-year accretion rates at the natural and canal transects. Because no replicate cores were taken at any of the distances, it is not possible to determine any significant trend of accretion with distance from the bank into the marsh. A visual inspection of Figure 16-10 reveals little difference in accretion with distance into the marsh. It is likely that there is no trend of accretion at either of the sites within the 1-year time frame of these results. Because the natural and canal 1-year transect sites were established and sampled at the same time, they can be compared as replicates for the two habitats. The overall accretion rate of the three combined natural transect sites is significantly higher ($P = 0.01$) than the three combined canal transect sites; the averages being 4.08 cm per year at the natural transect, versus 2.53 cm per year at the canal transect.

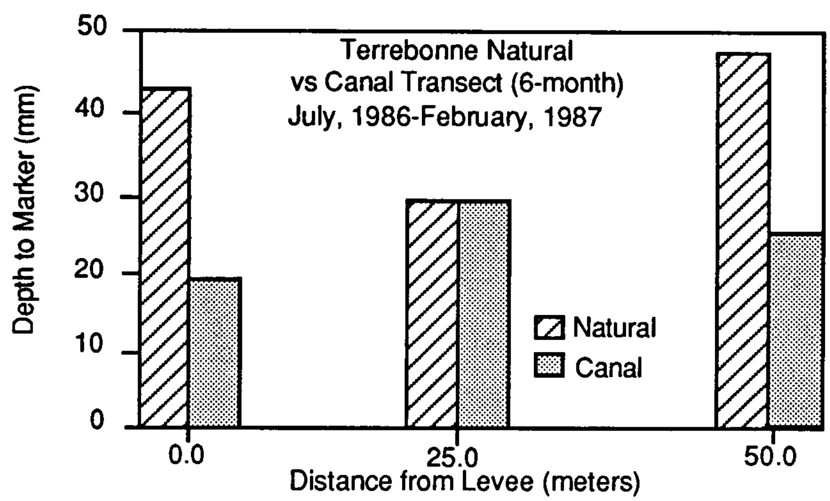

Figure 16-10. One-year accretion rates (mm) were measured at three distances (m) along the same transects depicted in Figure 16-9.

The data in Figure 16-11 show the 6-month and 1-year accretion rates at eight locations at natural bayous and canal sites, 50 m into the marsh. The data from TFC and TFN 50 m transect sites (for both 6 months and 1 year) are taken from Figure 16-9 and 16-10. No statistically significant trend is found among sites or among natural, canal, and impoundment-influenced sites. The marker at the impoundment-influenced site, TFI-2, 1-year, was not found in the core, and the lack of a value reported in Figure 16-11 does not imply a zero accretion rate. No 6-month cores were taken at the TFC-1 and TFI-2 sites.

A comparison of the 50-m data at 6-months and the 50-m data at 1 year (Figure 16-11) can be made. An unpaired *t*-test (6-month cores, n = 5; 1-year cores, n = 7) showed a significantly higher accretion level at 1 year versus 6 months (P = 0.05). A t-test was also performed to determine whether one-year accretion rates were greater than six-month adjusted to one-year accretion rates. There was no significant differences (P = 0.35). The overall accretion rates of the combined 50-m data show an average accretion rate of 1.8 cm for 6 months and an average rate of 2.86 cm for 1 year (excluding the TFI-2 site). (The difference in the average value of 1.8 and 2.86 cm accretion rates calculated from the data in Figure 16-11 and in the accretion values reported in Table 16-3 (1.42 and 2.97) results from the inclusion of the 0 and 25 m cores in the calculations of Table 16-3).

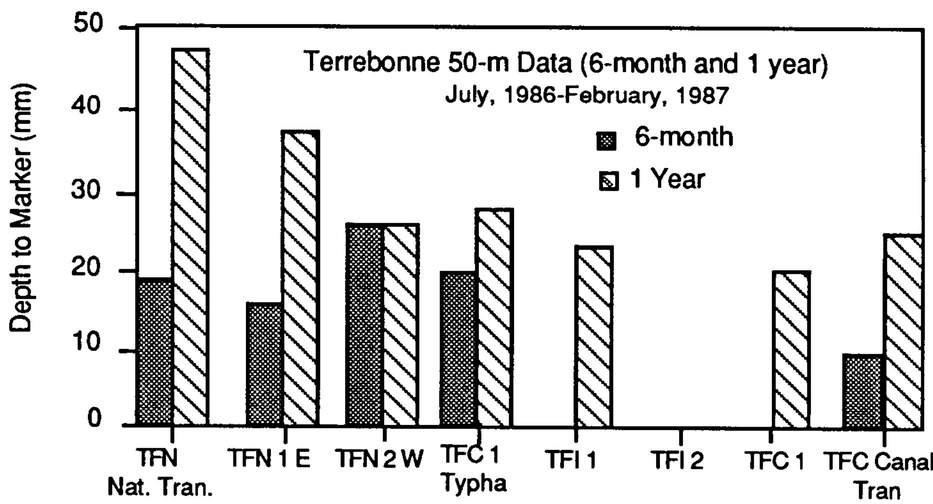

Figure 16-11. Six-month and one-year accretions (mm) were measured at eight different locations, 50 m into the marsh, perpendicular to freshwater waterways in Terrebonne Parish. The six-month and one-year values for the natural transect (TFN-Nat. Tran.) and canal transect (TFC Canal Tran.) at 50 m were taken from Figures 16-9 and 16-10, respectively.

The grand mean at the Terrebonne study site (Table 16-3) for all the 6-month data (eight cores) is 1.53 cm with a range of 0.45-2.6 cm and a standard deviation of 0.66 cm. The grand mean for all the 1-year data (11 cores) is 2.97 cm with a range of 1.95-4.75 cm with a standard error of 0.92 cm. These means are probably representative of the Terrebonne study area.

Cameron Parish Study Site

The data in Figure 16-12 show accretion rates at natural and canal sites at 6 locations, 50 m into the marsh. Two of the sites (CBN 1 and 2) were marked with rare earths eight months prior to sampling. The data in Table 16-3 for these two sites have been pro-rated for ease of comparison with the four 6-month sites (CBNr-1 and -2, and CBC 1 and 2). The mean value for the six cores is 0.51 cm accretion per 6 months. If the 6 cores are divided into two general collection area data, shown in parentheses in Table 16-3, the two cores taken near the unchanneled Grand Bayou (CBN 1 and 2) show accretion rate of 0.79 cm per 6 months, while the 4 cores, taken 5 km north of Grand Bayou in a recently leveed area along the eastern shore of Calcasieu Lake, show 0.38 cm accretion in 6 months. Evidently, a fraction of the source of sediment had been cut off from the marsh by the construction of the levee system 2 years prior to coring.

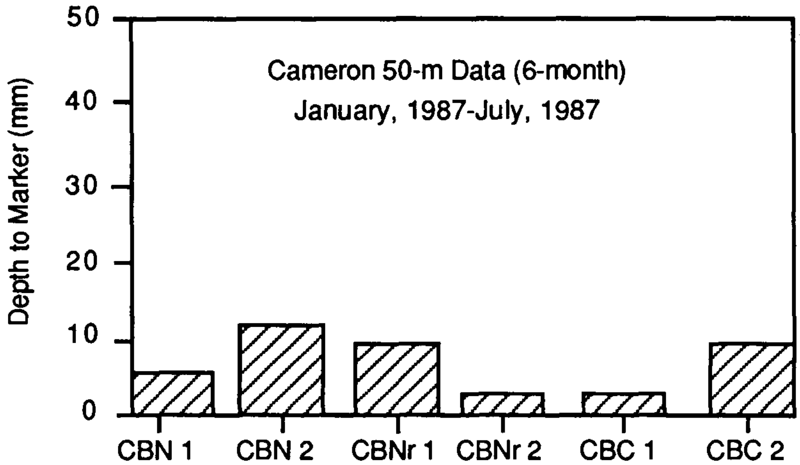

Figure 16-12. Six-month accretion (mm) was measured at six different locations, 50 m into the marsh, perpendicular to fresh waterways in Cameron Parish. The bars represent modes for the highest concentrations of the rare-earth marker with depth.

Conclusion

Results demonstrate that the rare-earth stable tracer method can be utilized to measure short-term wetland accretion rates. The stable tracer technique yielded useful results in 90% of the cores taken. Most importantly, the technique is useful in freshwater systems as seen in the Terrebonne study area. This is apparently the first time short-term accretion rates have been determined successfully in a freshwater habitat. The reason for this success may be related to the sorption of stable tracers to organic surfaces, as well as inorganic surface components. Large organic components, such as roots, are less mobile than the inorganic components under conditions of intense water flow (Baumann, 1980). For instance, if the marker sorbs to live roots in the soil matrix, it is more likely to remain in place in the soil, while marked inorganic components and loose detrital material wash away. (The sorption of rare earths to living roots in flowing streams has been demonstrated (Knaus and Curry, 1979; Knaus, 1981; Knaus and El-Fawaris, 1981).) The data also show that the stable tracer is behaving in a manner similar to the surface particulate matter making up the vertical soil profile. Thus accretion can be estimated without a bias that could be introduced by other marker methods that interfere with natural accretion processes.

In the three study sites of this project, results show:

1. There was no effect on accretion rates with distance (50 m) into the marsh, whether or not the site was located behind a canal or natural site.

2. At the Lafourche study site, there was no significant difference between canal and natural transect sites after six months ($P = 0.41$). There was a significant difference between canal and natural treatments after one year with the canal sites showing the greater accretion rate ($P = 0.02$).

3. At the Terrebonne study site, the overall one-year accretion rates were significantly higher for the natural sites as compared to the canal sites ($P = 0.01$). As expected, it was also demonstrated that the 50-m sites showed significantly higher accretion levels at one year as compared to 6 months ($P = 0.05$).

Chapter 17

MARSH ACCRETION, MINERAL SEDIMENT DEPOSITION, AND ORGANIC MATTER ACCUMULATION ALONG MAN-MADE CANALS AND NATURAL WATERWAYS: SUMMARY OF THREE DATING TECHNIQUES

by

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A detailed explanation of each methodology and results is presented in Chapters 14, 15, and 16. The results obtained from the three dating techniques have been synthesized and are presented below. Also, the compatibility of the three techniques was evaluated when used together.

Synthesis of Results

The results of the three dating studies (Chapters 14, 15, and 16) have been combined in an effort to answer the questions identified in Chapter 13. A summary of the influence of canals on marsh aggradation processes is presented in Table 17-1. Caution should be used when applying these results beyond the sites investigated because of the small sample sizes employed and the large variances encountered in the data.

1. Do man-made canals/channels influence the distribution of sediment across the marsh surface?

Evaluation of recent, 25, and 80 to 100 year events (vertical accretion and density of mineral and organic matter) at selected sites along the coast revealed that the influence of canals on the rate of vertical marsh accretion across the marsh surface depended on the local environmental setting.

Analysis of the influence of natural streamside and canal spoil bank levees (Comparisons 1 and 2, Table 17-1) on accretion processes in the three locales, determined primarily from recent sediment deposition patterns, revealed that mean rates 50 m behind canals were, in most cases, lower but rarely significantly different from rates behind natural waterways. A major exception to this was found at the distance analysis sites in the Lafourche Parish salt marsh (Comparison 4, Table 17-1) when comparing the mean of all cores taken at the site. Long canals bisecting the Leeville marsh region on an east-west plane (i.e., perpendicular to the basin hydrologic gradient) appear to have an important impact on marsh aggradation (OCS pipeline transect site) and marsh surface stability (coring success north and south of Southwestern Louisiana Canal). Consequently, alignment of the pipeline canal and navigation channel in the Leeville salt marsh may influence the sediment exchange along a portion of the hydrologic gradient of the lower Barataria estuary.

Distance analysis in the salt, brackish, and fresh marsh locales indicated no effect of distance on vertical accretion and density of mineral and organic matter at either the bayou

Table 17-1. The influence of canals on vertical accretion (VA, cm yr^{-1}), bulk density (BD, g cm^{-3}), mineral accumulation (MA, g cm^{-2} yr^{-1}), and organic accumulation (OA, g cm^{-2} yr^{-1}), presented as the difference between means ("a +" indicates that rates behind the canal were higher than those behind the natural waterway while "a -" means they were lower).[a] See text for detailed explanation of variation in the data.

Comparison	Marsh Type	Sediment Marker	Impact of Canal VA	BD	MA	OA
1. Behind Natural Levee vs Continuous Spoil Levee (LSN vs LSC, CBN vs CBC, TFN vs TFC)	Salt (n=3)	Recent (C,S)	-0.33	-0.04	-0.07	-0.02
	Brackish (n=2)	Recent (C,S)[b] 137Cs	-0.2	-0.06	nd	nd
	Fresh (n=3)	Recent (S)	+0.01	-0.05*	0.0	-0.01
2. Behind Natural Levee vs Discontinuous Spoil Levee (LSN vs LSCd)	Salt (n=4)	Recent (C,S)	-0.4	-0.08	-0.09	-0.03
3. Behind Continuous Spoil vs Discontinuous Spoil Levee (LSC vs LSCd)	Salt (n=3)	Recent (C,S)	+0.06	+0.04	+0.02	+0.01
4. Distance Analysis Behind Natural Levee vs Pipeline Spoil Levee (0-10-20-30-40-50m) (LSN vs LSCd Transect, TFN vs TFC Transect)	Salt (n=1)	Recent (C,S)	+0.6*	+0.04*	+0.10*	+0.01
		137Cs	+0.21*	+0.05*	+0.09*	+0.01*
	Fresh (n=1)	Recent (S)	+0.09	-0.04	-0.03	0.0
		137Cs				

[a] All values are based on 12 month cores unless otherwise noted. Significant differences (at the 95% level) are indicated by an asterisk (*). In Comparison 3, the + symbol indicates that the mean rate behind the continuous spoil bank was higher than that behind the discontinuous spoil levee. Under marsh type, n = the number of canal and natural sites compared. Under sediment marker, recent sedimentation estimates were obtained from clay (C) and stable isotope (S) plots.

[b] These rates are based on six and eight month samples, and nd means no data available.

or canal sites. The natural site in the fresh marsh, however, exhibited the typical "edge effect" by having significantly higher bulk densities at the streamside plots. The "edge effect" was not apparent at the pipeline site.

2. To what extent does continuity of the spoil banks influence accretion rates?

The canal spoil levees were higher than the natural streamside levees because they were vegetated with upland (mesic) species. The influence of natural versus canal spoil levee on vertical accretion and sediment distribution in the back marsh is discussed above.

Evaluation of recent sedimentation events indicated that continuity of the spoil bank (Comparison 3, Table 17-1) had no influence on sediment deposition in the saline marsh at Lafourche Parish (0.60 versus 0.66 cm yr^{-1}, discontinuous versus continuous, respectively).

3. What is the relative importance of mineral and organic matter accumulation to the land building process and what influence do canals have on this ratio?

Organic matter accumulation rates estimated from recent and 25 to 100 year cores were essentially the same for the canal and bayou sites studied, with between 0.02 and 0.05 g cm^{-2} yr^{-1} being involved in accretionary processes. Mineral sediment deposition varied, depending on marsh site and presence of any streamside effect, and equalled or exceeded organic matter accumulation at nearly every site. Mineral sediment constitutes an increasing fraction of marsh solids nearer the coast where there is greater tidal exchange.

Comparison of Dating Techniques

Vertical accretion rates obtained by recent ($\leq$ 1 yr, clay marker and stable isotope), 25 yr (^{137}Cs), and 100 yr (^{210}Pb) dating techniques were similar. Accretion rates obtained from ^{210}Pb techniques were, in all cases, less than ^{137}Cs, suggesting either oxidation and compaction of surface peats or more rapid accretion in recent years. The coherence between techniques indicates that an evaluation of recent impacts to accretion processes is possible through the concurrent use of recent ($\leq$ 1 yr) and long-term (25 to 100 yr) dating techniques. For example, comparison of recent with 25- or 100-year rates at the same site can be used to evaluate the impact on sedimentation processes of management efforts implemented during the interim.

Chapter 18

SEDIMENTATION-SUBSIDENCE WORKING GROUP CONSENSUS AND SYNTHESIS OF FACTORS INFLUENCING THE MARSH SURFACE/WATER SURFACE RELATIONSHIP

by

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Consensus of Research Effort

It was the mandate of the Sedimentation-Subsidence Working Group to investigate the components and processes of relative water level rise (RWLR) and marsh surface aggradation and to evaluate man's influence on them. Based on this evaluation, we estimated the potential for economic development to influence wetland loss in the coastal zone. The following is a summary consensus statement of our findings.

Relative Water Level Rise

(1) Rates of water level rise are variable and dominated by natural processes. During the past 80 years at selected sites along the coast, the rate of RWLR has fluctuated apparently on a 20 to 30 year cycle but has not accelerated. A portion of the variability can be attributed to basin water level changes that significantly influence marsh water levels and have produced decade-long rises and lowerings of water levels at tide gauge stations by as much as 60 cm.

(2) In Louisiana's marshes, subsidence is the major component of relative water level rise. Fluid withdrawal and levee subsidence may be important locally, but geologic consolidation is the dominant factor (100 to 300% of eustatic sea level rise) behind RWLR along the coast.

(3) Consequently, subsidence usually exerts at least three times more influence on RWLR than seasonal or global variations in water level. Therefore, natural processes exert a greater influence over relative water level change than man-influenced processes. However, man's influence may be very important on a local scale, but, unlike natural forces, man's influences can be purposefully changed.

Aggradation of the Marsh Surface

(1) During the past 100 years, the supply of sediment delivered from the Mississippi River to the coastal marshes has decreased dramatically. This decrease in supply, coupled with the elimination of direct input to the marsh via overbank flooding in the 1930s (when

the levees on the River were completed), undoubtedly has influenced marsh aggradation rates. Approximately 3% of the suspended mineral matter presently confined within the levees would be delivered directly to the marshes via overbank flooding and crevassing on an annual basis if the levees did not prevent it.

(2) In this comparatively sediment-poor setting, an extensive network of canals and spoil banks has been constructed throughout the coastal zone. The influence of canals and spoil banks on sediment distribution across the marsh varies with the environmental setting and ranges from less than to greater than the influence of natural waterways. On a local scale, canals and spoil banks may cause a ±6 mm change in mean annual vertical accretion rate compared to natural waterways, depending on alignment, local hydrologic patterns, and sediment supply. These differences are attributable to the influence of the spoil bank on marsh surface hydrology.

(3) The availability of sediment appears to be a limiting factor in several regions of the coast. Investigations in the salt marshes of lower Barataria Bay indicate that aggradation rates and bulk densities in the vicinity of both natural and man-made waterways are at the minimum level required to sustain marsh growth. In some areas of the brackish marshes east of Lake Calcasieu, current accretion rates appear to be much lower than rates of 25 years ago. Sediment supply is becoming a critical factor in marsh growth and stability in this region.

Marsh Surface/Water Surface Relationship

(1) A review of annual rates of RWLR and marsh surface accretion rates reveals that water level may be rising faster than accretion rates, at least over short time periods (≤ 25 years) and can result in a surface disparity (rate of RWLR > rate of marsh surface accretion). Natural processes dominate the marsh surface/water surface relationship, but human activities do influence it.

(2) The potential for surface disparities increases in the vicinity of canals because the localized effects of canal construction and oil and gas development (levee influence on subsidence of the marsh surface, surface hydrology and fluid withdrawal) are additive with all regional influences.

(3) We cannot confirm that surface disparities existed at any of our study sites because estimates of water level and marsh surface elevation were not available to compare with measured accretion rates. Future studies should focus on this issue.

Synthesis of Marsh Surface/Water Surface Relationship

Understanding the relationship between changes in relative water level (RWLR; sea level rise + subsidence) and land building (vertical accretion and sediment distribution) processes in Louisiana's coastal plain is difficult because of the multitude of natural and man-influenced processes that directly and/or indirectly affect marsh surface stability. For example, the two components of relative water level rise (eustatic sea level rise and geologic subsidence) are influenced by man through combustion (i.e., the greenhouse effect) and withdrawal (in fluid form) of fossil fuels. It is important to understand not only the natural processes but also how man's activities influence them. This relationship may not be direct and simple.

This section presents a quantitative accounting of aggradation and subsidence processes in coastal Louisiana, which we developed as a means to evaluate man's influence on the marsh surface/water surface relationship. The results of this cataloging procedure were interpreted and applied toward answering the central question posed by the working group: *"What is the contribution of organic and inorganic sedimentation in counteracting land loss (coastal submergence) on a local and regional basis and how do man's activities (OCS and non-OCS) influence this process?"*

We began our accounting procedure by seeking a solution to the following relationship:

$$\begin{aligned} \textit{Marsh surface/water surface} \approx \textit{fluid withdrawal}(-) &+ \textit{geologic compaction}(-) \\ &+ \textit{tectonics}(\pm) + \textit{eustatic sea level rise}(\pm) \\ &+ \textit{basin water level}(\pm) + \textit{aggradation min}(\pm) \\ &+ \textit{aggradation org}(\pm) \end{aligned}$$

The overall impact on the marsh surface/water surface relationship of each of the activities listed was quantified. In addition, an accretion budget was developed for the Barataria Hydrologic Basin in order to further analyze the influence of man's activities on sedimentation processes in coastal Louisiana. The budget quantifies sources and sinks of sediment within the basin and the potential for certain management techniques to influence the budget. This two-step accounting procedure is a practical way of highlighting key impacts and processes, identifying the limits of our knowledge, clarifying future research needs, and laying a foundation for formulating management policy.

The Marsh Surface/Water Surface Relationship

A schematic illustration of the relationship between man-made/natural influences and accretion–RWLR processes in coastal Louisiana is presented in Figure 18-1. Subsidence is affected locally by fluid withdrawal and levee construction while geologic consolidation, eustatic sea level rise, and basin water level variations exert a regional influence on relative changes in water surface elevation. On the other hand, natural and man-made channels (and associated levees) exert a local influence on sediment (mineral and organic) accumulation and land-building processes. The net effect of these influences, when considered *in toto*, is a change in the marsh surface–water surface relationship that ultimately results in wetland loss or gain. During most of this century, the changes have resulted largely in wetland loss, except in a few sediment rich environments like the Atchafalaya delta.

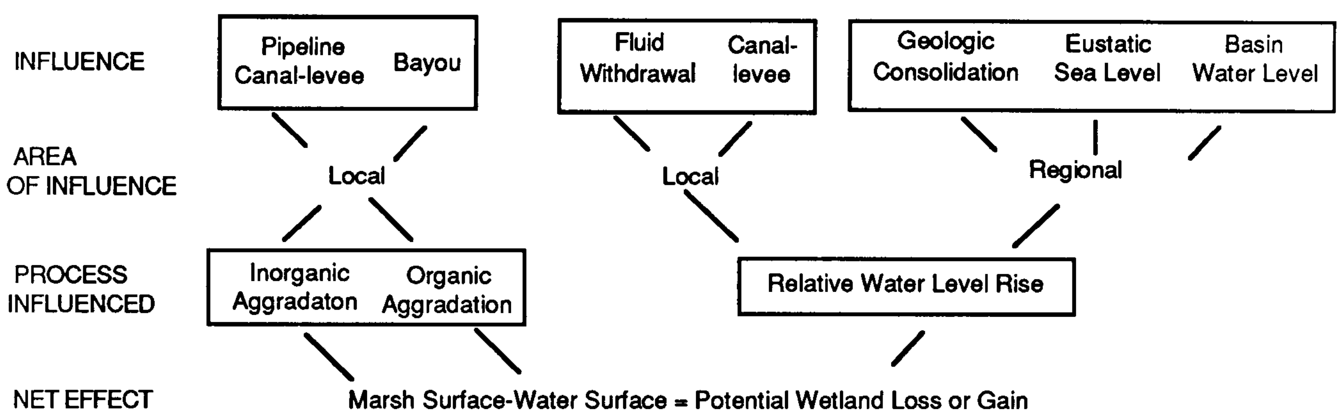

Figure 18-1. Relationship between man-made and natural influences on vertical aggradation and relative water level rise processes in Louisiana coastal marshes.

An estimate of the rate, duration, and areal extent of the influences diagrammed in Figure 18-1 is presented in Table 18-1. The rate estimates are based on direct measurements from this working group, values from the literature or a consensus opinion from this working group. Because of the heterogeneous nature of Louisiana's coast, rates are expressed as an approximate mean along with the range one might reasonably encounter along the coast. These means, therefore, represent a best estimate based on our knowledge of the coast. The duration estimates reflect the working group's consensus opinion on how long the marsh will experience the influence. An influence is determined to have a local impact if the impact on accretion or relative water level does not go beyond the boundaries of the hydrologic unit. For example, a canal dredged in one basin (that does not cross the basin boundary) will not influence the marsh surface/water surface relationship in another basin. However, a change in Gulf of Mexico eustatic sea level will affect all hydrologic basins and therefore has a regional influence. A local influence may even be limited to only a small portion of the basin, as in fluid withdrawal or subsidence caused by the weight of a levee. The area influenced by an activity was estimated from data collected by this working group or from the habitat summaries presented in Chapter 2, Direct Impacts.

Table 18-1. Estimated rate, duration, and area of man-made and natural influences on accretion and relative water level rise processes in the Louisiana coastal zone (see Figure 18-1 for illustration of relationships). Rate is expressed as an approximate mean, and the range is presented in parentheses.

Process Influence	Rate (cm yr-1)	Duration (yr)	Area (1000s ha)
Relative Water Level Rise			
Subsidence			
Fluid withdrawal	? (0-4)	25	Local (51)
Spoil levee[a]-subsidence	? (0-4)	2	Local (71)
Natural levee-subsidence	?	> 100	Local
Geologic consolidation (>1 m)	≈ 0.7 (0.2-2.5)	> 100	Regional (5000)
Marsh surface consolidation(<1 m)	≈ 0.2 (0.05-1.0)	< 100	Regional (5000)
Water level			
Eustatic sea level	≈ 0.23	> 100	Regional (5000)
Basin water level	≈ 0 (-0.2,+0.2)	20 cycle	Regional (5000)
" "	≈ 0 (-20,+20)	1 (annual)	Regional (5000)
" "	≈ 0 (-2,+2)	Monthly	Regional (5000)
Spoil levee[a]-hydrology	?	20-50	Local (71)
Aggradation of Marsh Surface (Organic ~ 30 (1-90%); Inorganic ~ 70 (10-99%))			
Natural waterway	≈ 0.7 (0.3-2.5)	< 100	Local (36)
Canal-levee[b]	≈ 0.7 (0.3-2.5)	< 100	Local (125)

[a] Man-made levees have two influences on submergence processes. They can compress or deflect the marsh surface downward within the immediate vicinity and they can influence water levels across the marsh surface.

[b] Levees of Mississippi River not included.

Relative Water Level Rise. Of the eight influences on RWLR, five are related to subsidence of the marsh surface and three to changes in water level (Table 18-1), with subsidence exerting the greatest influence. The variability in rate, duration, and areal extent of all eight influences precludes us from making coast-wide conclusions about RWLR. It would be misleading to sum all the rate estimates to obtain an average for the coast because some influences are strictly local in extent, relatively short in duration, or vary widely on a regional or seasonal basis.

Natural and human processes contribute to subsidence along the Louisiana coast. Fluid withdrawal is the most important human influence on subsidence at the local level. The average influence of fluid withdrawal on RWLR is difficult to quantify because the sphere of influence is restricted to the lateral extent, depth, and thickness of the individual underground reservoir. The subsidence rate also is influenced by the geologic province in which the reservoir is located. Although it may be locally important in the vicinity of shallow petroleum reserves, fluid withdrawal is a minor contributor to subsidence along the coast because it has the potential to substantially influence subsidence in only 1% (51,000 ha) of the coastal wetlands. Consequently, natural forces dominate subsidence processes along the Louisiana coast, with geologic consolidation being the major driving force. Geologic and marsh surface consolidation occur in all regions of the coast, with geologic consolidation exerting a long-term impact on subsidence.

Rates of water level rise are far more variable than subsidence rates and are dominated by natural processes. Over the past 80 years, water level rise has remained relatively constant at selected coastal locales (i.e., not accelerated) but has apparently fluctuated on a 20-30 year cycle. Eustatic sea level rise is relatively low, constant, and predictable. On the other hand, basin water levels are influenced by monthly, annual, and longer-term climatic variations (i.e., precipitation, run-off, river flow, winds, and storms) and, thus, can vary widely depending on basin physiography and regional climatic influences. However, man can also influence water levels by altering marsh topography. Swenson and Turner (1987) demonstrated that man-made levees can alter natural surface hydrology in the marsh when partial or complete impoundments are created. These hydrologic effects are localized, site specific, and difficult to quantify because they are influenced by local physiography, hydrology, and climate.

Accretion of the Marsh Surface. The rate of accumulation of mineral and organic solids on the marsh surface is determined by the supply and distribution of matter. Plant (i.e., organic) production is influenced by inundation, nutrition, and marsh type, while mineral accumulation depends on density at the source, distance from the source, and barriers to hydrologic exchange. The accumulation of mineral and organic matter are interwoven because they are both hydrologically mediated and one provides the nutritional source for the other. The organic and inorganic components of the substrate can vary widely (1 to 90% and 10 to 99%, respectively) depending on the factors described above, but a reasonable average for the Louisiana coast would be 30% organic and 70% inorganic.

In Table 18-1, the coast-wide average for marsh accretion is assumed to be the same for natural waterways and canals because of the wide range of environmental settings in coastal Louisiana. Accretion rates vary notably with marsh type and the range in yearly accretion rates is noteworthy for both waterway types. Apparently this sizable range is caused, in part, by the variable influences of canals.

Net Effect. Our rough accounting analysis has revealed that the rates of RWLR may often exceed marsh surface accretion, at least over short periods of time ($\leq$ 25 years), depending on the environmental setting. We define the situation in which the water level (rate of RWLR) exceeds the marsh surface elevation (rate of marsh surface accretion) as the surface disparity. This relationship can be expressed as follows:

R = Relative water level rise = 1.2 cm/yr
A = Vertical marsh accretion = 0.7

and

Disparity	=	R - A	Eqn 18.1
	=	1.2 - 0.7 cm/yr	
	=	0.5 cm/yr	

The net effect of RWLR and marsh accretion on wetland loss or gain should be considered on a case-by-case basis (i.e., within a hydrologic basin or a portion of one) because of the heterogeneous nature of the coast. Interpretations across hydrologic basins should be made only if substantial evidence exists for a broad-scale impact.

Natural processes dominate the marsh surface/water surface relationship. Nevertheless, it can be said that the potential for surface disparity to occur (for RWLR to exceed marsh surface accretion) increases in the vicinity of canals because the localized effects of canal construction and oil and gas development (levee influence on subsidence of the marsh surface and surface hydrology, and fluid withdrawal) are additive with all regional influences. It should also be noted that, even though accretion rates behind spoil levees were essentially the same as behind natural levees, it is still possible for a surface disparity to develop in the vicinity of a canal/spoil bank, but not at a nearby natural levee. It is known that spoil levees can influence marsh water levels (Swenson and Turner, 1987). If a spoil levee exerts a greater influence on water level than a natural levee, a disparity could develop behind spoil bank sites but not at natural sites.

Accretion Budget for Barataria Hydrologic Basin

Wetlands develop in dynamic depositional environments in which subsidence, oxidation, water level rise, aggradation of mineral and organic matter, and local hydrology influence the marsh surface/water surface relationship. Accretion rates depend on the quality, as well as the quantity, of material available and are influenced by particle size, source, distance from source, and local hydrology and topography. Such complexity makes it difficult to model or budget accretion processes. However, a general accounting of accretion processes can be useful, as explained earlier. Therefore, an accretion budget was developed for the wetland habitats of Barataria Hydrologic Basin (Figure 18-2) in an attempt to quantify some of the processes that influence land building in coastal Louisiana.

The accretion budget (Table 18-2) was developed for a single hydrologic unit, rather than the entire coast, because of the reduced complexity and better control encountered at the regional level. It is based on static estimates of these highly dynamic processes. All estimates are liberal because they assume all available matter was deposited only on wetlands, not open water. Therefore, these are estimates of maximum influence. Some of the estimates are based on findings from this study or from the literature, but most are best "guesstimates" based on our knowledge of related coastal processes in the vicinity of Barataria Basin. Hence, this budget is developed mostly from unpublished data.

The budget points out relationships between sources and sinks of suspended matter (Table 18-2). Recent investigations in Atlantic coastal estuaries have revealed that bottom transport of sediment may be an important aspect of sediment flux in these systems. However, such investigations have not been conducted in Louisiana estuaries. Therefore, the importance of bottom transport of sediment to the estuarine sediment budget of

Barataria Basin is not known, and we made no attempt to quantify it in the budget. Without considering the potential input of material through tidal passes and the sediment dynamics of bay bottoms, the amount of material made available for deposition (+2.5 to 13.0 mm/yr) ranges from 50 to 200 % and 20 to 100 % of minimum (+6 mm/yr) and maximum (+12 mm/yr) measured deposition rates, respectively.

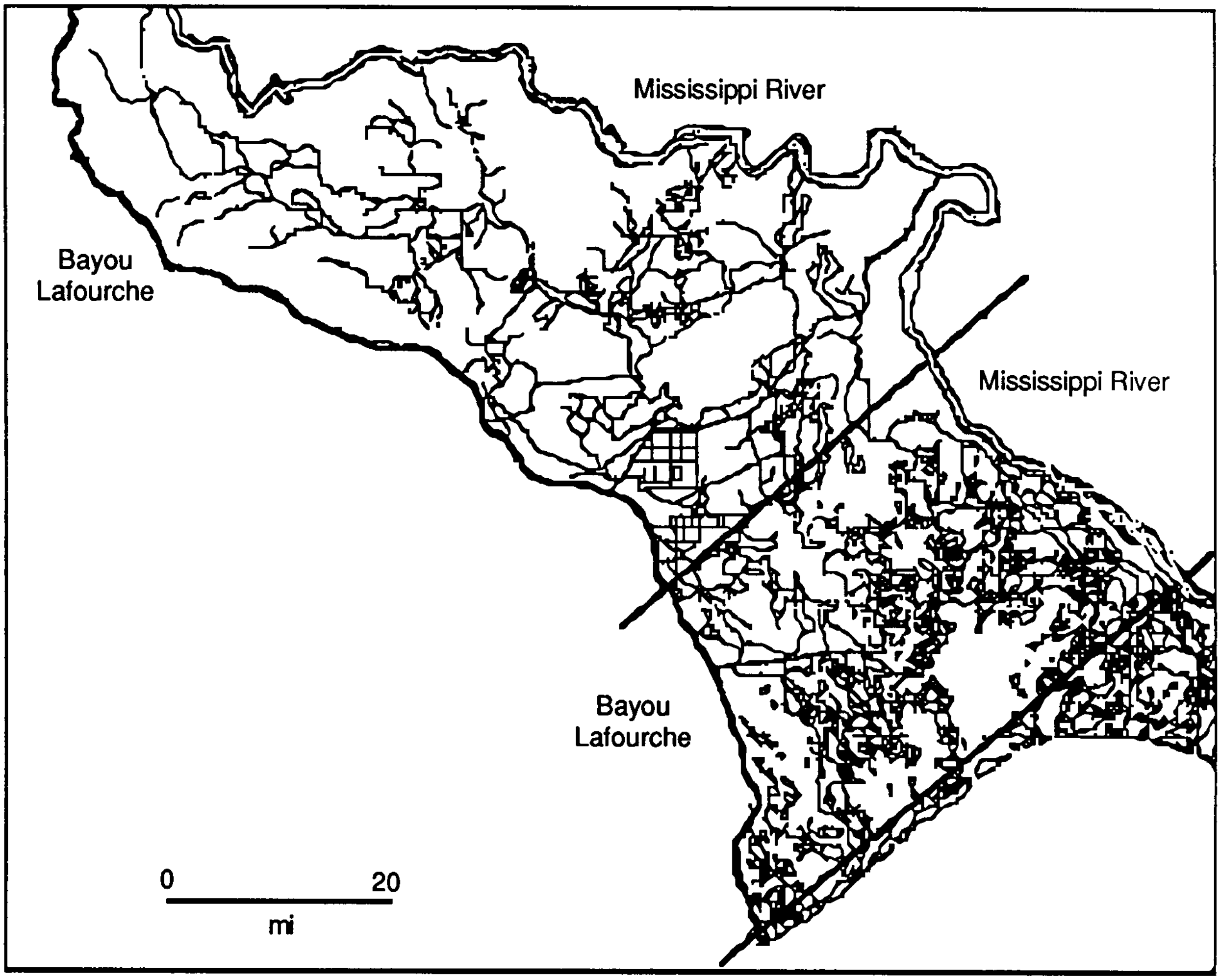

Figure 18-2. A map of the Barataria Hydrologic Basin.

If the net movement of matter through the tidal passes during the past 30 years was out of the basin, considerably less material was made available for sediment deposition through input of suspended matter than through resuspension of eroded and deteriorated marsh substrates. This is attributable, in part, to leveeing of the Mississippi River and attenuation of flow down Bayou Lafourche. Restoration of these sediment inputs (crevassing and overbank flooding) would increase vertical accretion in selected areas along the River by at least 1 to 2 mm/yr; even more in areas of crevassing.

In summary, this budget represents a gross approximation of many sedimentary processes in Barataria Basin. If a substantial amount of sediment is not entering the basin through the tidal passes via bottom transport, then the marshes are largely dependent upon resuspension of matter from disintegrating marshes which provide ≈ 4 to 8 mm/yr (75% of the amount deposited). Additional information is needed to more fully quantify the source and amount of sediment currently entering the basin and resuspended from marshes and bay bottoms.

Table 18-2. Estimated aggradation budget for wetland habitats in Barataria Hydrologic Basin showing the relationship between inputs, resuspension, and deposition.[a]

INPUT			
Storm Overwash	+0.35 mm/yr (coast) +0.035 mm/yr (inland)		
Tidal Pass Exchange	+ [? mm/yr from offshore] or -1.0 to -2.0 mm/yr from estuary		
Crevasse	insignificant (w/o levees = 109 mm/129.5 km2)		
Overbank	insignificant (w/o levees = 1.6 mm/647.5 km2)		
Controlled Flow (Bayou Lafourche)	insignificant		
Upland Runoff	+0.42-5.6 mm/yr		
RESUSPENSION		DEPOSITION	
Erosion of Banks Plus Pond Formation	+4 mm/yr (30 cm depth) +8 mm/yr (60 cm depth)	backmarsh streamside	+6-10 mm/yr +10-12"
Bottom Sediments	+ or -, or = subsidence		
TOTAL	+2.5 to 13.0 mm/yr	+6 to 12 mm/yr	

[a] Aggradation is expressed as vertical accretion (mm/yr) of sediment (with an assumed bulk density of 0.21 g cm^{-3}) on wetland habitat only, not open water or upland habitats. Therefore, these are liberal estimates of vertical accretion. The estimates for crevasse and overbank input and deposition rates, respectively, are based on direct measurements from this report or the literature (see below). All other estimates are best "guesstimates" based on our knowledge of sedimentation processes along this region of the coast. The rationale behind each estimate is explained below.

Storm Overwash

Maximum and minimum input averaged over 100 years, ≈ 750 kg/ha/yr calculated a bulk density of 0.21 g/cm^{-3} and a sediment concentration of flood waters of 50 ppm.

Tidal Pass Exchange

A loss from the basin of 200 x 10^6 kg/yr was calculated because observations (review of aerial photographs) of net non-tidal flow indicate that density of the suspended matter exiting the basin through the tidal passes appears to be greater than the density of matter entering the basin (5,000 cfs averaged over a tidal cycle with a sediment concentration of up to 50 ppm). The total amount of water transported into the basin during a tidal cycle is estimated at 3,600 million cubic feet, based upon a bay surface area of 7,200 million square feet and an average tidal height of 15 cm over the bays. If this mineral matter came entirely from the marsh surface, it would result in a 1-2 mm/yr decrease in aggradation. However, this material may come from bay bottoms and/or upland runoff. Thus, 1-2 mm/yr is a maximum estimate.

It is possible that tidal passes may be a net source of mineral sediment to the basin because basin water levels appear to be directly affected by Mississippi River flow (see Figure 6-10 in Chapter 6), indicating that net non-tidal flow may be into rather than out of the basin. It remains to be determined if Mississippi River water entering the basin brings in more sediment than leaves on an ebb tide. Unfortunately, there is no reliable data base on density of suspended sediment entering and leaving the basin.

Crevasse

If a crevasse occurred in the Mississippi River levee during the 1973 flood the size of Bonnet Carre spillway, and assuming the mineral matter is spread evenly on land, it would deposit 109 mm over a 129.5 km^2 area. (See Chapter 12 in this report).

Overbank Flooding

If the levees were removed, the 1973 flood would have deposited 1.6 mm of mineral matter over the 647.5 km^2 of marsh area indicated between the parallel lines in Figure 18-2. (See Chapter 12 in this report).

Upland Runoff

Upland runoff was estimated from the areal estimate of urban and agriculture land in the basin and sediment yield. Land use patterns were from the habitat map data for 1955 and 1978. A sediment yield for agriculture and urban zones of 3 cm/yr was from USDA soil loss equations for row crops. We divided that number by 10 to obtain a conservative runoff rate. This loss was averaged for the entire remaining portion of the basin.

Controlled Flow--Bayou Lafourche

We estimated that 18-36 million kg/yr of sediment is diverted from the Mississippi River into Bayou Lafourche at Donaldsonville, La. This estimate is based on a discharge from Bayou Lafourche of 200 cfs and a density of suspended solids of 100-200 ppm. If 50% of the discharge leaves the bayou through Southwestern Louisiana canal and is spread evenly over the western half of the marsh area indicated between the parallel lines in Figure 18-2, this would amount to ≤ 0.001 mm/yr vertical accretion.

Erosion of Banks plus Pond Formation

The amount of mineral matter available for resuspension through erosion and loss of marsh habitat was calculated from land loss rates between 1955 and 1978. The total area of wetland habitat lost was 86,518 ha and the total area of wetlands remaining in 1978 was 164,229 ha. The estimates are based on an average bulk density for the entire basin of 0.21 g cm^{-3}, an average mineral composition of the soil of 60%, and erosion occurring to a depth of 30 and 60 cm.

Bottom Sediments

We have no data on changes in the depth of open water lakes and ponds within the basin. Consequently, we do not know if bay bottoms are sinks or sources of sediment. If the depth of the water bottoms is not changing with time then bay bottoms would be a sink for sediments at a rate equal to local subsidence rates.

Deposition

These estimates are taken from the direct measurements in Chapters 14, 15, and 16 of this report and from the literature (DeLaune et al., 1978 and Hatton et al., 1983).

PART V

LANDSCAPE PATTERNS AND AERIAL IMAGERY

The principal responsibility of the landscape patterns working group was to test various hypotheses about habitat change on the scale of landscapes. The scale of habitat change investigated is unique among all projects in this report, and that was intentional. Our scale included mapping units ranging from 100 m2 to 16,000 ha, and distances from habitat change to geomorphic features from 0 to 100 m vertically and 0 to 50 km horizontally. The spatial overlap in study sites and scales was intentional for several reasons. First, although we had a variety of data resources to develop and techniques to apply, we did not know a priori what scale was most appropriate and did not presume that one was sufficient to meet all needs. Second, since the deltaic geomorphology and spatial distribution of man's impacts are heterogenous at a landscape scale, we judged that data collected at that same scale would minimize local impacts and the regional (landscape) impacts would become apparent. Thirdly, if regional interactions between local factors (geomorphology and man's impacts) vary significantly within landscape units (e.g., hydrologic basins), then we needed to test for them at a scale smaller than that landscape unit but larger than the factors.

The three projects analyzed aerial images of habitat changes from 1955/6 to 1978 based on maps produced by Wicker (1980, 1981). Habitat change was either modelled or described in each. Cowan and Turner (Chapter 19) used net habitat changes in the 7.5 minute quadrangle maps (comprising 76% of the coastal zone) to select appropriate variables for the subsequently developed statistical models of wetland loss and definition of regions. Leibowitz and Hill (Chapter 20) examined gross landloss rates in the three areas shown in Figure 1.1 (12.5% of the coastal zone land mass in 1955/6), using sample sizes ranging from 100 m2 to 1 km2 within randomly selected subsets of the three individual study area. Turner and Rao (Chapter 21) examined net pond formation in four size categories within selected 7.5 minute quadrangle maps (16,000 ha) which together accounted for 35% of the open water formation in the coastal zone from 1955/6 to 1978. Each examined the distribution and/or number of habitat changes with respect to geology, distance, and density measures.

Chapter 19

MODELING WETLAND LOSS IN COASTAL LOUISIANA: GEOLOGY, GEOGRAPHY, AND HUMAN MODIFICATIONS

by

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Louisiana's coastal wetland loss rate (> 100 km^2 yr^{-1}; 0.8% of total annually) is a chronic state problem. Some of the implications of this loss include decreased fisheries, economic loss (oil and gas revenues), and increased storm damage (Craig et al., 1979). Causes for these losses are complex but have been attributed to both natural and man-induced factors (Chapters 1 and 3; Walker et al., 1987).

Statistical (quantitative) methods can be used to test hypotheses about the relative importance and interaction of various potential causal factors (natural and man-induced) that are correlated with landscape changes, as well as to understand possible options for resource management and mitigation potential. Man may influence wetland loss in coastal Louisiana through flood control measures, urban and agricultural practices, and canal and spoil bank construction. Natural factors include changes in local geology and hydrology caused by the dynamic nature of Louisiana's sedimentary coastline. Although locally significant influences (causes) tend to be obscured as landscape size increases, analysis on a regional scale is one way to isolate and quantify regionally significant factors.

Unfortunately, few habitat data contain sufficient temporal and spatial resolution on a regional scale to be both useful in a quantitative analysis of land loss and helpful to natural resource managers, particularly for coastal ecosystems. An understanding of the potential causal mechanisms of wetland loss in Louisiana could be enhanced by combining in one analysis those factors previously absent, or only partially included in other studies, i.e, quantitative instead of qualitative data (Gagliano 1973; Craig et al., 1979; Walker et al., 1987), inclusion of the whole coast instead of just selected areas (Scaife et al., 1983), and combining geologic factors into the spatial analyses (Deegan et al., 1984).

There is one habitat change study of the Louisiana coast, completed for the U.S. Fish and Wildlife Service (USFWS) by Wicker et al. (1980), that we believe to have sufficient resolution to allow a quantitative analysis of some of the mechanisms believed to influence land loss. That study determined changes in 200 habitat categories (following the Cowardin et al., 1979 classification scheme) from 1955 to 1978 in 464 (232 for 1955; 232 for 1978) 7.5 minute (1:24,000 scale) topographic quadrangle map units (Figure 19-1). Scaife et al. (1983) used these data to describe land loss in selected geologic substrates as a function of canal density. In a preliminary analysis of land loss in the Mississippi River Deltaic Plain, Deegan et al. (1984) used the habitat data to integrate the regional geologic influences with man-induced factors into one model; for reasons to be discussed below, their analysis was flawed. We report here our use of the Louisiana habitat data to quantitatively relate land loss (primarily wetlands in the form of coastal marshes) to both man-induced and natural geologic factors believed to influence (i.e, cause) habitat change.

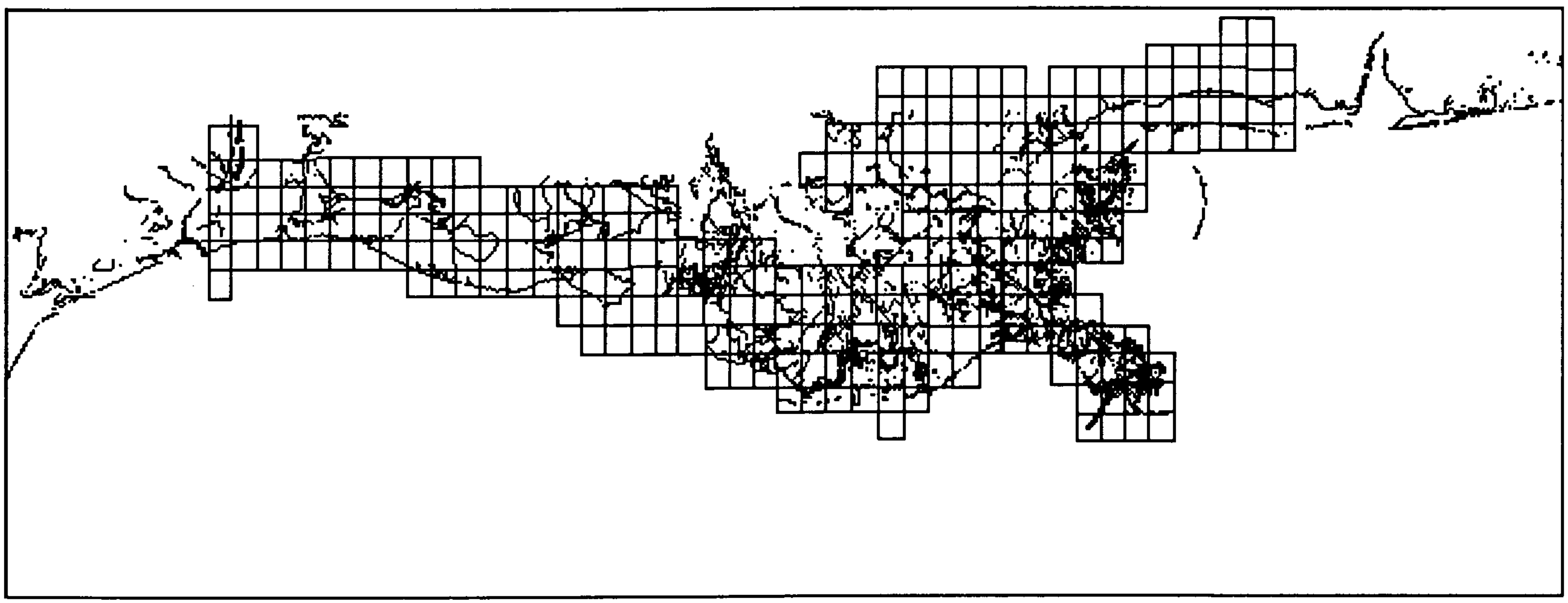

Figure 19-1. Location of topographic map units interpreted and measured within the study area (adapted from Wicker et al., 1980).

The Louisiana Coast: Wetland Loss and Contributing Factors

The Louisiana coastal zone was formed over the last 7,000 years by sediments deposited during a series of 16 major Mississippi River deltaic episodes (Frazier, 1967; Kosters et al., 1987; Walker et al., 1987). The landscape now consists of narrow ridges of high ground a meter or so above sea level located along abandoned river distributaries, between which lie vast expanses of low-lying marshes. These wetlands comprise more than 40% of the coastal wetlands in the conterminous United States and more than 65% of the marshes surrounding the U.S. Gulf of Mexico (Turner and Gosselink, 1975; Deegan et al., 1984). These expansive wetlands are rich in renewable resources; Louisiana supports the nation's largest commercial fishery with landings approaching 2 billion pounds in 1986 (U.S. Department of the Interior, 1987) and leads the U.S. in fur-bearer harvest (Chabreck, 1979). Hunting and recreational fishing contribute $235 million annually to the economy of the state (U.S. Department of Commerce, 1982). Coastal Louisiana is also rich in oil and gas resources and ranked third in crude oil and first in natural gas production in the United States (American Petroleum Institute, 1981). Oil and gas activities in these coastal wetlands consist primarily of canal dredging for pipeline construction and drilling site access; construction of major navigation channels has also taken place (Deegan et al., 1984). Dredging and its associated activities (e.g., spoil banking) represent significant development pressure on Louisiana's coastal wetlands and nearly one-third of all U.S. Army Corps of Engineer (COE) permitted dredge and fill activities occur in Louisiana (Mager and Hardy, 1986). Consequently, development of oil and gas resources in Louisiana has spurred much habitat alteration in the coastal zone.

Historically, the biological and physical factors which contribute to wetland development or loss have been nearly in balance along the Louisiana coast, resulting in wetland gain with some periodic episodes of localized wetland loss (Cowan et al., 1986). Currently, however, the amount of sediments deposited by riverine systems or accumulated by biological processes appears to be less than necessary to match relative sea level rise (subsidence and eustatic sea level rise combined) (Chapter 3; Turner, 1985; Cowan et al., 1986; Cahoon and Cowan, 1987). Virtually all of the land loss in Louisiana occurs as wetland loss which is a complex process influenced directly and indirectly by natural and man-induced activities. The term wetland loss refers to the conversion of wetland habitat type, to either open water or upland habitat (spoil bank, agriculture, or urbanized; Cahoon et al., 1986; Cahoon and Cowan, 1987). Land loss can result from a variety of interrelated causes: 1) natural and man-induced erosion of shoreline or the banks of waterways and canals; 2) dredging and filling of marshes by man, primarily associated with the oil and gas extraction industry; and, 3) submergence of interior marshes. Submergence occurs when natural land building or maintenance processes (sedimentation and accumulation of plant matter) lag behind geologically mediated rise of relative sea level (subsidence, compaction, consolidation, etc.). Sediment compaction rates of deltaic deposits depend on the age and location within the delta lobe, the amount and type of sediment input, and the depth of sediment overlying the down-warped Pleistocene terrace (Kolb and Van Lopik, 1958; Morgan, 1963; Walker et al., 1987). In general, sediment compaction in Louisiana's coastal wetlands decreases as the distance from the coast increases (Kolb and Van Lopik, 1958; Deegan et al., 1984). Sediment input and organic accumulation counteracts compaction and contributes to land accretion. Sediment input to marshes is achieved by overbank flooding of rivers (e.g., Mississippi and Atchafalaya) bays, bayous, and other waterways. However, the supply and distribution of sediments are not static in recent time. Suspended sediments in the Mississippi River apparently declined by more than 50% since the early 1950s (Chapter 12; Meade and Parker, 1984) . Furthermore, the lower Mississippi River has flood-protection levees which reduce over-bank flooding except near the River's lower reaches.

The loss of wetlands by erosion and dredge/fill activities is caused by a direct disruption of the substrate by natural or man-induced mechanical stress (i. e., waves, boat wakes, and dredges), resulting in either open water or upland habitat (Chapter 3; Cahoon et al., 1986). Some of these impacts are immediately apparent as dredging activities have directly converted 55,000 ha (~10%) of Louisiana's coastal wetlands to open water habitat since 1900 (Lindstedt and Nunn, 1985). However, canals and their associated spoil banks also restrict or eliminate regular overbank flooding (Chapter 3; Davis, 1973; Gosselink et al., 1979; Craig et al., 1979; Turner et al., 1982; Turner, 1985; Day et al., 1986; Cahoon et al., 1986; Cahoon and Cowan, 1987). This results in an indirect, less readily apparent impact on the submergence of interior marshes. Indirect impacts have been estimated to cause 25-90% of wetland loss in Louisiana (Turner, 1985, for review). Coastal submergence is influenced by these activities because levees (spoil banks) affect the duration and frequency of tidal inundation, which in turn, affect sediment and nutrient supply, as well as the availability of oxygen and toxins that may influence plant growth and the deposition of organics (Cahoon et al., 1986; Swenson and Turner, 1987). This is particularly true for areas that unintentionally become semi-impounded and no attempt is made to manage the hydrologic regime (Cahoon et al., 1986; Cowan et al., 1986, for review; Cahoon and Cowan, 1987).

Methods

Data used in this analysis were derived from the Louisiana habitat mapping study of Wicker et al. (1980). Habitat area in the Louisiana coastal zone in each map unit (7.5 minute quadrangle; 1:24,000 scale) was planimetered from images built from aerial photography. Wicker et al. (1980) constructed the images for 1955/56 and 1978 because the entire Louisiana coastal zone was flown and photographed from high altitude during those years.

We combined the 200 habitat types into 20 broad categories (Table 19-1) following Costanza et al. (1983) and restricted our analysis to 166 quadrangle map units. Maps were eliminated from this analysis if: (1) wetlands within the quadrangle were part of an active river delta and receiving significant new sediment deposition (Kosters et al., 1987); (2) wetland area within the quadrangle was less than 2.5% (405 ha) of the total area; or, (3) the total measured area in a quadrangle in 1955-56 was not within 0.5% (80 ha) of the total measured area in the 1978 map. Our reasons for selecting these criteria were (1) to limit the analysis to marshes not undergoing rapid change caused by sediment deposition, and, (2) to decrease the proportion and opportunity for error since mapping errors decrease with increased areal coverage of marshes in a map. Several (~30) quadrangle maps were eliminated because they contained large-scale human modifications (i. e., changes in the area of urbanization, agriculture or impounded wetlands) that occurred between 1955/56 and 1978; consequently, these appeared as outliers. Data analyzed in this study represent approximately 76% of the area of coastal marshes in Louisiana (Table 19-2).

Table 19-1. Habitat classifications used to develop land use groups analyzed in chapter 19.

Land use Category	Habitat Classification (based on Costanza et al. 1983)
Marsh	Brackish Marsh, Fresh Marsh, Salt Marsh
Swamp	Bottomland Hardwood, Cypress-tupelo Swamp, Mangrove
Forest/Upland	Fresh Shrub-scrub, Upland Forest
Aquatic Grass Bed/Mudflat	Fresh Aquatic Bed, Estuarine Aquatic Bed, Mudflat
Canal and Spoil	Canal, Spoil
Open Water	Fresh Open Water, Estuarine Open Water
Urban/Agriculture	Agriculture, Urban/Industrial
Beaches and Dunes	Beaches, Sand Dunes

Table 19-2. A summary of the area (ha) of the land use categories in the original Louisiana habitat data (Wicker et al., 1980) compared with the area in the analyzed data set.

1978 Habitat Category	Complete Data Set (A)	Analyzed Data Set (B)	B/A X 100 (percent)
Marsh	1,009,320	827,642	82.2
Swamp	177,078	127,496	72.0
Forest/Upland	57,550	45,465	79.1
Aquatic Grass Bed/Mudflat	26,788	16,341	61.1
Canal and Spoil	80,426	64,904	80.7
Open Water	2,162,776	1,641,547	75.9
Urban/Agriculture	211,848	169,478	80.0
Beaches and Dunes	4,758	3,331	70.1
Total Area All Habitats	3,728,402	2,831,949	76.0

We chose variables to represent both natural and man-induced factors which we believe to influence marsh loss. Natural factors include coastal morphology and sediment input and age, compaction, and subsidence; man-induced factors are related to development in the coastal zone, e g., canal dredging, agriculture, and urbanization. Wetland loss (primarily coastal marshes) is defined as the difference between marsh area in 1955/56 and 1978. A positive number represents a loss in marsh area. The initial wetland (marsh) area is given as the total marsh area in the 1955/56 quadrangles. The variable "age" is the estimated age (years) of the sub-delta lobe that underlies the coastal marshes (Frazier, 1967). Depth is the depth (m) of sediment overlying the Pleistocene Terrace (Fisk and McFarlan, 1955). The variable distance corresponds to the distance (m) from the center of each quadrangle to the Louisiana coast on 1974 USGS, 1:250,000 maps. The estimates of age and sediment depth are not precise because the Louisiana coast was formed by a series of overlapping deltaic episodes stacked one on top of another. Consequently, the estimated age of the last deltaic episode was used in this analysis following Deegan et al. (1984).

The percent of marsh lost in a quadrangle map unit was modeled as a function of the area of agriculture and urban development (DEVDENS) in 1978, the area of canals and spoil (CANDENS), the estimated age of sediments (SEDAGE), the depth of sediments (DEPTH), and distance to the coast (DISTANCE). These 5 independent variables and their units were calculated in the following manner:

PERCENT = change in marsh area (ha) in a quadrangle between 1955/56 and 1978 divided by area (ha) of marsh in 1955/56;

DEVDENS = the area (ha) of urban and agricultural development combined (1978) divided by area (ha) of marsh in 1955/56;

CANDENS = the area (ha) of canals and spoil combined (1978) divided by area (ha) of marsh in 1955-56;

SEDAGE = estimated age (yrs) of sediments;

DEPTH = depth (m) of sediments; and,

DISTANCE = distance (m) to the coast.

All variables were standardized in a correlation matrix to prevent problems caused by different units of measurement. These variables are similar to those that Deegan et al. (1984) used to model land loss (based on the original Wicker et al. (1980) data), for the Mississippi River Deltaic Plain, and their initial analysis positively influenced this current study. However, their analysis was flawed. Although they determined that their predictive variables were not interdependent, the dependent variable representing marsh loss was calculated by subtraction (Marsh loss = the area of marsh in 1955/56 (IMARSH) minus the area of marsh in 1978) from the predictive variable which accounted for the greatest amount of variability in their modeled data (i.e., IMARSH). Consequently, their data were multicolinear, and their conclusion that marsh loss in a quadrangle was biologically related to stability caused by resistance to erosion may be erroneous.

In this study, the dependent variable PERCENT, as well as other variables representing area in a habitat category in 1978 (e.g., DEVDENS, CANDENS) were first normalized to the area of marsh in 1955-56. Principal components analysis (PCA) was used to test for multicolinearity among variables (dependent and predictive) and to determine if they accounted for a significant portion of the variability in the original data set (StatView 512+, BrainPower, Inc.). Factor score weights were calculated by using an orthogonal transformation solution and varimax rotation (Muliak, 1972). Scores greater than 0.314 were considered significant. The value of 0.314 is arbitrary, but implies that at least 10% of the variance for any given variable is accounted for by the factor on which it loads. Variables identified as significant in the PCA were employed in regression analysis (StatView 512+, BrainPower, Inc.) to quantitatively model their relationship to the percent of marsh lost in a quadrangle map unit between 1955/56 and 1978. As a final examination, the quadrangle map units were ordinated (clustered) by using a clustering procedure (PROC FASTCLUS; SAS Institute Inc., 1982), which uses the nearest centroid clustering algorithm following Anderberg (1973).

Results and Discussion

Principal Components Analysis

Results from principal components analysis (PCA) indicate that the variables chosen to model marsh loss combined together account for a significant portion of the variability in the original data (77.2%; $P < 0.0001$) and that the variables are not interdependent. However, interpretation of the variable factor score weights (Table 19-3), along with the proportionate variance contributions of each factor (0.395, 0.310, and 0.294 for factors 1-3, respectively), suggest that the relationship among the variables is complex. No variable loaded highly on more than one factor and no factor accounted for a disproportionate amount of the variability. However, some patterns are evident and warrant discussion. The only variables to score highly on factor 1 were sediment age and sediment depth; these variables were inversely related to one another. The variables representing land loss and canal density loaded (PERCENT) highly on factor 2, while distance to the coast and developed area scored high on factor 3. The inverse relationship between sediment depth and age on factor 1 reflects that younger sediments apparently overlie the Pleistocene Terrace more thickly than older ones. Consequently, these young sediments may be more susceptible to compaction and consolidation. The variable scores on factor 2 suggest a relationship between increased canal and spoil area in the coastal zone with increased marsh loss. Finally, the variable scores on factor 3 show that the area of development is related to distance, indicating that activities associated with urbanization and agriculture most frequently occur some distance away from the Louisiana coastline. The variable representing marsh loss also loaded highly (but not significantly) on factor 3.

Table 19-3. Factor score weights on principal components calculated by using an orthogonal transformation solution and varimax rotation. The factors account for 77.2% of the variability ($P < 0.01$) in the original data set. Factor loading greater than 0.314 are considered significant.

Variable	Factor 1	Factor 2	Factor 3
Percent[a]	0.158	0.508	0.299
Distance (m)	0.041	-0.206	0.648
Sediment Age (yrs)	-0.516	-0.120	-0.118
Depth (m)	0.486	0.035	-0.148
Canal Density (%)	-0.148	0.590	-0.127
Development Density (%)	-0.051	0.215	0.509

[a] Percent = marsh (ha) 1955-1978 x 100 divided by marsh (ha) 1955

Multiple Regression Analyses

The results of the PCA suggest that each of the selected variables needs to be included in this linear model to quantify the potential causal influences of marsh loss. Consequently, multiple regression analysis was employed to develop the following relationship (Table 19-4A):

$$\begin{aligned} \text{PERCENT} = 15.46 &+ 4.971 \text{ *CANDENS} \\ &+ 0.648 \text{ *DEVDENS} \\ &+ 0.216 \text{ *DEPTH} \\ &- 0.000995 \text{ *SEDAGE} \end{aligned} \qquad \text{Eqn.19.1}$$

The relationship of percent marsh loss in a quadrangle map unit to the predictive variables was highly significant ($P < 0.0001$, $R^2 = .40$) (Table 19-4A). The precision of the model is in Figure 19-2, which is a plot of the modeled percent marsh loss based on Equation 19.1 versus the observed percent marsh loss obtained empirically from the data. The regression analysis agrees well with the PCA results, even though the variable representing distance to the coast (DISTANCE) was dropped from the regression model because of lack of significance.

Although the regression relationship was significant, the variability in the original data explained by this linear model was relatively low ($R^2 = .40$). Consequently, care must be exercised when attempting to predict or back calculate land loss (PERCENT) in any given map unit in this data set; the model is particularly poor when landloss rates are high. As will be discussed later, this may be a consequence of attempting to build a model to predict regional land loss for an area where local conditions differ enough across the region to necessitate a more localized approach, rather than our failure to include variables representing important potential causes.

Nevertheless, the variable representing canal and spoil area (CANDENS) was highly significant ($P < 0.0001$) and accounted for the greatest amount of marsh loss. The positive regression coefficient (+4.971) indicates that quadrangles with high canal density exhibited greater percent marsh loss from 1955/56 to 1978 than quadrangles with lower canal density. Canal and spoil impacts have been implicated in Louisiana wetland loss by several studies (Chapter 3; Scaife et al., 1983; Turner, 1985).

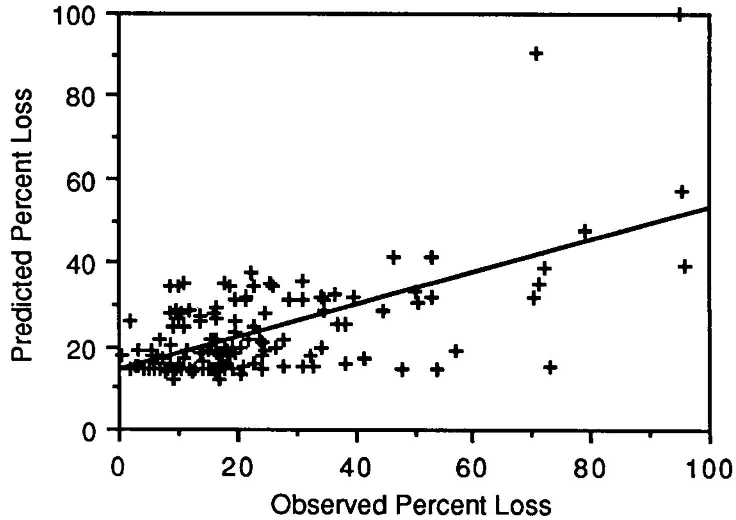

Figure 19-2. Predicted percent marsh loss versus observed percent marsh loss based on the results of the regression model given in Eqn. 19.1.

The area of development (DEVDENS) was also highly significant ($P < 0.0001$) in this linear model; nearly 65% of the increase in urbanized and agricultural area occurred at the expense (positive regression coefficient = +0.648) of coastal marshes. These results support the PCA, which suggests an inverse relationship between development and proximity to the Louisiana coast. Historical data show that urban and agricultural development in the Louisiana coastal zone first occurred on upland levee ridges inland from the coast. Recently, development has occurred in adjacent marshes (e.g., urban expansion and impoundment for agriculture) because much of the suitable uplands have already been developed (Deegan et al., 1984).

The most important geologic variable related to marsh loss is the depth of sediment which overlies the down-warped Pleistocene Terrace. The variable representing sediment depth (DEPTH) is highly significant ($P < 0.0003$), and the positive regression coefficient (+0.216) indicates that percent marsh loss is greater in quadrangles with greater sediment depth. Sediment compaction, dewatering, and the resulting subsidence is greater as the depth of sediment increases (Fisk and McFarlan, 1955). Because this and a previous attempt to quantitatively model land loss in Louisiana (Deegan et al., 1984) show no significant relationship between distance from the coast and marsh loss, the empirical relationships between distance and sediment compaction (Kolb and Van Lopik,1958; Scaife et al.,1983) and distance and land loss may be caused instead by sediment depth.

The last variable to enter the regression model ($P< 0.0368$) was the geologic variable representing the age of underlying sediments (SEDAGE). A negative regression coefficient (0.000995) indicates an inverse relationship between the percent of marsh loss in a quadrangle with the age of underlying sediments. Compaction and dewatering rates of deltaic sediments depend on several factors, including the age in years of the delta lobe in which the sediments were deposited (Walker et al. 1987); i.e., subsidence caused by compaction decreases with sediment age. Consequently, marsh loss is apparently higher in areas overlying more recently deposited sediments.

Table 19-4. Analysis of regression summary values for a multiple regression model relating marsh loss to natural and man-induced causal influences. A. All map units combined, B. The Mississippi River Deltaic Plain, and, C. The Chenier Plain.

	Source	DF	P>F	Estimate	R^2
A. All Map Units Combined:					
	Model	4	0.0001[b]	—	0.40
	Error	1330			
	Total	137			
	Intercept	—	—	15.46	
	CANDENS	1	0.0001[b]	4.971	
	DEVDENS	1	0.0001[b]	0.648	
	DEPTH	1	0.0003[b]	0.216	
	SEDAGE	1	0.0368[a]	-0.000995	
B. Mississippi River Deltaic Plain					
	Model	4	0.0001[b]	—	0.46
	Error	104			
	Total	108			
	Intercept	—	—	14.90	
	CANDENS	1	0.0001[b]	36.91	
	DEVDENS	1	0.0451[a]	0.322	
	DEPTH	1	0.0005[b]	0.197	
	SEDAGE	1	0.0459[a]	-0.002253	
C. Chenier Plain					
	Model	3	0.0001b	—	0.58
	Error	25			
	Total	28			
	Intercept	—	—	6.31	
	CANDENS	1	0.0001[b]	4.70	
	DEVDENS	1	0.0143[a]	104.42	
	DEPTH	1	0.5618(NS)	0.576	

[a]Statistically significant ($P<0.05$)
[b] Highly significant ($P<0.01$)
(NS) Not significant

We previously suggested that the regression model's lack of precision and low R^2 were a consequence of trying to predict land loss on a regional scale for a highly variable ecosystem (i.e., Louisiana coastal zone) rather than a consequence of excluding variables representing other important potential causal factors. In order to illustrate this point, we sub-divided the Louisiana coastal zone into several smaller units, based on either geology (e.g., Mississippi River Deltaic Plain (MRDP) versus Chenier Plain (CP)) or hydrologic unit (9 units; Figure 19-3). In each case, the sub-dividend was analyzed by using the linear regression model employed for the whole coastal zone. The results of the analyses for the MRDP and CP are in Table 19-4B and 19-4C, respectively. In both cases the regression model was highly significant and the R^2 improved over the original analysis. However, the variable representing sediment age was not included in the CP model since it was singular for all quadrangles (i.e., no variance). There are several differences in the regression coefficients between models which are noteworthy. For example, in the MRDP model the coefficient for canal density was seven times higher (+36.91) than in the original model (+4.971). This suggests that canal and spoil indirect impacts in the MRDP are high relative to the whole coastal zone. The regression coefficient for sediment age was also

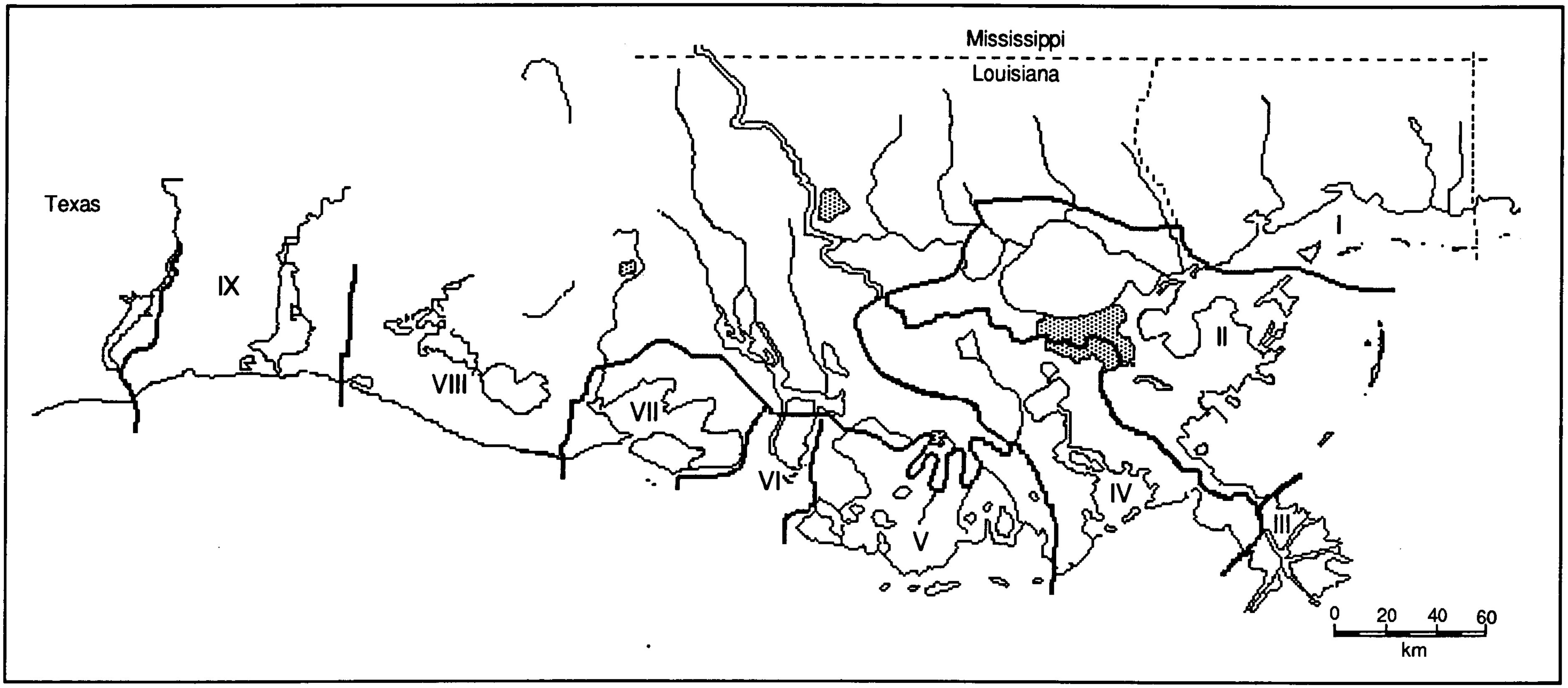

Figure 19-3. Location of the Mississippi River Deltaic Plain region study area. Note that Hydrologic Unit III includes that portion of the Mississippi River between crests of the east and west bank protection levees, and that the coastal zone boundary in Mississippi follows the 5-meter contour line and is only drawn approximately on this map (adapted from Wicker et al., 1980).

larger in the MRDP model, compared with the original analysis (-0.002253 versus -0.000995, respectively); in this case, differences were an order of magnitude. This reflects the relative importance of local geology (e.g, chronology of delta lobe deposition) to land loss in the MRDP. In the CP model, the variables representing both canal density and developed area were highly significant. However, the magnitude of the regression coefficients indicates that the area of urban and agricultural development is important in the CP, compared with the probable role of canals in land loss (+104.42 for DEVDENS vs. +4.70 for CANDENS). The area of marsh that is impounded or semi-impounded for agricultural, urbanization or other purposes in the CP is large (17 to 20% of total area; Cowan et al., 1986, for review) compared to the whole coast or the MRDP (<10%). The impounding of coastal marshes has been implicated in increasing land loss rates within and adjacent to the impounded areas through direct conversion of marsh to upland (agriculture, urban) habitat and effects on local hydrology (Cowan et al., 1986), rather than the desired effect of reducing land loss in the managed area. These regression data bolster this implication, even though we do not suggest that the analysis quantifies the relationship between land loss and impoundment effects in the CP.

The sub-dividends of the whole coast based on hydrologic unit (HU) were, again, analyzed with the linear model described earlier. The results (Table 19-5) provide insight into the behavior of the potential causal factors of land loss in Louisiana in more specific areas along the coast. For example, the magnitude of the significant regression coefficients in the model for HU II (Pontchartrain) (Table 19-5A) suggests that the density of canals and spoil is highly (+) correlated with land loss in that area, relative to the other potential causal factors. Scaife et al. (1983) suggested a similar relationship between canals and land loss, based on data obtained from selected quadrangles in HU II. However, the regression coefficients from the HU IV (Barataria) model (Table 19-5B) show that the geologic variable representing sediment depth is important to account for the land loss in a quadrangle from that region. The regression coefficients from the models for HU IV (Terrebonne) and HUs VI and VII (Atchafalaya and Vermilion; Table 19-4C and 19-4D, respectively) also show important differences. Interpretation of the regression coefficients from these combined analyses implies that the most important factors affecting land loss rates in the Louisiana coastal zone vary depending on location and geologic history, and that the coastal zone is not homogeneous with respect to the potential causal factors or their magnitude. This is not to say, however, that these analyses indicate that any of the causal factors represented by variables included in this linear model do not account for, or contribute to, land loss in the whole coastal zone to some degree because they lack statistical significance. Rather, we believe that the factors influencing land loss are locally variable and complex. More data are needed, perhaps from different sources (e.g, soil types, salinities, sedimentation rates) to precisely quantify and model the factors influencing marsh loss, particularly in areas where this linear model does not perform well (e.g., HUs VI and VII).

Cluster Analysis and Ordination

The combined results from the regression analyses (Table 19-4 and 19-5) imply that considerable local variability exists in the modeled relationships between percent marsh loss in a quadrangle and the factors (predictive variables) that influence that loss. Therefore, we clustered (ordinated) our landloss data to identify quadrangles (areas) along the coast which are similar, i.e., more or less susceptible to land loss, with respect to both landloss rates (PERCENT) and the predictive variables from the regression model. The cluster analysis (CA) created 3 clusters around the mean values that are shown in Table 19-6 following 10 iterations of the original data. The quadrangle map units were then ordinated and placed in one of the clusters based on their nearest centroid distance from the cluster mean for each variable.

Table 19-5. Analysis of regression summary values for a multiple regression model relating marsh loss to natural and man-induced causal influences. Degrees of freedom for model = 4.

	Source	DF	P>F	Estimate	R^2
A. Pontchartrain (HU II)					
	Total	34	0.0001[b]	—	0.78
	Intercept	–	—	17.953	
	CANDENS	1	0.0001[b]	139.179	
	DEVDENS	1	0.0012[b]	-0.942	
	SEDAGE	1	0.3585(NS)	-0.001406	
	DEPTH	1	0.9864(NS)	-0.002038	
B. Barataria (HU IV)					
	Total	23	0.0003[b]	—	0.58
	Intercept	—	—	2.512	
	CANDENS	1	0.1105(NS)	29.653	
	DEVDENS	1	0.5821(NS)	2.505	
	SEDAGE	1	0.327 (NS)	-0.001775	
	DEPTH	1	0.0006[b]	0.377	
C. Terrebonne (HU V)					
	Total	31	0.0055[b]	----0.32	
	Intercept		—	0.996	
	CANDENS	1	0.0139[a]	54.419	
	DEVDENS	1	0.2298(NS)	-15.908	
	SEDAGE	1	0.3729(NS)	-0.0004336	
	DEPTH	1	0.0465(NS)	0.383	
D. Atchafalaya and Vermilion (HU VI and VII)					
	Total	17	0.4047(NS)	—	0.02
	Intercept	—	—	20.969	
	CANDENS	1	0.6683(NS)	5.447	
	DEVDENS	1	0.4799(NS)	4.92	
	SEDAGE	1	0.2912(NS)	-0.0002339	
	DEPTH	1	0.5438(NS)	-0.085	

[a] Statistically significant ($P < 0.05$)
[b] Highly significant ($P < 0.01$)
(NS) Not significant

The results of the ordination (Figure 19-4) agree well with the combined interpretations of the regression analyses (Tables 19-4 and 19-5) and suggest that the Louisiana coastal zone is comprised of several distinct areas relative to land loss and its apparent causes. Cluster 1 is indicative of relatively high canal density and developed area in wetlands which overlie sediments of moderate depth and age. Percent land loss was also moderate in cluster 1 (Table 19-6). Many of the quadrangles in HUs II (Pontchartrain) and IV (Barataria) were placed in cluster 1, which illustrates the complexity of the relationship between land loss and the predictive variables. In HU II, the regression analysis results (Table 19-5A) suggest that canal density and developed area were important factors which correlated with land loss, while sediment depth is most important in HU IV (Table 19-5B). These data, at first, seem to contradict the CA; however, when the magnitude, sign, and significance of each regression coefficient (including the intercepts) for each model are compared, the two analyses agree and aid in interpreting of the CA. In HU II, relatively high canal density has reportedly encouraged (positive regression coefficient = +139.179) moderate to high land loss (intercept = +17.953), while in HU IV lower canal density,

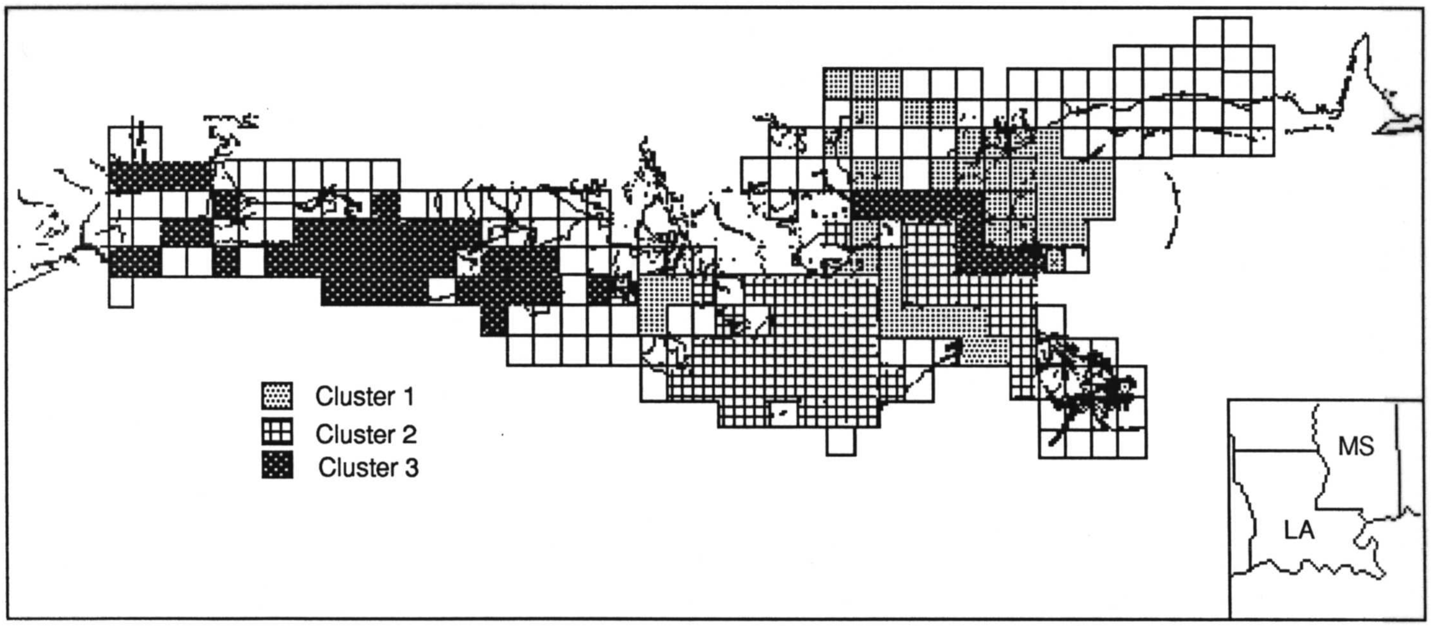

Figure 19-4. Regions of the Louisiana coastal zone determined from the cluster analysis.

combined with the moderating effects of older and more thinly deposited sediments, keeps landloss rates lower (intercept = +2.512). Interpretation of these combined data suggests that the indirect effects of canal density and development in quadrangles ordinated in cluster 1 are lessened by the geology of the area and land loss is moderated.

On the other hand, the apparent effects of canal density and development on land loss in the quadrangles that ordinated with cluster 2 are higher. The cluster mean for PERCENT is highest in cluster 2 while the means for the other variables, including sediment age, were low; the mean of sediment depth was highest for cluster 2. Consequently, it appears that the potential for rapid land loss in the quadrangles of cluster 2 increases dramatically with an increase in man-induced activity (e.g, canals, spoil banks and development); that is, local geology makes the area sensitive. Conversely, land loss in quadrangles in cluster 3 (Figure 19-4) is least influenced by canal density and development because of local geology (old and thin sediments), even though those quadrangles have experienced significant habitat alteration (i.e., cluster means for CANDENS and DEVDENS are highest) and subsequent landloss.

Table 19-6. Summary data from a cluster analysis performed to ordinate the quadrangles based on land loss rate (PERCENT) and the predictive variables from the multiple regression analyses. The number of quadrangles in each cluster are in parentheses.

	Frequency and Cluster Means		
Variable	Cluster 1 (47)	Cluster 2 (64)	Cluster 3 (55)
PERCENT	22.00	36.00	20.00
CANDENS	0.60	0.11	2.25
DEVDENS	1.90	0.17	2.42
SEDAGE	2144.91	516.36	4124.02
DEPTH	27.43	67.98	16.40

Summary And Conclusions

Interpretation of the combined results from three quantitative analyses (principal components analysis, multiple regression, and cluster analysis) suggests the following conclusions:

(1) The complex and regional differences in landloss rates reflect variations in geology and the delta cycle (sediment age and deposition depth over the Pleistocene Terrace), man-induced changes in hydrology (principally canal dredging and spoil banking), and land-use changes (principally urbanization and agricultural expansion). Interpretation of the results of principal components and regression analyses suggests that the most important factors (represented here by the predictive variables) that are correlated with changes in land loss rates in the Louisiana coastal zone vary depending on location and geologic history, and that the coastal zone is not homogeneous with respect to causal factors or their magnitude. These analyses also indicate that each of the causal factors represented by variables included in this study probably contribute to land loss in the whole coastal zone to some degree. However, the data also indicate that the interaction between causal factors is locally variable and complex.

(2) The relationship between land loss, hydrologic changes, and geology can be described with statistically meaningful results, even though these data are insufficient to precisely quantify the relationship. However, these data support the hypothesis that the indirect impacts of man-induced changes (hydrologic and landuse) may be as influential as the direct impacts of converting wetlands to open water (canals) or modified (impounded) habitat (Turner, 1985). For example, the mean land loss in all quadrangles used in this analysis was 23.5%. By interpolating with the regression coefficients obtained from this analysis, a 50% reduction in canal density would result in a nearly 10% decrease in land loss (x = 21.5%), while the direct impacts of canal and spoil account for only 8.0% (≈ 23,000 of 288,414 ha) of the marsh loss (at zero canal density, land loss is reduced by 10.3% ; Table 19-7). If the direct impacts of canal, spoil, and development are eliminated by interpolation, marsh loss is reduced by nearly 20% while their direct impacts account for only 15% of that loss. These back-calculations are based on interpolation, however, and care must be exercised during interpretation of results.

Table 19-7. Estimates of wetland loss, by region, at zero canal and spoil density, based on interpolation in the regression equations obtained in Tables 19-5 and 19-6. Care must be exercised, however, when interpreting the results of back-calculation.

Spatial Unit	Model R^2	Wetland loss (%) Currently 1955/6-1978	Zero Canal and Spoil	Reduction
Louisiana Coastal Zone	0.40	23.5	21.1	10.3
Deltaic Plain	0.46	23.9	20.2	15.3
Chenier Plain	0.58	22.2	10.8	51.3
HU II	0.78	23.7	13.0	45.1
HU IV	0.58	25.0	24.1	4.0
HU V	0.32		not calculated	
HU VI and VII	0.02		not calculated	

(3) Three regions within the Louisiana coastal zone can be defined, based on the potential causal factors used in the linear regression model. The moderate (mean = 22%) wetland loss rates in region 1 (cluster 1) are a result of relatively high canal density and developed area in marshes that overlie sediments of moderate age and depth; local geology acts, in this case, to lessen indirect impacts. On the other hand, landloss rates in region 2 are high (mean = 36%) despite fewer man-induced impacts; the potential for increased land loss due to both direct and indirect effects of man's activity in these areas is high. Conversely, land loss (mean = 20%) in region 3 is apparently least influenced by man's activity in the coastal zone because of sedimentary geology (old, thin sediments), even though these areas have experienced significant habitat alterations and land loss.

Chapter 20

SPATIAL ANALYSIS OF LOUISIANA COASTAL LAND LOSS

by

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Introduction

As part of the overall MMS study, three areas were analyzed using high resolution (10 m x 10 m) digital imagery. The general goal was to determine what factors, both qualitative and quantitative, are correlated with land loss by using a spatial database with a scale intermediate to those used in Chapters 19 and 21. A sub-goal was to test whether two specific factors (saltwater intrusion and effects of canal and spoil) were related to land loss, since these have both been discussed extensively in the literature (e.g., Deegan et al., 1984; Leibowitz et al., 1987; Mendelssohn et al., 1983; Scaife et al., 1983; Turner et al., 1984).

The qualitative variables (site, habitat type, and change in salinity) are all broad scale environmental factors. The quantitative variables that were chosen include 6 distance (nearness of a point of land to a particular spatial feature, such as canal and spoil) and 12 density (area of a feature, such as agriculture, per unit land area) measures. These factors represent both human activities and natural processes.

Database Description and Methods

Three study areas in southern Louisiana were chosen for spatial analyses (Figure 1-1). The Cameron site is the westernmost of the study areas, in a portion of the state known as the Chenier Plain. This region consists of a series of stranded beach ridges (Gould and McFarlan, 1959). It is outside the direct influence of the Mississippi River, and therefore receives sediments either indirectly from the Mississippi (e.g., through longshore currents) or from local rivers, such as the Sabine or Calcasieu. The Cameron site is bordered on the west by Lake Calcasieu. The Calcasieu Ship Channel, built in 1941, runs through this lake. The Cameron site had a land area of 448 km^2 in 1956.

The Terrebonne site, in western Terrebonne parish, contains the Atchafalaya River. This is the newest of the Mississippi River distributaries and currently receives 30% of the total Mississippi flow. This new source of sediments led to the emergence of the Atchafalaya delta in 1973 (van Heerden, 1983). The Terrebonne site had a total land area of 894 km^2 in 1956.

The Lafourche site is the easternmost area, bordered on the east by Barataria Bay and on the west by Bayou Lafourche, the most recently abandoned of the Mississippi River distributaries (Frazier 1967; Morgan 1972). In 1956, this site had a total land area of 832 km^2.

The three study areas were chosen to represent three of the different coastal geological environments in Louisiana: chenier; emerging river/delta; and, abandoned river/delta. Together, they have a total 2,174 km^2 of land area, representing 12.5% of the land area in the Louisiana coastal zone.

Digital habitat maps for 1956* and 1978 were obtained from the Coastal Management Division of the Louisiana Department of Natural Resources for the three sites. This imagery was derived from habitat maps of the Louisiana coast produced by Coastal Environments, Inc., of Baton Rouge (Wicker, 1980, 1981). The habitat maps were originally prepared at a scale of 1:24,000 using black and white (1956) or color infrared (1978) aerial photography and were then digitized by the U.S. Fish and Wildlife Service (USFWS). These data were reformatted to a grid of 10 m x 10 m cells for the present study.

The original habitat maps were coded using the Cowardin classification system (Cowardin et al., 1979). These codes were then aggregated into nine land and four aquatic habitat types (Leibowitz et al., 1987): agriculture, beach/dune/reef, fresh marsh, mudflat, saline marsh, spoil, swamp, upland, urban/industrial, canal, coastal open water, inland open water,** and natural channel. The habitat maps also contained boundaries between saline and freshwater zones. These boundaries were based on vegetation maps published by O'Neil (1949) and Chabreck and Linscombe (1978). Thus, vegetation type was used as an indication of ambient salinity levels (e.g., *Spartina alterniflora* is an indicator of high salinity, while the presence of *Sagittaria falcata* indicates low salinity). Each cell in the database was classified according to its salinity change between 1956 and 1978. Four categories result: areas that were fresh in 1956 and fresh in 1978; areas that were fresh in 1956, but saline in 1978; areas that changed from saline to fresh between 1956 and 1978; and, areas that remained saline during the 22-year duration (Figure 20-1).

For this study, land loss was defined as a cell that changed from land to water between 1956 and 1978 (Figure 20-2). This therefore represents a gross loss, since change from water to land is not included (this definition differs from that used in Chapter 19, which defines *marsh loss* as the net change in marsh area between 1956 and 1978). Two different measures of land loss were considered. The rate of conversion from land to water within a category (R_c) is defined as:

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* Aerial photography of the Louisiana coast was obtained during the 1955-1956 period (Wicker, 1980, 1981). Both 1955 and 1956 have been used in the literature in referring to this data set. The later date is used here because 23 out of the 31 maps used for the specific study areas were from 1956 photography.

** The open water category was divided into coastal and inland open water to distinguish between true shoreline erosion (loss of land at the barrier islands and southernmost boundaries by storm erosion, sea level rise, etc.) and breakup of interior wetland. For the Cameron and Terrebonne study sites, identification of inland water was straightforward, since these sites are mostly composed of contiguous land. Thus, inland open water refers to ponds or lakes, whereas coastal open water refers to either Atchafalaya Bay, Four League Bay or the Gulf. This distinction was more difficult to make for the Lafourche study site, however, since its eastern boundary (Barataria Bay) is in a stage of advanced breakup. The distinction between inland ponds or lakes and coastal bays is therefore fuzzy. It was decided to classify Barataria Bay as inland open water for the Lafourche study site, whereas coastal open water would only consist of Gulf of Mexico waters.

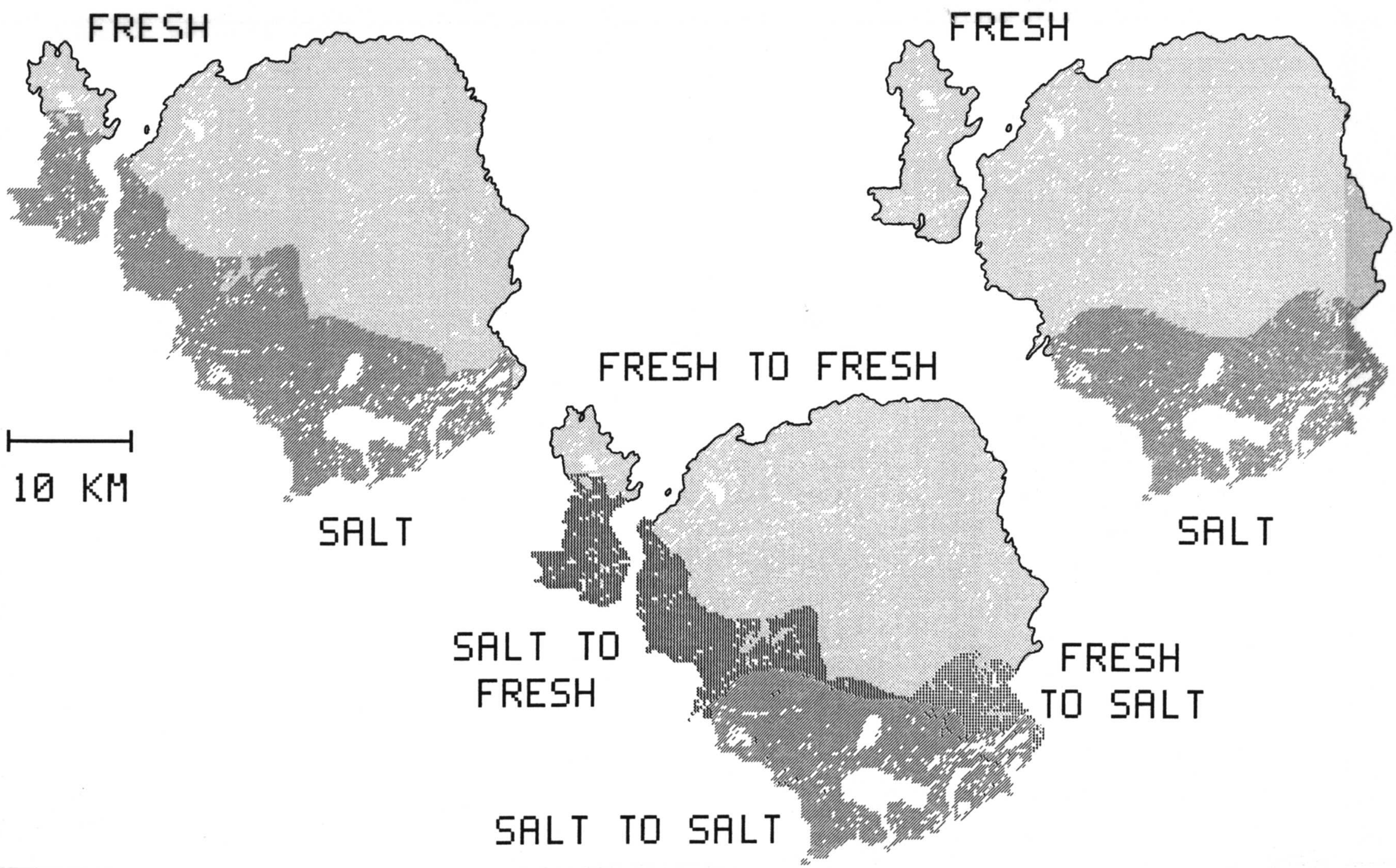

Figure 20-1. Illustration of salinity change zones for the Terrebonne study site. Upper left shows the boundary between fresh and saline vegetation in 1956. The same is shown in the upper right for 1978. This information is combined to show the four salinity change zones for the period 1956 to 1978 (lower figure).

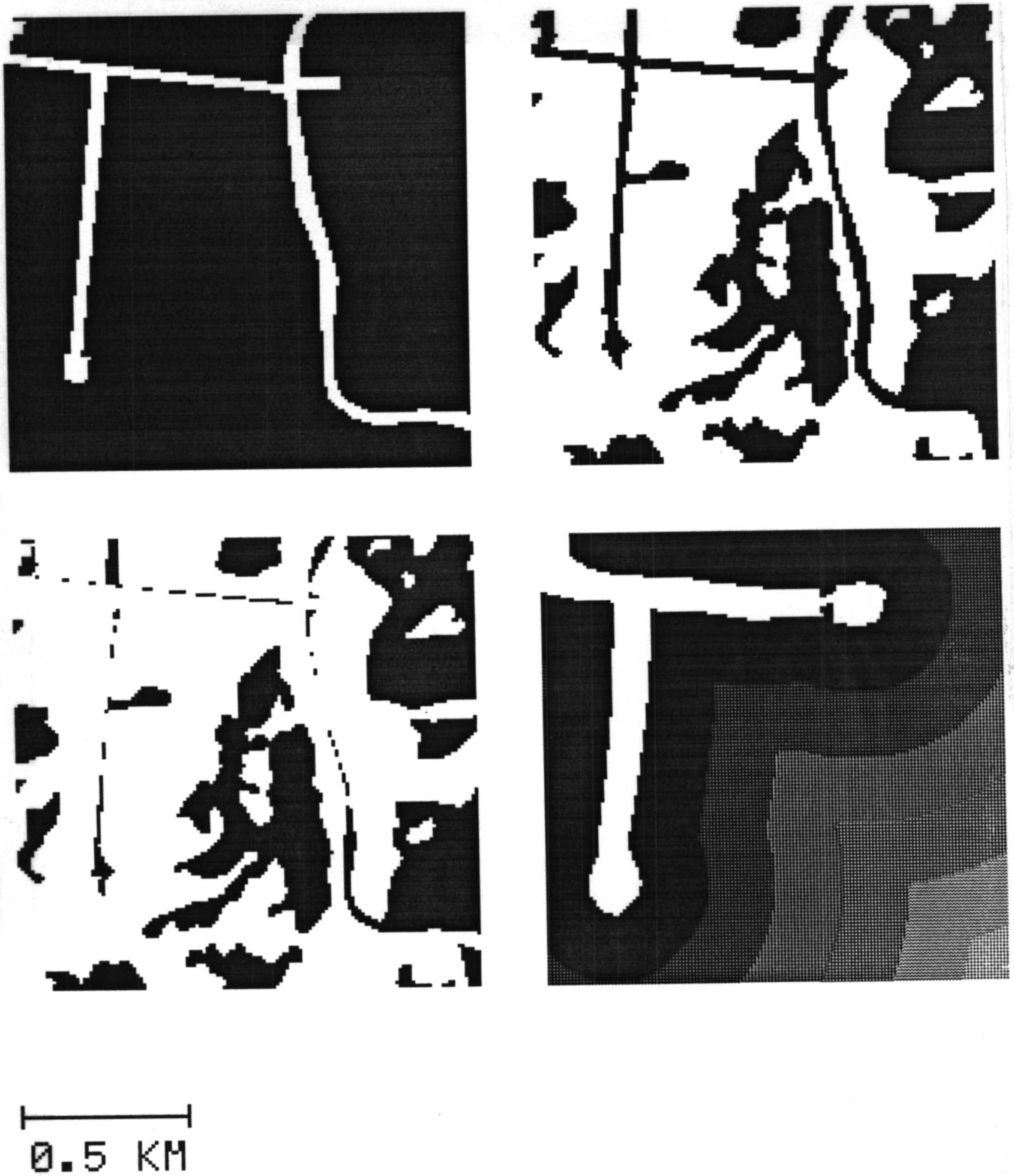

Figure 20-2. Illustration of land loss and distance intervals. Upper left shows land in 1956 for a sub-section of the Terrebonne study site. Upper right shows water in the same area for 1978. Land loss is defined as all areas that were land in 1956 and water in 1978 (dark regions in lower left). The lower right is an example of how equal distance contours are "grown" around a feature, in this case 1956 canal or spoil. Land falling within an interval (shown in 200 m contours) is the same distance to the nearest occurrence of that feature (e.g., all land in the darkest contour is 200 m from the nearest 1956 canal or spoil).

$$R_c = 100 \times L_c / N_c \qquad \text{(Eqn. 20.1)}$$

where R_c = the percent conversion of land to water within category c;
L_c = the amount of land that converted to water within category c between 1956-1978;
N_c = the total amount of land in category c in 1956.

The second measure is the proportion of loss accounted for by categories within a factor (P_c):

$$P_c = 100 \times L_c / L_t \qquad \text{(Eqn. 20.2)}$$

where P_c = the percent of all land loss that occurred within category c;
L_c = the amount of land that converted to water within category c between 1956-1978;
L_t = the total amount of land that converted to water between 1956-1978, where $L_t = L_{c1} + L_{c2} + ... + L_{cn}$.

Since the proportions in Equation 20.1 are rates within categories, they do not sum to 100% (i.e., $R_{c1} + R_{c2} + ... + R_{cn} \neq 100$). The values in Equation 20.2 are true percentages, however, and therefore add to 100 (i.e., $P_{c1} + P_{c2} + ... + P_{cn} = 100$).

The effect of five major factors on land loss was investigated: site, habitat type, salinity change, and several associated distance and density factors. Distance is a measure of the proximity of a cell to the nearest occurrence of a feature of interest, obtained by "growing" equal distance contours around the feature of interest (Figure 20-2). If a feature, such as canal and spoil, contributes to land loss, rates of loss should be greater near the feature than at a distance (i.e., $R_{c1} > R_{c2}$, where c_1 and c_2 are distance intervals in km, and $c_1 < c_2$). The converse would indicate that the factor helps alleviate land loss. Six distance factors were studied: 1956 canal and spoil; 1978 canal and spoil;* 1956 natural channel (rivers, streams, and bayous); 1956 major channel (Lake Calcasieu/Calcasieu Ship Channel, the Atchafalaya River, and Bayou Lafourche, respectively, for the Cameron, Terrebonne, and Lafourche study sites); 1956 open water (lakes, ponds, and Barataria Bay); and 1956 coastal shore.

Another measure of a spatial impact is the density of a feature. A square-shaped, 1 km x 1 km window was centered around each cell. The density of a feature for that cell was then defined as the area of that feature within the 1 km^2 window divided by the 1956 land area in that square.

$$D_{cx} = F_{cx} / N_x \qquad \text{(Eqn. 20.3)}$$

where D_{cx} = the density of category c for the square km window surrounding cell x;

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* A canal could have been built before the acquisition of the 1956 photography, in which case it would show up in both the 1956 and 1978 imagery; or the canal could have been constructed between the 1956 and 1978 photo dates, in which case it would only be seen on the 1978 imagery. Using the 1956 imagery to assess canal impact under-estimates their effects, since canals constructed anytime after the 1956 photographic acquisition are not included. If the 1978 imagery is used, the effects are over-estimated, since any long-term effects may have not had time to be expressed. These two can be used together, however, to place upper and lower bounds on canal impacts.

F_{cx} = the area of category c within the square km surrounding cell x;
N_x = the total amount of land within the cell x window in 1956.

Therefore, D_{cx} is a unitless variable and is similar to the density used in Chapter 19 (in Chapter 19, however, the denominator is marsh land, rather than total land). Six density factors were studied: canal, spoil, canal plus spoil, agriculture, urban/industrial, and agriculture plus urban/industrial, each for 1956 and 1978.

Results in this study are based upon either the complete database, designated as FULL, or two sub-sampled databases, POINT and WINDOW. The FULL database represents the entire digital database, and contains over 28.5 million cells. Results using these data represent a complete census of the three study areas, within the resolution of the 10 m x 10 m cells. The POINT database represents a sample of 25,000 randomly chosen cells (5,000 from the Cameron site and 10,000 each from the Terrebonne and Lafourche sites, representing 0.22, 0.11, and 0.12% of their total land area, respectively). The criterion for selection was that the cell had to be land in 1956 (this was to avoid selecting cells offshore or in the middle of a bay). Each POINT cell was classified by site, 1956 habitat type, salinity change, and the six distance factors. Loss was defined for these cells as either 0 or 100%, depending on whether the cell remained land or had converted to water by 1978. The WINDOW database included all of the POINT information, plus the 12 density measurements. For WINDOW data, loss was calculated as the rate of land conversion to water within the 1 km^2 window:

$$R_x = 100 \times L_x / N_x \qquad \text{(Eqn. 20.4)}$$

where R_x = the loss rate within the square km window surrounding cell x;
L_x = the amount of land that converted to water between 1956-1978 within the square km surrounding cell x;
N_x = the total amount of land within the cell x window in 1956.

An additional constraint in selecting for POINT and WINDOW samples was that the window had to lie completely within the study area boundary, and had to contain at least 25 land cells (this was to remove edge effects and to ensure that the window was not completely open water). The reduced data sets contained 23,361 samples. The FULL database was constructed and analyzed using the Earth Resources Laboratory Applications Software (ELAS) developed by NASA/Earth Resources Laboratory (NASA, 1986) and modified by the Remote Sensing and Image Processing Laboratory at Louisiana State University. POINT and WINDOW data were analyzed with the SAS statistical package (SAS Institute, Inc., 1985 a,b,c).

Effect of Site, Habitat Type, and Salinity Change on Land Loss

The three study sites are shown as they appeared in 1956 in Figure 20-3. Table 20-1 shows that these areas underwent extensive habitat alteration in the period between 1956 and 1978; in addition, all three sites experienced considerable land loss (Figure 20-4). Conversion rates (R_{site}) from land to water were 16.9, 14.4, and 20.8% for the Cameron, Terrebonne, and Lafourche study sites, respectively (Table 20-2). The fresh and saline marsh habitats accounted for 95.5, 93.6, and 85.5% of all land loss (P_{marsh}) at the Cameron, Terrebonne, and Lafourche study sites, respectively (Table 20-3). Each other habitat accounted for 0.0-3.8% of the loss at a site, with the exception of agriculture at the Lafourche study site (7.5%).

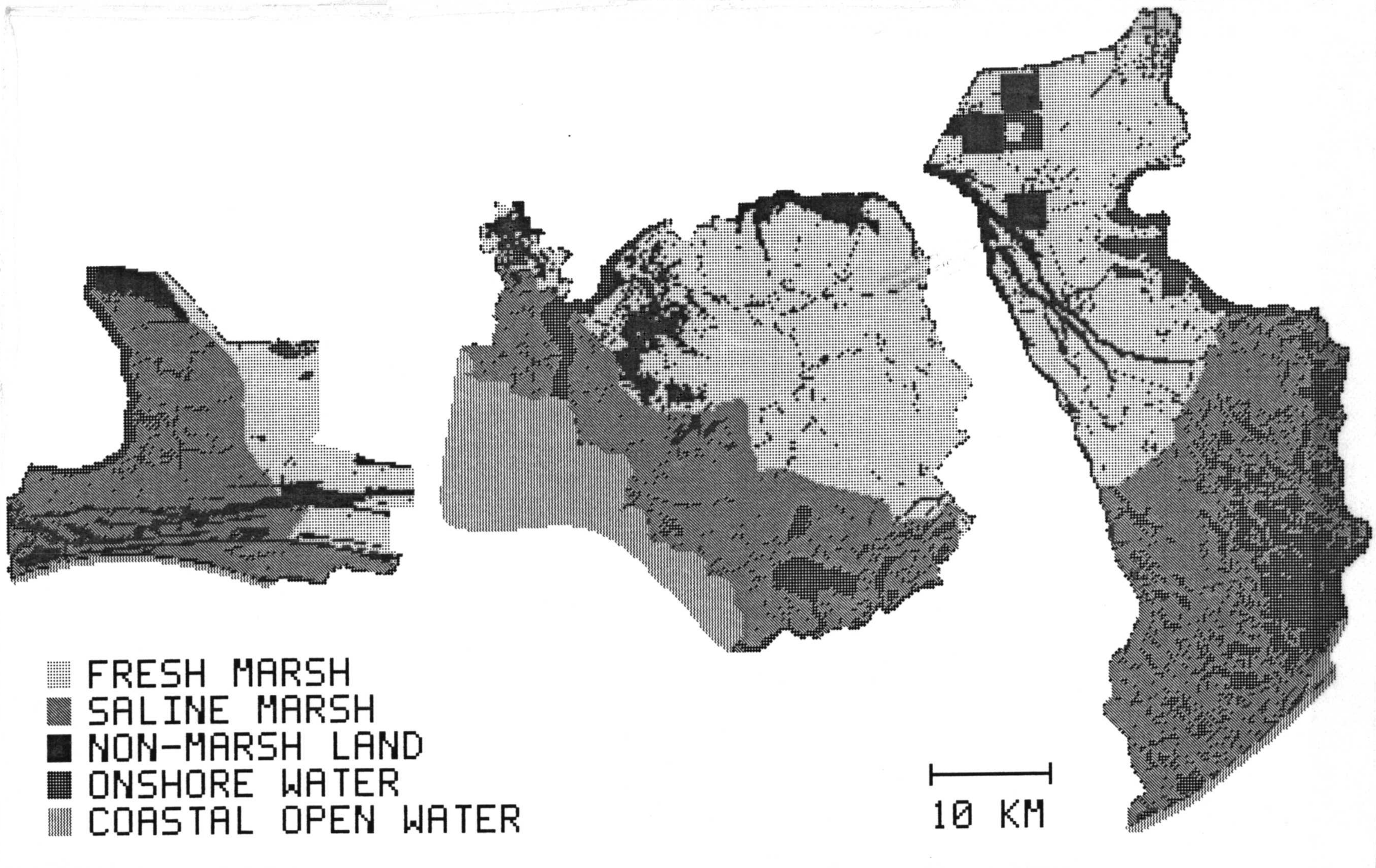

Figure 20-3. Original (1956) habitat distribution for the Cameron (left), Terrebonne (center), and Lafourche (right) study sites.

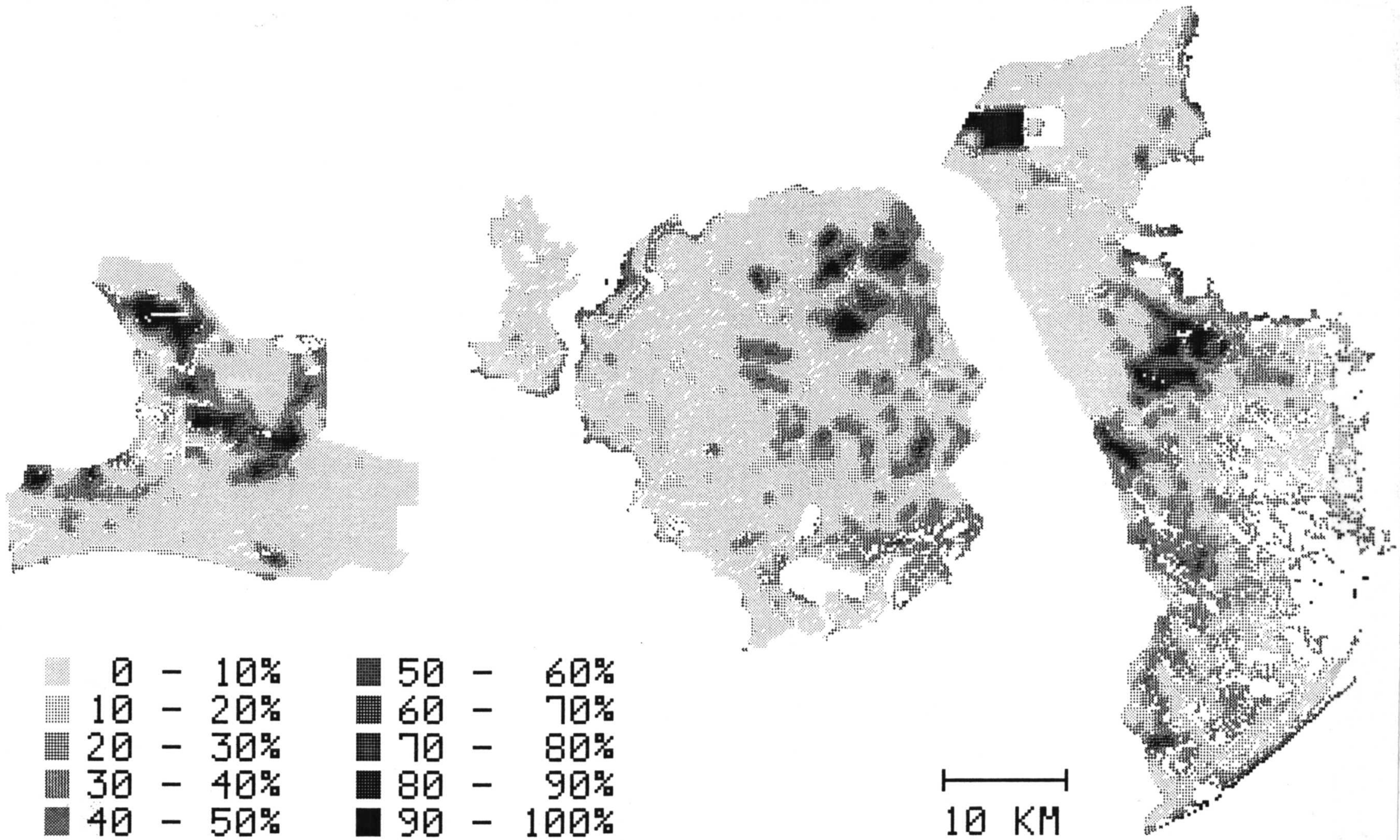

Figure 20-4. Landloss trends (1956 to 1978) for the Cameron (right), Terrebonne (center), and Lafourche (right) study sites (percent land loss per km^2).

Table 20-1. Habitat distribution for the Cameron, Terrebonne, and Lafourche study sites (1956 and 1978). Values are based on the complete digital database (FULL).

Area (km^2):	Cameron 1956	Cameron 1978	Terrebonne 1956	Terrebonne 1978	Lafourche 1956	Lafourche 1978
Habitat						
Land:						
Agriculture	72.65	72.81	0.00	0.00	41.01	53.93
Beach/Dune/Reef	1.96	1.25	0.03	0.22	2.83	1.56
Fresh Marsh	115.82	38.40	475.95	475.38	368.35	50.12
Mudflat	1.98	3.99	3.31	7.29	0.47	0.00
Saline Marsh	244.41	243.15	325.25	198.98	366.37	513.76
Spoil	1.92	6.36	7.26	32.26	17.65	31.65
Swamp	0.00	0.00	80.25	78.59	29.09	22.62
Upland	2.54	3.14	0.00	0.05	0.49	0.26
Urban/Industrial	7.06	8.94	1.62	0.63	5.70	10.02
Total Land	448.34	378.04	893.67	793.40	831.96	683.92
Net Land Loss		70.30		100.27		148.04
Water:						
Canal/Pipeline	3.69	6.32	7.79	24.57	16.29	38.98
Coastal Open Water	11.77	13.52	199.19	186.97	19.26	25.86
Inland Open Water	35.54	103.26	45.49	132.31	254.54	370.72
Natural Channel	9.16	7.37	56.53	65.43	18.49	21.04
Total Water	60.16	130.47	309.00	409.28	308.58	456.60
Net Water Gain		70.31		100.28		148.02

Table 20-2. Gross landloss rates (1956-1978) at the Cameron, Terrebonne, and Lafourche study sites.

Site	1956 Land Area (km^2)	Land Loss (km^2)	Gross Loss Rate
Cameron	448.34	75.95	16.94
Terrebonne	893.67	129.06	14.44
Lafourche	831.96	173.22	20.82

Note: Rates are defined as the gross conversion of land to water between 1956 and 1978 (Rs = 100 x Ls / Ns, where Rs is the percent conversion of land to water at site S, Ls is the amount of land at site s that converted to water between 1956-1978, and Ns is the total amount of land at site s in 1956). Values are based on the complete digital database (FULL).

Conversion of land to inland open water (P_{open}) accounted for 69.7 to 92.8% of all land loss (Table 20-4). Transformation of land to water through new canal construction or canal widening (direct impacts of canals, P_{canal}) accounted for 2.5, 14.4, and 15.3% of all land loss at the Cameron, Terrebonne, and Lafourche study sites, respectively. Ranges for conversion of land to natural channel ($P_{channel}$) were low, except at the Terrebonne site: 2.1, 13.8, and 5.4%, respectively. For the three sites, loss of land from shoreline erosion (P_{shore}) accounted for only 2.6, 2.1, and 3.2% of all loss. Thus, the major form of land loss for all three regions is the conversion of land to inland open water (lakes, ponds or bays).

Table 20-3. Characterization of land loss by original land habitat (numbers may not add due to rounding).

	Cameron Land Loss (km²)	Cameron %	Terrebonne Land Loss (km²)	Terrebonne %	Lafourche Land Loss (km²)	Lafourche %
Habitat						
Agriculture	0.94	1.23	0.00	0.00	12.98	7.49
Beach/Dune/Reef	0.32	0.43	0.03	0.02	2.76	1.60
Fresh Marsh	18.46	24.30	84.62	65.56	68.06	39.29
Mudflat	1.63	2.14	2.50	1.94	0.16	0.09
Saline Marsh	54.09	71.23	36.16	28.02	80.10	46.25
Spoil	0.19	0.25	1.86	1.44	6.50	3.76
Swamp	0.00	0.00	3.88	3.01	2.04	1.18
Upland	0.01	0.01	0.00	0.00	0.09	0.05
Urban/Industrial	0.31	0.41	0.01	0.01	0.52	0.30
Total	75.95	100.00	129.06	100.00	173.22	100.00
Total Marsh	72.55	95.52	120.78	93.58	148.16	85.53
Total Wetland	74.18	97.67	127.16	98.53	150.36	86.80

Note: Percentages are the proportion of all loss accounted for by a habitat type within each site ($Pc = 100 \times Lc / Lt$ where Pc is the percent of loss for habitat c, Lc is the gross change from habitat c to water between 1956-1978, and Lt is the total conversion from land to water at that site). The marsh category includes fresh and saline marsh, and wetland is comprised of marsh, mudflat, and swamp. Values are based on the complete digital database (FULL).

Table 20-4. Characterization of landloss conversion by final aquatic habitat (numbers may not add due to rounding).

	Cameron Land Loss (km²)	Cameron %	Terrebonne Land Loss (km²)	Terrebonne %	Lafourche Land Loss (km²)	Lafourche %
Canal	1.91	2.51	18.52	14.35	26.50	15.30
Coastal Open Water	1.95	2.57	2.72	2.11	5.63	3.25
Inland Open Water	70.48	92.80	89.98	69.72	131.68	76.02
Natural Channel	1.61	2.13	17.85	13.83	9.40	5.43
Total	75.95	100.00	129.06	100.00	173.22	100.00

Note: Percentages are the proportion of land that converted to a particular water class within each site ($Pc = 100 \times Lc / Lt$ where Pc is the percent of loss for water body c, Lc is the gross change from land to water body c between 1956-1978, and Lt is the total conversion from land to water at that site). Values are based on the complete digital database (FULL).

Loss rates between habitat types ($R_{habitat}$) were not uniform: values ranged from 0.0% for uplands to 55.1% for mudflat. Habitats could be divided into either two or three groupings (Table 20-5). Rates for uplands, urban/industrial, swamp, agriculture, saline marsh, and fresh marsh were significantly lower ($P < 0.05$) than rates for spoil, beach/dune/reef, and mudflat. Loss rates for the first group ranged from 0.0 to 18.0% while the range for the second group was 36.2 to 55.1%. It should be pointed out, however, that habitats in the latter group are narrow and, therefore, prone to mis-registration of imagery between years. In addition, mudflat is particularly susceptible to mis-classification from tidal effects. Therefore, these higher rates may be artifacts.

Table 20-5. Habitat groupings for the three combined study areas based on landloss rates.

Habitat	Mean	Number	Grouping
Mudflat	55.10	49	A
Beach/Dune/Reef	46.67	30	A
Spoil	36.15	213	A B
Fresh Marsh	18.00	10,447	B C
Saline Marsh	17.75	10,256	B C
Agriculture	12.10	1,141	C
Swamp	5.12	1,074	C
Urban/Industrial	2.63	114	C
Uplands	0.00	37	C

Note: Mean is the mean loss rate (R) for each habitat type, based on the 23,361 samples of the POINT database, and number is the number of samples within that category. Means with the same letter are not significantly different at the 95% confidence level. Significance is based on Scheffe's multiple-comparison procedure, using the following model: Land Loss = Site + 1956 Habitat Type + Salinity Change.

Loss rates for the saline marsh, fresh marsh, and spoil habitats were significantly higher (17.8 to 36.2%) than rates for the upland, urban/industrial, swamp, and agricultural habitats (0.0 to 12.1%). These rates were also significantly lower than values for the beach/dune/reef and mudflat habitats (46.7 to 55.1%). Therefore, the terrestrial habitats can be aggregated into three basic categories: the highest rates of land loss are found on the mudflat and beach/dune/reef habitats, both of which are at the land/water interface and are most directly exposed to erosive forces. Because these are narrow habitats, however, they are also most prone to classification errors. Uplands, urban/industrial, swamp, and agriculture are terrestrial habitats that occur at higher elevations on more stable soils, and they have the lowest rates of land loss. Marsh and spoil habitats are found at lower elevations on less stable soils and have intermediate values.

To assess the importance of saltwater intrusion to land loss, we hypothesized that if saltwater intrusion were a major cause of land loss, then rates of loss would be higher in areas that went from fresh-to-salt or were saline originally.* Observing this would not prove that saltwater intrusion caused the land loss, however, since increased salinity and higher loss could both be caused by some third factor, such as subsidence or sea level rise. The opposite finding, that rates are high in the salt-to-fresh and fresh-to-fresh categories, would indicate that factors other than saltwater intrusion were responsible for the land loss.

The effect of salinity change on land loss ($R_{salinity}$) exhibits several trends. For the Cameron and Lafourche study sites, there was no significant difference ($P < 0.1$) between the fresh-to-salt or salt-to-salt categories, with values ranging from 21.6 to 26.7% (Table

* It was not possible to distinguish between intermediate, brackish, and salt marsh with the 1956 black and white photography (Wicker, 1980, 1981). Thus, these three vegetation types were all grouped as saline. In an area where saltwater intrusion was occurring, however, increasing salinity would take place in the salt-to-salt, as well as fresh-to-salt categories.

20-6). There was also no difference between rates in the fresh-to-fresh or salt-to-fresh categories at these two areas, with values of 0.0 to 2.2%. The rates for the former pair were significantly greater than the latter two, however. Thus, there is no evidence to disprove saltwater intrusion as a major factor for these two areas (it must be stressed again that this does not prove that saltwater intrusion is the causal factor).

Table 20-6. Salinity change groupings based on landloss rates.

	Cameron			Terrebonne			Lafourche		
	Mean	No.	Group	Mean	No.	Group	Mean	No.	Group
Salinity Class									
Fresh-to-Salt	21.56	937	A B	15.52	348	B C D	26.74	3,650	A
Salt-to-Salt	24.10	2,469	A B	12.45	2,065	C D	22.29	4,007	A B
Fresh-to-Fresh	0.19	1,038	E	16.18	5,569	B C	2.19	1,509	E
Salt-to-Fresh	0.00	168	E	7.10	1,437	D E	0.00	164	E

Note: Mean is the mean loss rate (R) within each salinity class, based on the 23,361 samples of the POINT database, and number is the number of samples within that category. Means with the same letter are not significantly different at the 90% confidence level, e.g., values for Cameron fresh to salt, Cameron salt to salt, Lafourche fresh to salt, and Lafourche salt to salt are not significantly different. Significance is based on Scheffe's multiple-comparison procedure, using the following model: Loss = Site + 1956 Habitat Type + Salinity Change.

At the Terrebonne study site, land loss was highest in the fresh-to-fresh category (16.2%). There was no significant difference between salt-to-salt, fresh-to-salt, and fresh-to-fresh at this site (12.4 to 16.2%). Further, values for this group were significantly lower than the fresh-to-salt and salt-to-salt categories at the Cameron and Lafourche study sites. The fact that there is no difference at the Terrebonne site between the rate in areas that remained fresh versus those that stayed saline or went from fresh to saline, plus the fact that these rates are lower than at the other two sites, lead to the conclusion that saltwater intrusion is not the major cause of land loss at the Terrebonne site. The pattern of land loss in this area is inconsistent with the saltwater intrusion hypothesis.

The findings thus far are consistent with what would be expected based on the geology of these areas. Those habitats that are found on more stable sediments (uplands, urban/industrial, swamp, and agriculture) have the lowest loss rates. The Terrebonne site, which has the Atchafalaya River as a major sediment source, had loss rates significantly lower than the other two areas. In addition, it is the only site where large areas changed from saline marsh to fresh marsh. This area, which is becoming less saline, does not appear to be affected by saltwater intrusion. The Cameron and Lafourche sites, however, do not presently have a major sediment source and have undergone conversion of large areas from fresh to saline marsh. Saltwater intrusion and land loss both occur at these two sites. The question to be considered next is how does a human activity, such as construction of canal and spoil, affect landloss rates? It has already been shown (Table 20-4 and Chapter 4) that the direct effect of canals (construction and widening) is significant. In the next section we will mostly be concerned with the indirect effects that canals may have (e.g., hydrologic alteration).

Effects of Distance and Density Factors on Land Loss

For each distance or density feature, the loss rate was modeled as a linear function of site, the feature, and their interaction:

$$R_f = \text{Site} + D_f + D_f \times \text{Site} \qquad \text{(Eqn. 20.5)}$$

where R_f is the loss rate for factor f within a 1 km^2 window for each of the 23,361 WINDOW samples, Site is either the Cameron, Terrebonne or Lafourche study site, and D_f is the distance or density measure for factor f.

All six distance models were highly significant, with $P < 0.0001$ (Table 20-7). However, it is concluded that the statistical significance is the result of the large number of samples and not because of any real ecological effects, since all factors had low R^2 values (0.0261 to 0.1086). Therefore, only a small amount of the variation in land loss was accounted for by any one of these factors. In addition, trends between sites or between years were not consistent. For example, loss rates increased with proximity to 1956 canal and spoil at the Cameron and Terrebonne study sites but decreased at the Lafourche site. However, rates at all three sites decreased with proximity to 1978 canal and spoil.

Table 20-7. Model results for effects of distance factors on land loss.

Factor/Source	DF	PR > F	R^2
1956 Canal and Spoil		0.0001	0.0312
Site	2	0.0067	
Distance	1	0.2227	
Distance x Site	2	0.0001	
1978 Canal and Spoil		0.0001	0.0261
Site	2	0.0001	
Distance	1	0.0001	
Distance x Site	2	0.0001	
1956 Natural Channel		0.0001	0.0650
Site	2	0.0001	
Distance	1	0.0001	
Distance x Site	2	0.0001	
1956 Major Channel		0.0001	0.0595
Site	2	0.0001	
Distance	1	0.0001	
Distance x Site	2	0.0001	
1956 Inland Open Water		0.0001	0.0524
Site	2	0.0001	
Distance	1	0.0001	
Distance x Site	2	0.0001	
1956 Coastal Shore		0.0001	0.1086
Site	2	0.0001	
Distance	1	0.0001	
Distance x Site	2	0.0001	

Note: For each factor (e.g., 1956 canal and spoil), land loss was modeled as Land Loss = Site + Distance + Distance x Site, where loss is the loss rate (R) for each of 23,361 WINDOW samples; Site is either the Cameron, Terrebonne, or Lafourche study site; distance is the distance from the center of the square km window to the nearest occurence of that factor; and distance x site is the distance/site interaction. DF are the degrees of freedom for each term in the model, PR>F is the significance either of the model or each of the terms (based on the type III sum of squares), and the remaining column is the R^2 value for that model.

Results for the density factors were generally similar to those for the distance data. In all cases, model results were highly significant, with $P < 0.0001$ (Table 20-8). The amount of the variance accounted for was similarly low, however, with R^2 values ranging from 0.0202 to 0.0882.

The analyses of distance and density factors show that, for the entire range of the data, these features account for only a small portion of the variation in land loss. It is possible, however, that these features affect land loss but only over a portion of their range. Distances for 1956 canal and spoil, for example, range from 0 to 6.5 km and in the regression a straight line is fit over this entire interval. Canal and spoil could have significant effects on land loss only over a portion of this distance. To measure this impact, we calculated the cumulative percent of total loss (P) with respect to a distance or density factor. To illustrate the difference between this and the regressions of rate (R), consider the loss rate for mudflat at the Terrebonne study site. In 1956, this habitat had a total area of 3.3 km^2 (Table 20-1); by 1978, 2.5 km^2 of the original area had converted to open water (Table 20-3). Thus, based on Equation 20.1, the loss rate for mudflat was $R_{mudflat} = 100 \times 2.5 / 3.3 = 75.6\%$. Although this rate is high, it accounts for a small portion of the total 129.1 km^2 land loss at the Terrebonne study site between 1956 and 1978 ($P_{mudflat} = 100 \times 2.5 / 129.1 = 1.9\%$).

Similarly, the overall contribution of canals and spoil can be assessed by considering the cumulative amount of land loss versus distance or density. Areas in close proximity to canal and spoil account for less than 5% of all indirect land loss (direct loss to canals is not included); the rate sharply rises, however, so that 90% of all loss occurred within 3.2 to 4.4 , 1.7 to 2.8, and 1.4 to 3.6 km for the Cameron, Terrebonne, and Lafourche study sites, respectively (Figure 20-5). For natural channel, 90% of all loss occurs within 4.5, 2.2, and 3.2 km, respectively. Therefore, while a large amount of land loss may occur in areas near canals or spoil, this response is similar to that found for natural channels. This may indicate that a similar mechanism is responsible for the loss adjacent to these two features (e.g., compaction of sediment caused by spoil or natural levee).

For the Cameron and Lafourche sites, 90% of all land loss occurs within 2.2 and 2.0 km of inland open water. This seems to indicate that much of the loss at these sites is enlargement of ponds and lakes or the breakup of Barataria Bay. For the Terrebonne site, however, less than 90% of all land loss had occurred within 8 km of this feature (Figure 20-6), again highlighting the difference between this site and the other two. This may indicate that land loss at the Terrebonne site is a recently initiated process and not an enlargement of prior loss.

The effect of distance to major channel on cumulative land loss is also shown in Figure 20-6 (note the difference in the distance scale for this feature). The distances at which 90% of all loss occurs are much greater for the major channel than for the nearest channel. For the Cameron and Lafourche sites, 90% of all land loss occurs at distances of 12.6 and 14.3 km, respectively. For the Terrebonne study site, the effect of the Atchafalaya River is even more dramatic: loss rates are greatly reduced to a distance of 12 km, and the 90% loss interval is at a distance of 28.7 km. This underscores the importance of sedimentation in land maintenance.

Table 20-8. Model results for effects of density factors on land loss.

Factor/Source	DF	PR>F	R^2
1956 Canal		0.0001	0.0514
Site	2	0.0001	
Density	1	0.0001	
Density x Site	2	0.0001	
1956 Spoil		0.0001	0.0217
Site	2	0.0001	
Density	1	0.0001	
Density x Site	2	0.0001	
1956 Canal and Spoil		0.0001	0.0357
Site	2	0.0001	
Density	1	0.0001	
Density x Site	2	0.0001	
1956 Agriculture		0.0001	0.0643
Site	2	0.0001	
Density	1	0.0001	
Density x Site	1	0.0001	
1956 Urban/Industrial		0.0001	0.0404
Site	2	0.0001	
Density	1	0.0001	
Density x Site	2	0.0001	
1956 Developed		0.0001	0.0663
Site	2	0.0001	
Density	1	0.2506	
Density x Site	2	0.0001	
1978 Canal		0.0001	0.0210
Site	2	0.0001	
Density	1	0.0001	
Density x Site	2	0.0042	
1978 Spoil		0.0001	0.0206
Site	2	0.0001	
Density	1	0.7421	
Density x Site	2	0.0001	
1978 Canal and Spoil		0.0001	0.0202
Site	2	0.0001	
Density	1	0.0434	
Density x Site	2	0.1871	
1978 Agriculture		0.0001	0.0856
Site	2	0.0001	
Density	1	0.0001	
Density x Site	1	0.0002	
1978 Urban/Industrial		0.0001	0.0443
Site	2	0.0001	
Density	1	0.0111	
Density x Site	2	0.0001	
1978 Developed		0.0001	0.0882
Site	2	0.0001	
Density	1	0.0391	
Density x Site	2	0.0001	

Note: For each factor (e.g., 1956 canal), land loss was modeled as Land Loss = Site + Density + Density x Site, where loss is the loss rate (R) for each of 23,361 WINDOW samples; Site is either the Cameron, Terrebonne, or Lafourche study site; density is the area of that factor within a square km window divided by the 1956 land area in that window; and density x site is the density/site interaction. DF are the degrees of freedom for each term in the model, PR>F is the significance either of the model or each of the terms (based on the type III sum of squares), and the remaining column is the R^2value for that model. Developed is equivalent to agriculture plus urban/industrial.

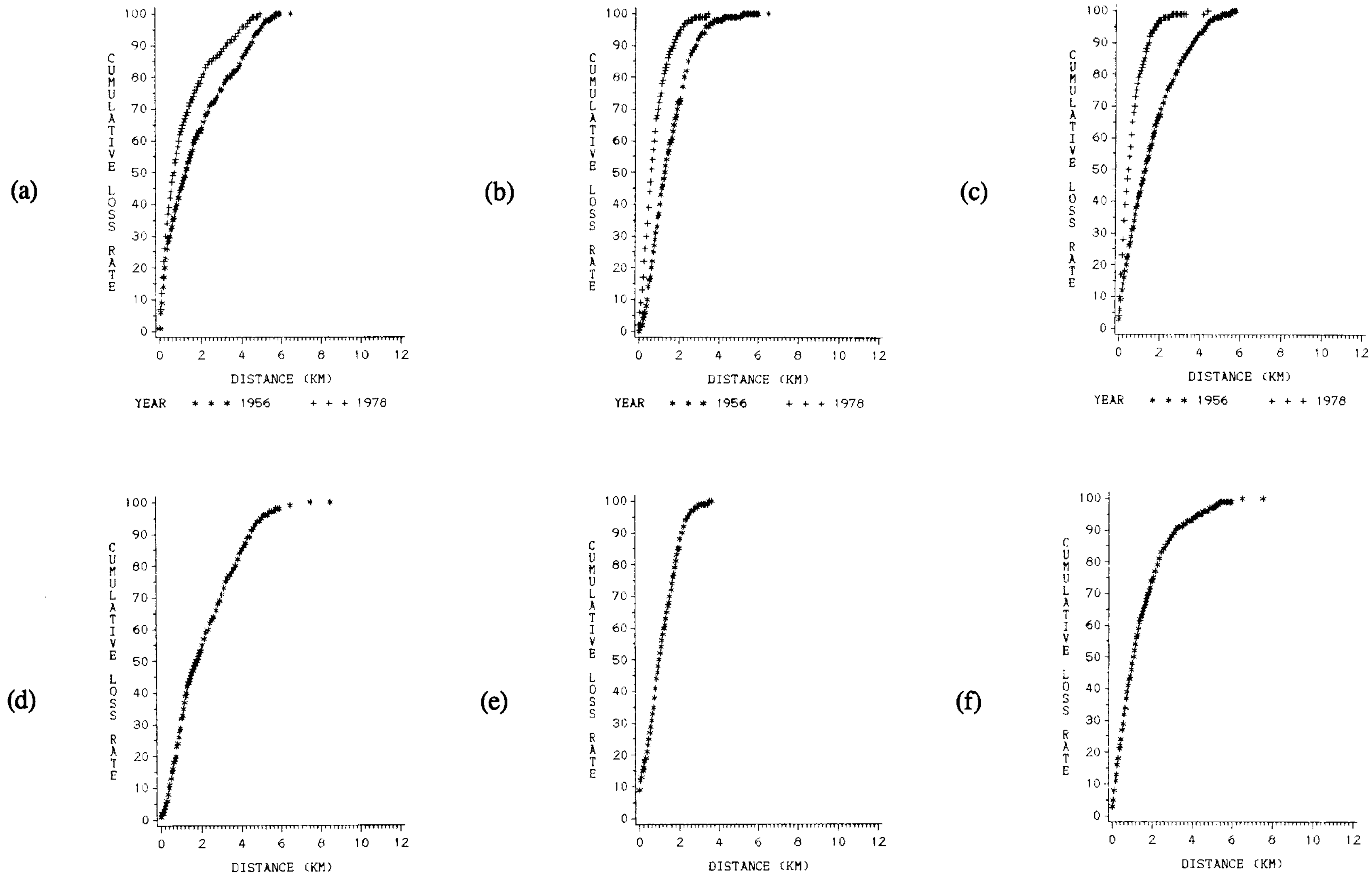

Figure 20-5. Cumulative land loss (excluding direct loss caused by canals) versus distance to 1956 and 1978 canal and spoil (a-c) and distance to 1956 natural channel (d-f) for the Cameron (a and d), Terrebonne (b and e), and Lafourche (c and f) study sites. Cumulative loss is the proportion of all indirect loss within a site that occurred in the interval between 0 km and some other distance.

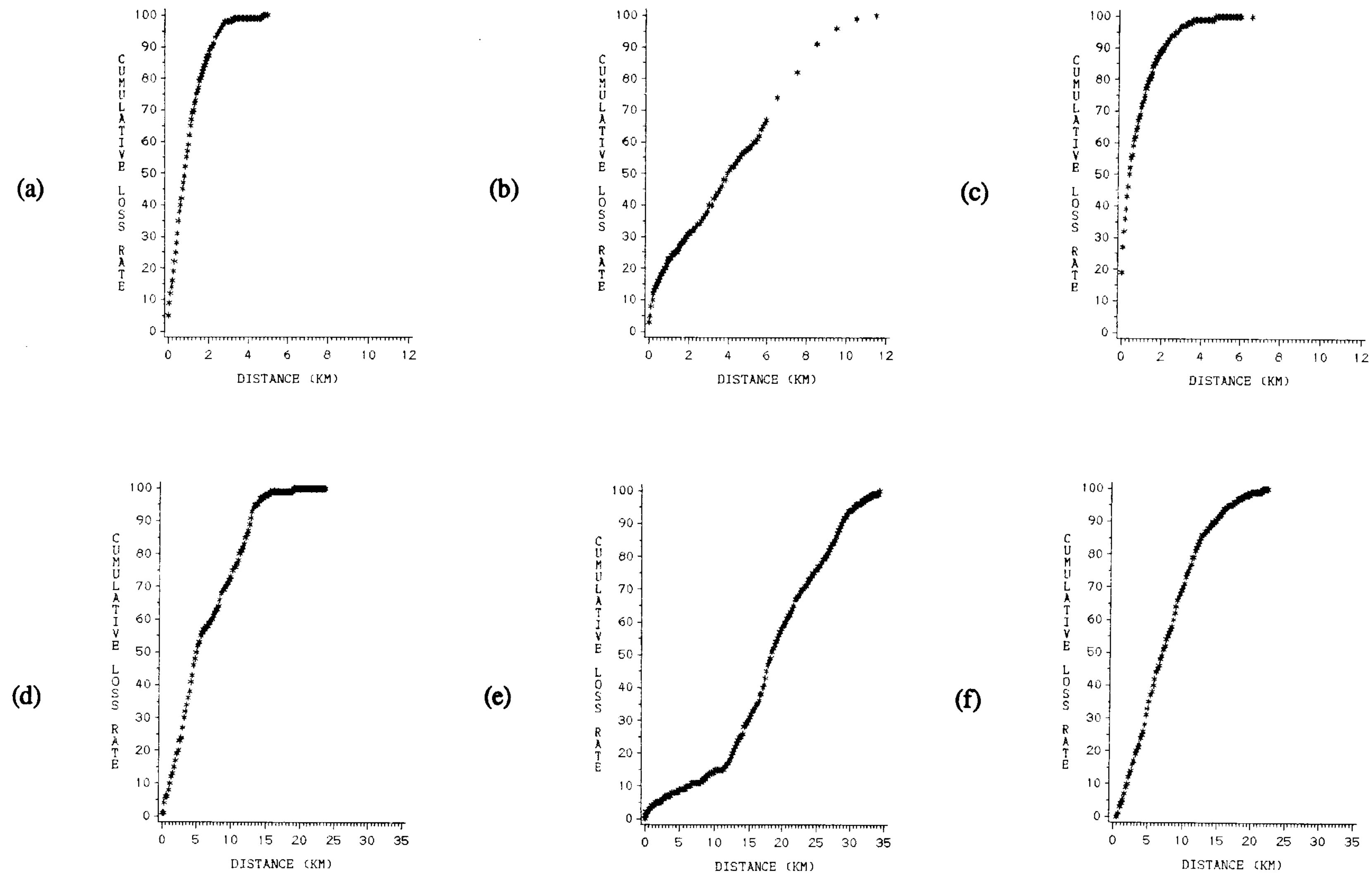

Figure 20-6. Cumulative land loss (excluding direct loss caused by canals) versus distance to 1956 inland open water (a-c) and distance to 1956 major channel (d-f) for the Cameron (a and d), Terrebonne (b and e), and Lafourche (c and f) study sites. Cumulative loss is the proportion of all indirect loss within a site that occurred in the interval between 0 km and some other distance.

The results of cumulative land loss with respect to canal and spoil density give a better indication of how much these features contribute to land loss (Figure 20-7). For the Cameron, Terrebonne, and Lafourche study sites, 59 to 69, 53 to 82, and 42 to 73% of all land loss occurs in areas with a canal and spoil density close to zero. Although these curves show that the majority of land loss occurs in low density regions, they also show that it is the lowest densities of canal and spoil that have the largest additional impact on land loss. (Since the density of canal and spoil is measured within a square window, it is possible that some land loss is affected by canal and spoil outside the window; the large number of randomly chosen samples, however, should minimize this source of error.) Within a short density range, canal and spoil may contribute to 18 to 58% of the total land loss at these sites (this does not prove that canal and spoil are responsible for this because other factors could be causing the loss in areas that also happen to contain canal and spoil). These areas are saturated beyond a certain point. The critical densities (the density where 99% of all land loss has occurred) are 0.06 to 0.11, 0.14 to 0.26, and 0.27 to 0.35 km^2 canal and spoil per km^2 land for the Cameron, Terrebonne, and Lafourche study sites, respectively. Thus, construction of an additional canal does not contribute much to indirect land loss beyond this critical point, since the area is already saturated. The argument that there is little harm in building a canal in an area of "virgin" marsh is flawed, however, because this is exactly where any impacts of canal and spoil would be greatest.

Landloss Model

The distance and density factors discussed previously all had low R^2 values, indicating that a large amount of variation remained unaccounted for. In an attempt to rectify this, a model was constructed (Model "A") that included the qualitative variables (site, 1956 habitat type, and salinity change) along with the quantitative variable distance to coastal shore (this was the quantitative factor with the highest R^2):

$$R = \text{Site} + \text{Habitat Type} + \text{Salinity Change} + \text{Distance To Shore} + \text{Site} \times \text{Habitat} + \text{Site} \times \text{Salinity} + \text{Site} \times \text{Distance To Shore} \quad \text{(Eqn. 20.6)}$$

This model was highly significant, with $P < 0.0001$ and an R^2 of 0.2410 (Table 20-9). Including habitat type and salinity change, therefore, improved the model.

Although the model was significant, 75% of the variation in land loss was still unaccounted for. We then investigated the distribution of outliers: whether they were randomly distributed or spatially clumped. For example, it was possible that land loss in a few large, contiguous areas ("hot spots") could skew the results, even though a large portion of the area was accurately predicted. For each of 23,361 points, the predicted land loss, based on Model A, was compared to the actual loss in the square kilometer window. If the observed loss was within the 90% confidence interval of the predicted value, the datum was plotted as a point ("."). Otherwise, the values were plotted as a "+" or a "-" to indicate that the predicted value was greater or less than the observed value (Figure 20-8).

The plot of residuals shows that the outliers are indeed spatially clumped, and that land loss is accurately predicted for the majority of points. Further, by comparing Figure 20-8 with Figure 20-4, we see that these outliers are associated with zones of abnormally high land loss. Therefore, it appears that land loss actually consists of two separate groups: a few "hot spots" of high loss embedded within a background of much lower loss. A third, smaller group can also be differentiated which contains loss rates near zero. These data were divided into three residual classes by examining the histograms of the residuals. Loss rates (R_{class}) for the low, medium, and high residual classes were 2.1, 11.3, and 61.4%,

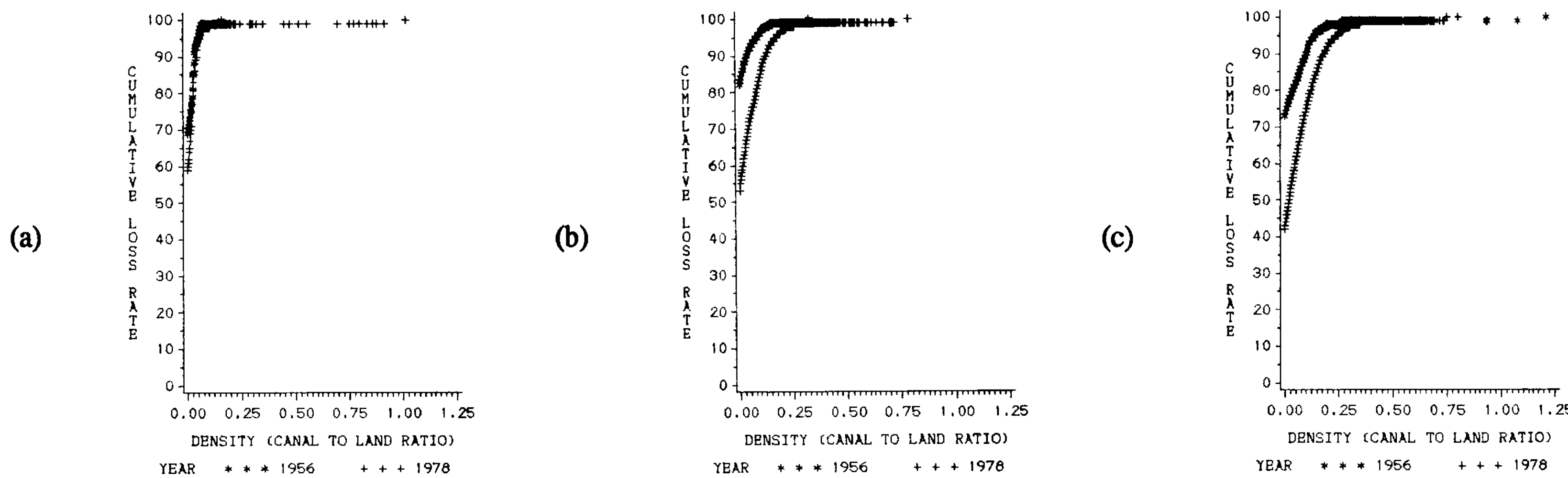

Figure 20-7. Cumulative land loss (excluding direct loss caused by canals) versus density of canal and spoil (1956 and 1978) for the Cameron (a), Terrebonne (b), and Lafourche (c) study sites. Cumulative loss is the proportion of all indirect loss within a site that occurred in the interval between 0 and some other density.

respectively. Although the high residual class contained only 11.9% of all land, it accounted for 42.7% of all land loss (P_{class}).

Table 20-9. Model results of landloss factors without and with residual classes.

Model/Source	DF	PR > F	R^2
Model A		0.0001	0.2410
Site	2	0.0001	
1956 Habitat Type	8	0.0001	
Salinity Change	3	0.0001	
1956 Distance to Coastal Shore	1	0.0001	
Site x Habitat	12	0.0001	
Site x Salinity	4	0.0001	
Site x Distance to Shore	2	0.0001	
Model B		0.0001	0.7536
Site	2	0.0001	
1956 Habitat Type	8	0.0001	
Salinity Change	3	0.0001	
Residual Class	2	0.0001	
1956 Distance to Coastal Shore	1	0.0001	
Site x Habitat	12	0.0001	
Site x Salinity	4	0.0001	
Site x Residual Class	3	0.0001	
Site x Distance to Shore	2	0.0001	

Note: For both cases, loss rate (R) is modeled as the sum of the subsequent terms, based on the 23,361 WINDOW samples. DF are the degrees of freedom, PR>F is the significance either of the model or each of the terms (based on type III sum of squares), and the remaining column is the R^2 value for that model.

Based on these residual classes, a new model (Model B) was constructed:

$$\begin{aligned} R = {} & \text{Site} + \text{Habitat Type} + \text{Salinity Change} + \text{Residual Class} + \\ & \text{Distance To Shore} + \text{Site x Habitat} + \text{Site x Salinity} + \\ & \text{Site x Residual Class} + \text{Site x Distance to Shore} \end{aligned} \qquad \text{(Eqn. 20.7)}$$

This model was significant at $P < 0.0001$ and with an R^2 of 0.7536 (Table 20-9). The amount of variation accounted for by the residual variable was an order of magnitude greater than any of the other factors. What this essentially means is that a small proportion of the total study area (the outliers) accounts for a large amount of the variation found in land loss.

Equation 20.7 was modified to include 1978 canal density and the site x 1978 canal interaction to determine whether inclusion of canal would improve the model. The R^2 went from 0.7536 to 0.7578, indicating that this factor has little utility in predicting land loss for these sites. Similar findings were obtained when 1956 canal density was used.

The question, then, is what makes these "hot spots" different? Mean values for the distance and density variables are presented in Table 20-10 by residual class and by site. Although differences appear, they are not consistent within a site or between sites. For example, at the Lafourche site, the 1978 canal density is significantly lower for the low residual class; at the Terrebonne site, however, 1978 canal density is significantly lower at

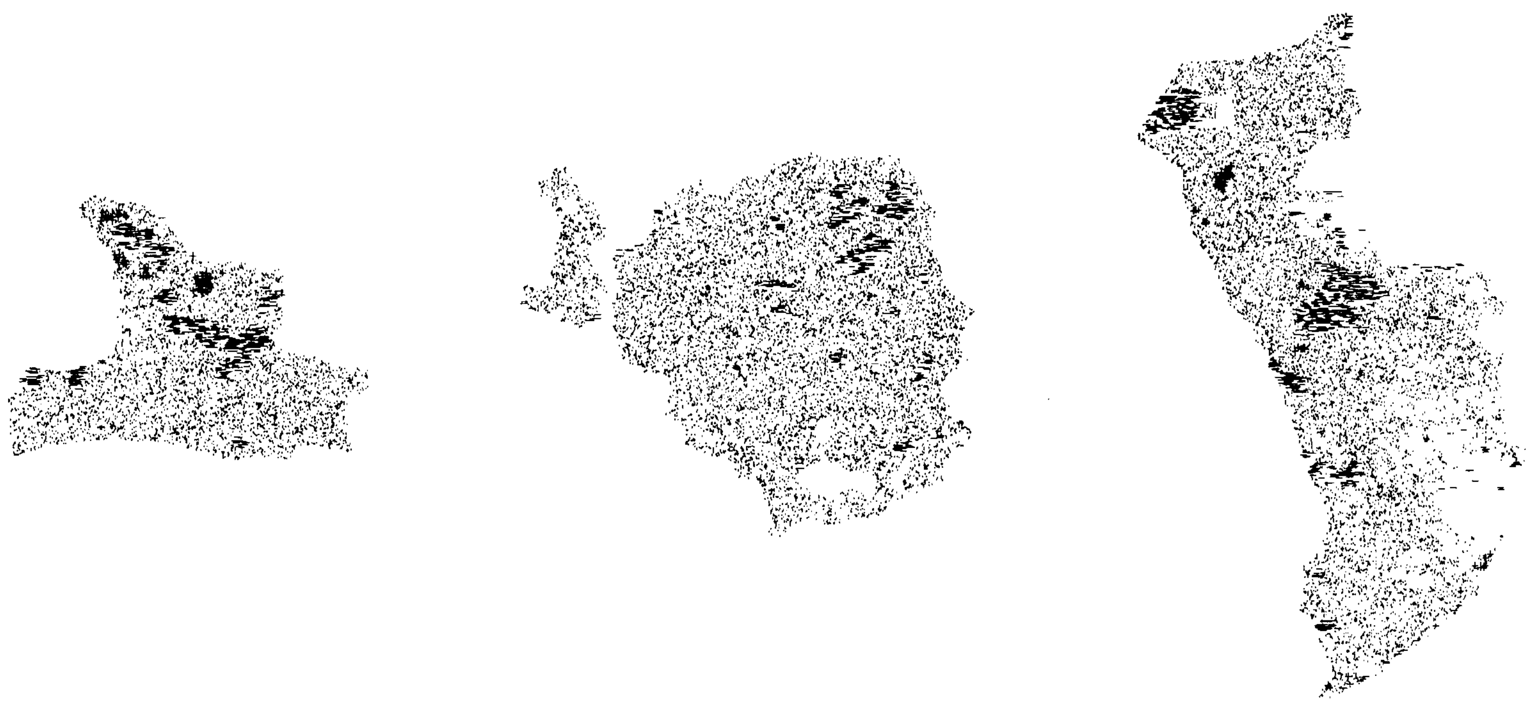

Figure 20-8. Plot of residuals for the Cameron (left), Terrebonne (center), and Lafourche (right) study sites based on landloss model A (Table 20-9). For each datum, the value is plotted as a point (•) if the observed loss was within the 90% confidence interval of the loss predicted by the model. Otherwise, values are plotted as a plus (+) or minus (-) if the predicted loss was greater or less than the observed loss, respectively. Model is based on the 23,361 window samples.

Table 20-10. Comparison of distance and density factors by residual class and site.

Factor	Residual Class	Cameron Mean	Cameron Group	Terrebonne Mean	Terrebonne Group	Lafourche Mean	Lafourche Group
Loss Rate	High	66.62	A	51.60	A	63.49	A
	Medium	8.92	B	10.54	B	14.33	B
	Low	3.17	C	-	-	1.33	C
1956 Canal and Spoil[a]	High	1.83	A	1.43	A	1.40	A
	Medium	1.80	A	1.63	B	1.39	A
	Low	1.65	A	-	-	0.89	B
1978 Canal and Spoil[a]	High	1.20	A	0.80	A	0.53	A
	Medium	0.95	B	0.60	B	0.54	A
	Low	1.48	C	-	-	0.74	B
1956 Natural Channel[a]	High	2.19	A	1.25	A	1.84	A
	Medium	3.05	B	0.80	B	1.29	B
	Low	1.79	A	-	-	1.42	B
1956 Major Channel[a]	High	6.72	A	17.27	A	7.23	A
	Medium	9.04	B	14.88	B	8.50	B
	Low	4.71	C	-	-	4.80	C
1956 Inland Open Water[a]	High	1.05	A	4.78	A	0.98	A
	Medium	1.23	A	4.11	B	1.10	A
	Low	1.25	A	-	-	2.10	B
1956 Coastal Shore[a]	High	11.86	A	17.14	A	27.36	A
	Medium	8.33	B	12.23	B	26.04	A
	Low	16.87	C	-	-	33.40	B
1956 Canal[b]	High	84.48	A	48.70	A	140.35	A
	Medium	28.09	B	59.88	A	140.90	A
	Low	35.56	B	-	-	79.12	A
1956 Spoil[b]	High	2.96	A	28.51	A	163.80	A
	Medium	14.48	A	73.51	B	135.02	A
	Low	0.00	A	-	-	97.69	A
1956 Agriculture[b]	High	237.57	A	-	-	1,027.09	A
	Medium	1,815.49	B	-	-	258.27	B
	Low	612.36	A	-	-	5,491.55	C
1956 Urban/Industrial[b]	High	11.71	A	17.26	A	14.94	A
	Medium	158.20	B	10.76	A	18.89	A
	Low	11.04	A	-	-	134.83	B
1978 Canal[b]	High	49.02	A	190.18	A	316.35	A
	Medium	45.71	A	239.07	B	394.20	A
	Low	6.73	B	-	-	118.81	B
1978 Spoil[b]	High	62.69	A	161.14	A	279.35	A
	Medium	84.25	A	321.10	B	304.61	A
	Low	100.75	A	-	-	154.90	B
1978 Agriculture[b]	High	145.63	A	-	-	91.34	A
	Medium	1,878.79	B	-	-	580.82	B
	Low	449.91	A	-	-	5,683.94	C
1978 Urban/Industrial[b]	High	16.18	A	6.36	A	16.45	A
	Medium	189.70	B	3.83	A	56.60	A
	Low	0.59	A	-	-	190.23	B

Note: Values are means for different factors in each residual class, based on 23,361 WINDOW samples. For each factor and site, means with the same letter are not significantly different at the 99% confidence level (Scheffe's multiple-comparison procedure). For example, there is no significant difference between the mean 1956 canal and spoil distance at Lafourche for the high and medium residual classes, because both values are followed by an A. Comparisons were made within each factor and site, however, and thus a value followed by an A at the Cameron site bears no relationship to the means at the Lafourche site. The Terrebonne study site had neither agriculture nor low residual data. Rate=land loss rate (R).

[a] distance (km) from the center of the km^2 window to the nearest occurence of that factor

[b] density (the ratio of the area of a factor within the km^2 window to the 1956 land area in that window).

the high residual class. The lack of a clear trend could indicate one of two possibilities: first, that clumping the high loss outliers together is an artificial grouping and that each "hot spot" must be analyzed separately; or secondly, that other factors not included account for land loss (e.g., brine disposal, subsidence caused by mineral extraction, topography, soil characteristics). This analysis points out, however, that more attention should be paid to those areas with anomalously high land loss rates. Field investigations would be especially beneficial at these locations.

Summary

Based on analysis of the Cameron, Terrebonne, and Lafourche study sites with 10 m x 10 m digital habitat data, the following findings have been reported:

(1) Loss rates were consistent with what would be expected based on the geology of the site. The lowest rate of land loss was at the Terrebonne study site (14.4%), which contains a major sediment source. The highest rate was at the Lafourche study site (20.8%), adjacent to a recently abandoned distributary. The rate at the Cameron study site (16.9%), in the Chenier Plain, was intermediate.

(2) Marsh habitats account for the most land loss at the three study sites. Loss of marsh (both saline and fresh) accounts for 85.5 to 95.5% of all loss. Each of the other habitat types accounts for less than 5% of the loss at a site, except for agriculture at the Lafourche study site (7.5%).

(3) Loss of land within the marsh interior is the major form of land loss and not loss of shoreline. Conversion of land to inland open water (lakes, ponds, and Barataria Bay), accounts for 69.7 to 92.8% of all loss. Direct loss from canals accounts for 2.5 to 15.3%, and values for natural channel range from 2.1 to 13.8%. Loss of shoreline (coastal erosion) accounted for less than 5% of the total loss in all cases.

(4) Loss rates are highest in narrow, transitional habitats and lowest in the more upland areas. Loss rates for beach/dune/reef and mudflat are highest (46.7 to 55.1%), while values for uplands, urban/industrial, swamp, and agriculture are lowest (0.0 to 12.1%). Values for marshes and spoil are intermediate (17.8 to 36.2%).

(5) Saltwater intrusion is not believed to be the major cause of land loss at the Terrebonne site, although this could not be ruled out for the Cameron and Lafourche sites. For the Cameron and Lafourche sites, there is no significant difference between loss rates in areas of increasing or high salinity (fresh-to-salt or salt-to-salt categories) nor is there a significant difference between areas of low or decreasing salinity (fresh-to-fresh or salt-to-fresh). However, values in the two former classes are significantly higher (21.6 to 26.7%) than rates in the latter two (0.0 to 2.2%). Because land loss is correlated with saltwater intrusion, it is not possible to disprove this as a causative factor at the Cameron or Lafourche study sites. At the Terrebonne site, however, land loss was highest in areas that remained fresh for the 22-year period (fresh-to-fresh), with no significant difference between this category and the areas of increasing (fresh-to-salt) or high (salt-to-salt) salinity (12.4 to 16.2%). Further, these values are significantly lower than those found for the fresh-to-salt and salt-to-

salt categories at the other two sites. Therefore, it is concluded that saltwater intrusion was not a major factor at the Terrebonne site.

(6) Distance measures, such as distance to canal, and density measures, such as density of canal area, were not useful in predicting land loss. Although statistical models of land loss that included distance factors (distance to canal and spoil, natural channel, major channel, inland open water, and coastal shore) and density factors (density of canal, spoil, canal and spoil, agriculture, urban/industrial, and developed) were highly significant, it is concluded that this significance was the result of the large sample number rather than because of real ecological effects. This is concluded because (1) the effects within factors or between sites or years are not consistent and (2) R^2 values are low (0.0202 to 0.1086) in all cases . Although these factors do not account for a significant portion of the variation in land loss over their entire range of values, they may be significant within a narrower range.

(7) The effect of land loss with respect to distance to canal and spoil is similar to the effect with respect to distance to natural channels. Curves of cumulative land loss versus distance to canal and spoil or distance to natural channel are similar, indicating that the same mechanism may be responsible for the loss near these features (e.g., compaction of sediments by spoil or natural levee).

(8) Land loss at the Terrebonne site may be result of a newly initiated process. For the Cameron and Lafourche study sites, a large portion of the loss is located adjacent to inland open water. Since land loss previous to 1956 would show up as inland open water, this may be an indication that this is the continuation of a previously initiated landloss process. This is not so at the Terrebonne site, however, and may be because loss at that site is the result of a recently initiated process.

(9) Nearness to a major sediment source reduces land loss. Plots of cumulative loss versus distance to major sediment source show that most loss occurs at distances far from these features. For a major sediment source, such as the Atchafalaya River, loss rates are greatly reduced to a distance of 12 km.

(10) A large portion of the land loss at the three study sites occurs in areas with canal densities near zero. Cumulative curves of percent land loss versus canal density indicate that 42 to 82% of all land loss occurs in areas with nearly zero canal and spoil density. Canal and spoil do appear to contribute to land loss below a certain critical density (0.06 to 0.11, 0.14 to 0.26, and 0.27 to 0.35 km^2 canal and spoil per km^2 land for the Cameron, Terrebonne, and Lafourche study sites, respectively). Beyond these densities, however, an area is saturated, and construction of an additional canal apparently adds little to overall indirect land loss. Canal and spoil appear to cause the most damage in areas of solid marsh.

(11) Land loss is not a uniform, homogeneous process; rather, "hot spots" of high loss occur. A model, which includes site, habitat type, salinity change, distance to coastal shore and the interaction of each with site (e.g., site x habitat), is significant but had an R^2 of only 0.241. Therefore, 75% of the variation in land loss is still unaccounted for. The residuals from this model are not randomly scattered, however, but are spatially clumped. Further, outliers are associated with areas of extremely high land loss. It is possible to

separate the data into three classes based on residuals. Loss rates for the low, medium, and high residual classes are 2.1, 11.3, and 61.4%, respectively. While the high residual class contains only 11.9% of all land, it accounts for 42.7% of all loss. Introducing the residual class into this model raises the R^2 value to 0.754. These "hot spots" account for a large amount of land loss, and field studies are needed to specifically determine the factors that cause their development.

Chapter 21

NEW HOLES IN AN OLD MARSH

by

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The causes of the severe wetland loss rates in coastal Louisiana (0.86% annually or 20,000 ha/yr from 1955/6-78) have been attributed to both natural and man-made causes (Craig et al., 1979; Scaife et al. 1983; Walker et al., 1987). The coastal zone, built over 7,000 years by a series of 15 major deltaic episodes, has gradually submerged by global sea level rise and the sinking of land caused by sediment compression, de-watering and a geosynclinal shift in the basement material. Both sediments and plant materials must accumulate to balance the loss of vertical position relative to sea level. But, the supply and distribution of sediments, organic materials, and plants is not static in recent time. Suspended sediments in the Mississippi River have apparently declined by 50% since the early 1950s (Meade and Parker, 1984)). Further, the flood-protection levees along the lower Mississippi River reduced overbank flooding, except at the river's terminus. The hydrologic patterns, which initially built these wetlands and now sustain the plants and sediments holding it together, changed as 7% of the wetlands were converted into canal and spoil banks (built for oil and gas recovery operations; Swenson and Turner, 1987). It is not clear how much or how quickly the landscape can adjust to such perturbations. The relative effect on wetland loss rates of the changes in sediment supply, the relatively constant water level rise, and man-made influences are the subjects of several studies aimed at determining solutions and minimizing impacts.

Analysis of landscape changes is useful to test hypotheses about the relative influence and interaction of various proposed causal factors that lead to wetland loss. It is readily observed that wetlands do not disappear without first disintegrating; wetlands fragment as they turn into open water. It is the result of this fragmentation into small and then larger open water bodies that we studied here. We wanted to know how the distribution of new open water bodies, or holes in the marsh, was related to the geologic framework, time for development, and to man-made influences, principally canals and associated spoil banks. We examined: 1) the distribution of new holes in the marsh from 1955/6 to 1978; 2) the distances from these holes to natural and man-made hydrologic influences; and, 3) if new or old canals and associated spoil banks have the greater influence on marsh fragmentation.

Methods And Materials

We mainly used digitized habitat maps for coastal Louisiana stored in the Geographical Information System (GIS) in the Data General computer at the Department of Natural Resources (DNR), Baton Rouge, LA. The GIS has the habitat data in the form of 244 7.5 minute quadrangle maps (approximately 41,000 acres or 16,600 hectares) for 1955/6 and 1978. Habitats on these maps are described by the habitat codes of Cowardin et al. (1979).

The digitized data can be viewed, extracted and analyzed with the aid of a Map Overlays and Statistical System (MOSS, version 8509) interactive graphics software developed and updated under the direction of the U.S. Bureau of Land Management. Further information about MOSS can be obtained from The Bureau of Land Management, Division of Advanced Data Technology, Branch of Technology Applications and Assistance, Bldg. 50, Denver Federal Center, P.O. Box 25047, Denver, CO 80225-0047.

The habitat codes, for each quad map, were grouped into 7 categories: open water, upland, wetlands, canals, levees, area out of coastal zone and "unassigned". The category unassigned represents the habitat codes which do not signify anything, were unidentifiable or were misrepresented while digitizing; these areas amount to less than 0.001% of coastal Louisiana.

The maps stored in the GIS are permanent, i.e., the maps are write-protected to protect data integrity. Because grouping the habitat codes on these original maps into different categories is not possible, maps from GIS were copied to another directory, renamed, and their access control list (ACL) and permanence rules changed. This procedure enabled any user to group the habitat codes into desirable categories using the ASSIG program available on the system (written by Decision Associates, Inc.).

Routines were developed for the MOSS to simplify and speed up the operations by creating 'macros' which execute the CLI (Command Line Interpreter) commands under the AOS/VS environment on the Data General Computer.

The open water category was sub-divided into ponds in the ranges of 0-20 ha (0-50 acres), 20-40 ha (50-100 acres), 40-50 ha (100-150 acres) and >60 ha (>150 acres). We also mapped ponds in the 0-8, 8-20, and 20-60 ha size range to use for discussion purposes. The areas of the open water, uplands, wetlands, canals, levees etc. for each map was obtained along with the percentage amount of each category in the whole map. Also, the area and number of the ponds in the four different ranges were obtained. Hard copies of the maps were plotted on the Versatec Plotter (Model 7224; 200 dots per inch) and some were re-entered into a personal computer program to be used as examples below.

We obtained maps and information for seventy-two 1955/6 and 1978 habitat maps. For each pair of maps the plots showing the ponds in each size range was compared and the new, persistent and transient ponds were distinguished. New ponds are ponds not existing in the 1955/6 map, but found in the 1978 map. Ponds found in both the 1955/6 and 1978 maps were termed persistent ponds. Ponds found in the 1955/6 map, but not in the 1978 map, were termed transient ponds.

The analysis of pond distribution was done considering the fact that two or more ponds in the lower size range could have merged causing it to disappear from the lower size range of ponds and therefore appear as a new pond in the higher size range of ponds. Also, it is possible for a pond in the higher size range to have disintegrated between 1955/6-1978, causing it to disappear from the higher size range and appear as a new pond in the lower size range. Such discrepancies were closely scrutinized and not included in the analysis.

The distribution of new ponds in the vicinity of the canals was determined for sixty-three map pairs. All three pond types (new, persistent, and transient) of each size range were counted as a function of distance to a canal present in 1955/6 and/or 1978, for distances up to 4.2 km from the canal. All the new ponds <20 ha within 1 km of the 1955/6 and 1978 canals were also counted. Distance was measured by hand from the canal water edge to the nearest part of the pond-water interface.

We examined the relationship canal area and the number of new ponds of different sizes for four different map groupings: maps in the Chenier Plain, the St. Bernard delta, the Barataria and Terrebonne hydrologic units, and those maps containing the Mississippi River Gulf Outlet (MRGO; Figure 21-1). These correspond to the groupings in Cowan and Turner (Chapter 19), with the exception of the additional grouping for maps including portions of MRGO. The creation of this latter map grouping is justified on the basis of the unusual length and width of this man-made channel (it extends across the St. Bernard sub-delta from the Gulf of Mexico to New Orleans) and because the channel is dredged frequently for ocean-going barges and most of the dredged spoil is placed on the western side of the channel. All together, 38 % of the wetland loss from 1955/6 to 1978 in the Louisiana coastal zone is represented in these maps. Data used for open water changes vs canal area relationship determination were omitted from portions of the analysis when: (1) map data were clearly entered incorrectly (1 map); (2) significant urbanization or agricultural development occurred (6 maps); or (3) pre-1955/6 agricultural impoundments were evident (2 maps).

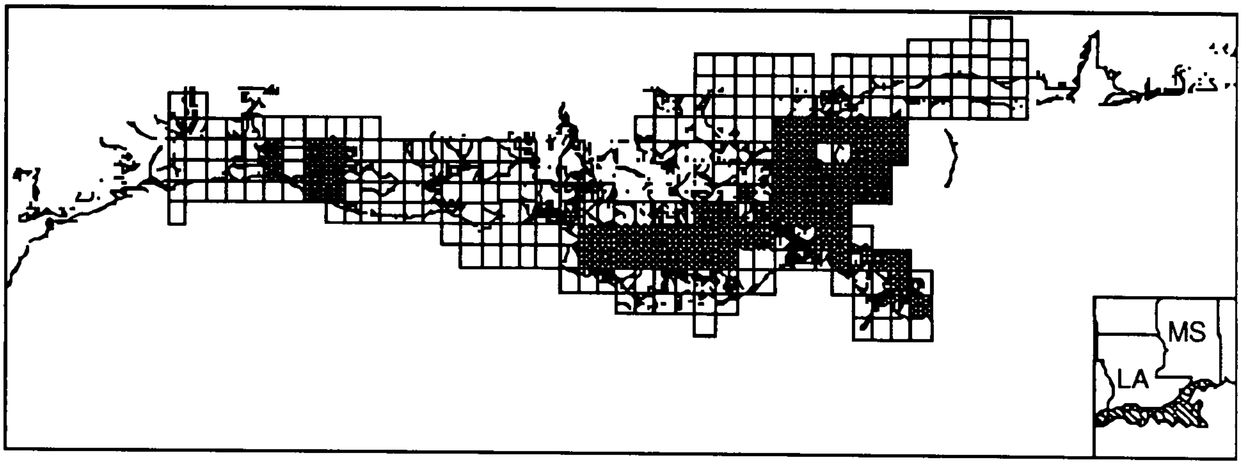

Figure 21-1. The study area, showing quadrangle maps; the shaded areas indicate quadrange maps used in analyses.

Results

Qualitative Changes

The observed diversity of landscape change cannot be systematically or fully explored in this brief summary. However, some general features were apparent and are presented here. These features involve changes in the types of open water bodies formed, their location with regards to spoil banks and canals, and, apparent changes in natural hydrology. We will use a few maps for examples.

Virtually all maps have a higher number of the smallest ponds (0-20 ha) relative to the other pond sizes. An example is shown for Mink Bayou (Figure 21-2). Here the number of ponds 0-20 ha is especially numerous, and the number declines with increasing pond size. A smaller number of ponds mapped in 1955/6 were not mapped in 1978, but most ponds present in 1955/6 were also present in 1978. Many of the ponds were either parallel to the canal or had an apparent hydrologic connection leading directly to the canal.

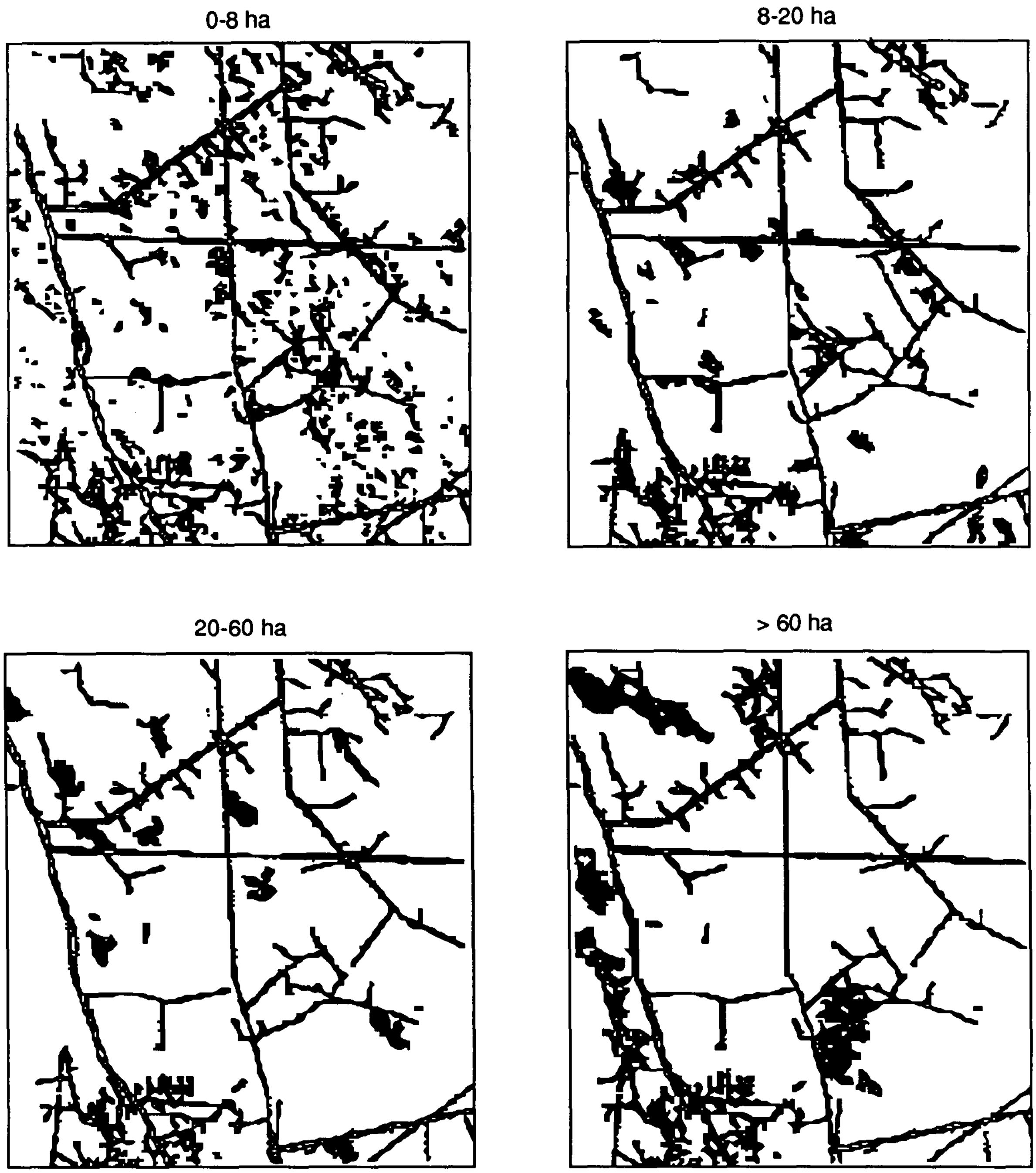

Figure 21-2. The wetland loss as ponds (in black) in the Mink Bayou quadrangle map from 1956-1978. Four different pond sizes are shown. Canals present in 1978 are the straight, dark lines.

The smaller new ponds appeared to be clustered near canals, and the association became more pronounced as the pond size increased. The larger ponds were most often clustered next to a canal (quantified below), especially, it appeared, when the pond was bordered on two or more sides by a canal. We did not estimate the degree of enclosure of ponds by canals, because of lack of resources and ideas to precisely evaluate the degree of "enclosure" efficiently and objectively. A few examples are shown in Figure 21-3 using 4 quadrangle maps depicting new ponds 20-60 ha (in black) and canals (straight lines). These four quadrangle maps were mostly wetland in 1955/6, except for the Bastian Bay quadrangle map, whose southwestern quadrat was open water. The Mink Bayou (previous figure), Golden Meadow, and Bastian Bay maps show the clumping of new ponds (of this size) next to the canals, and often where canals come together. The Cutoff map, in contrast, had no ponds form of that size, despite the presence of many canals. The only pond to form in the entire Larose map area, otherwise wetland, is next to a canal.

The largest new water bodies were occasionally the apparent consequence, or by-product, of impounding the wetland with a man-made levee. A particularly clear example of this was in the Cutoff quadrangle map (Figure 21-4). An agricultural impoundment of the early part of this century (Turner and Neill, 1984) apparently failed at some point, and the vegetation present in 1955/6 was no longer apparent in 1978. Ninety-nine percent all of the wetland loss (1359 ha) in this quadrangle map was inside of the impoundment. Such examples taught us to examine each map for unusual circumstances leading to wetland loss.

We often observed that the natural stream channel network seemed to disappear when canals and spoil banks blocked tidal exchange. The upstream, not the downstream, channel was more likely to disappear. These observations are supported by Craig et al. (1979) who measured a logarithmic decrease in natural stream density with a linear increase in canal density for 1 km square grids around Leeville, La.

Quantified Relationships

The distribution of the new, persistent, and transient ponds within 4.2 km from a 1978 canal is shown for all the ponds in the study maps in Figure 21-5. Two data distributions are described. The upper panel shows the percent of all 0-20 ha ponds (the total number is in the figure legend) within the ten range divisions going away from the canal, left to right. The lower panel shows the same data presentation for ponds sized 20-50 ha. No standard deviations are depicted since all ponds in each map were included in the analysis.

New ponds between 0-20 ha are 6 times more numerous than all other pond sizes combined. The numerical distribution of the new 0-20 ha ponds with distance from a canal is not the same as for ponds classified as persistent or transient. Two patterns are evident. First, the numerical distribution of transient ponds is not spatially distributed as the new and persistent ponds are distributed, which is coincidental. The percent of new and persisting ponds is inversely related to distance from the canal. Forty percent of these two pond types is found within 0.4 km from a canal, compared to 15 %, or less, for the transient ponds. Second, the percent of transient ponds does not peak nearest the canal, but peaks at about 1.0 to 1.5 km from the canal, and then declines with distance either away from or towards a canal. In other words, the new and persisting ponds are located nearest the canal and the transient ponds are most frequently further away from the canal. All three distributions decline to near zero percent several km from the canal. Clearly something about the canals and spoil banks, or something related to them, influences the pond formation and disappearance; if otherwise, the spatial distribution of ponds with distance from a canal would be coincidental, and probably would reflect the percent of marsh with distance from a canal. It is also not a mapping error since the cartographers had no a priori information to distinguish between new, persisting, and transient ponds of any size.

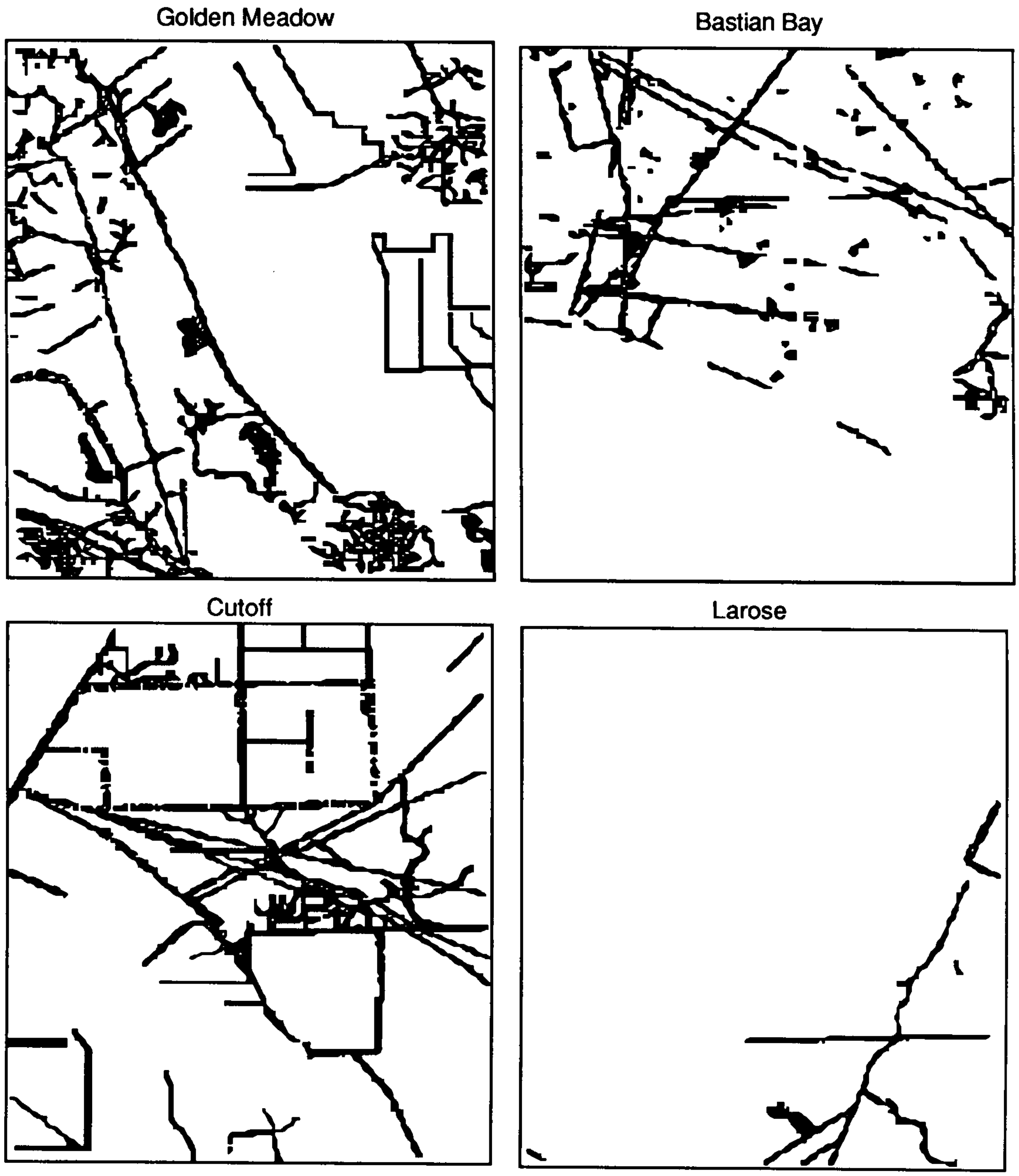

Figure 21-3. The wetland loss from 1955/6-1978 as ponds 20-60 ha (in black) for four quadrangle maps in the Barataria hydrologic unit. Canals present in 1978 are the straight dark lines.

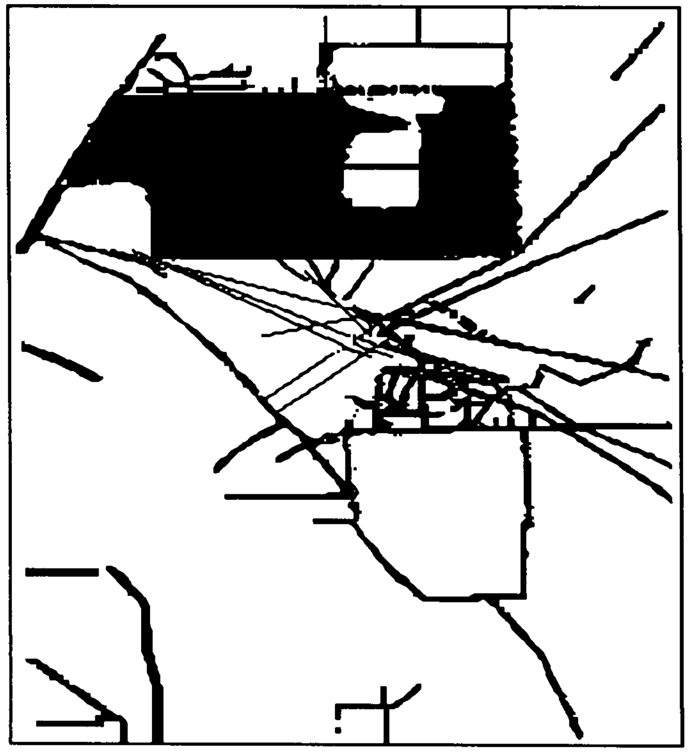

Figure 21-4. The wetland loss (in black) from 1956-1978 as ponds >60 ha for the Cutoff quadrangle map. Canals present in 1978 are the straight dark lines. Dashed lines are canals present in 1956 but not mapped in 1978. One massive pond formed in the map area, but no where else.

The new and persistent ponds of 20-50 ha are located much more closely to the canal than the smaller ponds. At least 75 % of this size pond is within 0.4 km of a canal, or about twice that the density of the 0-20 ha ponds. No transient ponds 20-50 ha, or larger, were mapped in the entire study site (the reader is reminded that ponds were not considered transient if they enlarged). If the 0-20 ha and 20-40 ha new ponds were randomly distributed and not influenced by canals and spoil banks, then their distributions should be similar; they are not.

The number of new ponds <20 ha per ha canal within 1 km from a 1955/6 or 1978 canal, was about the same; no statistically-significant differences in density were found within 1 km of a canal for those two canals for that sized pond. We could not, therefore, disprove (with this analysis) the hypothesis that old canals have a similar influence as newer canals in creating new open water bodies.

In most cases, it was apparent that if the area of ponds 0-20 ha increased from 1955/6 to 1978, then the area of ponds in the other size classes was likely to increase. The slope of a linear regression between net pond areal growth for ponds of different sizes was always positive, with one exception (for 0-20 vs 20-40 ha ponds in the quadrangle maps including the Mississippi River channel). Regional differences were apparent in the relationship between these two variables. However, a linear regression analysis of the data generally showed the highest coefficient of determination, R2, between the 0-20 ha and 20-40 ha pond area, than between the area of 0-20 ha and area of larger pond sizes. In other words, small ponds can be a surrogate for estimating the area in numerous intermediate sized ponds, than for the fewer, but larger ponds. But, predicting wetland loss using area or number of the smaller ponds gets more difficult as pond size increases.

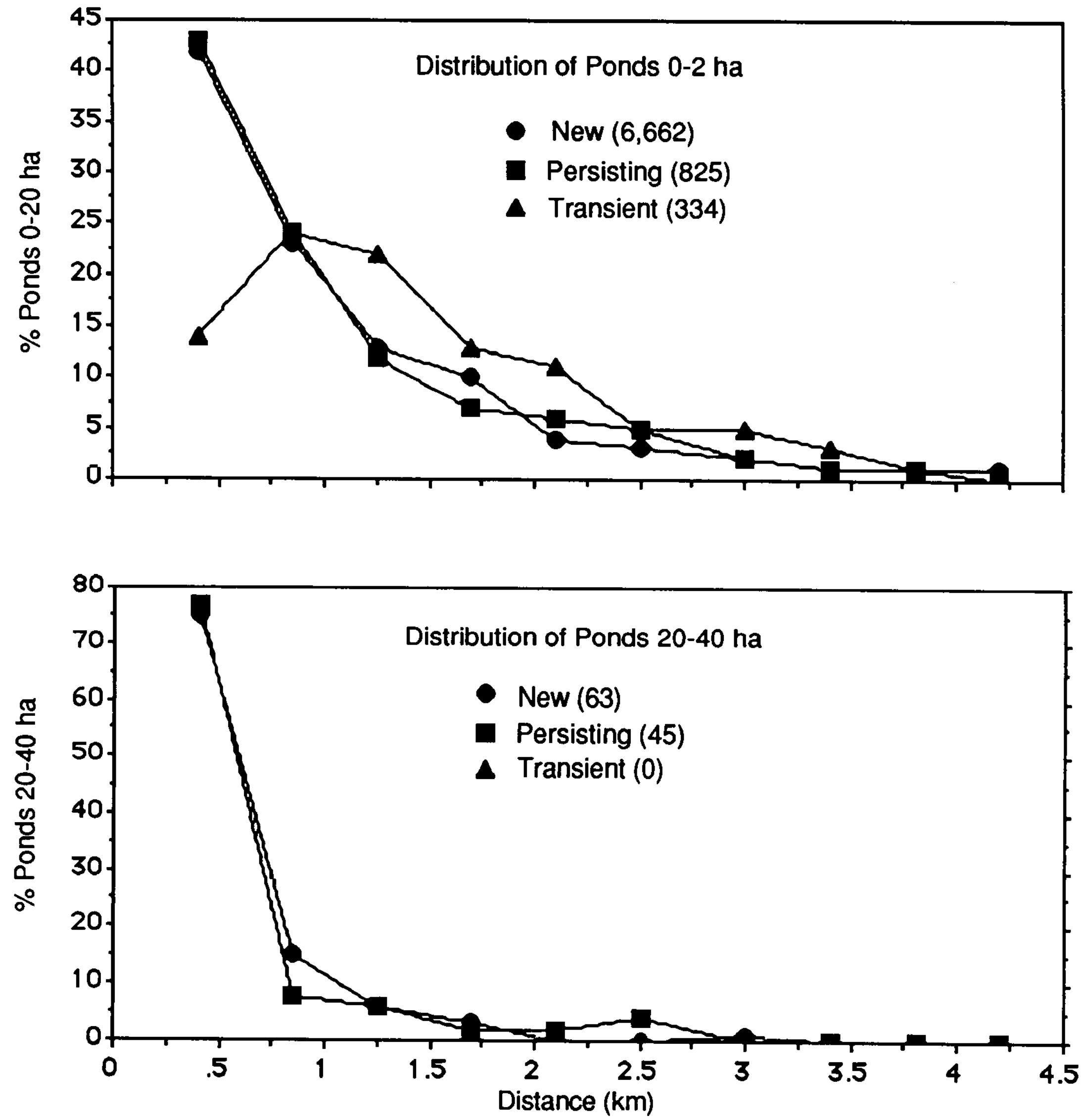

Figure 21-5. The distribution of new, persistent and transient ponds <20 ha and 0-40 ha within 4.2 km from 1978 canals for the study area. Zero distance is at the canal edge. The number of ponds in each category is in parentheses.

There was a positive relationship between number of new ponds formed and canal area for four study regions, but not for the Chenier Plain which had a relatively smaller range in values (Figure 21-6). The statistically-significant slopes were similar for the St. Bernard, Barataria and Terrebonne maps, but lowest in the map group with a portion of MRGO in them. Maps combined from all areas except the MRGO maps also had a significant relationship between the two variables. For each map the slopes are positive and the Y intercept is near zero when canal density is zero.

A better estimator of total open water area change involved summing all area of net change for ponds <60 ha. The relationship between pond area <60 ha and total open water formation in each quadrangle map was highly significant and therefore used as a surrogate for total wetland loss (Figure 21-7). The areal extent of these new ponds is well related to the net gain in open water, but the relationship appears to be curvilinear. Further, there is also a significant relationship between the net gain in area of ponds <60 ha and depth to the sediments deposited in the Pleistocene. A multiple regression model of pond formation for new pond area <60 ha (Y) vs canal area (C) and depth to the Pleistocene (D) yielded the following formula:

$$Y = 252.3 + 1.53\ (C) + 7.885\ (D)$$

with a coefficient of determination (R^2) of 0.71, where n=34 quadrangle maps including the St. Bernard delta and the Barataria and Terrebonne hydrologic units.

Discussion

New holes in the marsh are the dominant form of wetland loss, not erosion at the shoreline. Understanding the wetland to water conversion processes within the marsh, rather than at the marsh edge, is therefore extremely important. Map analyses are useful in this regards. But predicting the distribution, number, and size of these holes in area the size of 7.5 minute quadrangle maps is complicated by the spatial heterogeneity of the wetlands, the natural ridges, and the use of only 2 time periods for comparison. The choice of map scales must match the task at hand. In this case we were interested in reducing the variation due to mapping errors (which increase with a decrease in mapping units); minor manipulations in the marsh due to one individual digging a pirogue ditch, for example; marsh fires; and animal grazing (when severe, these are known locally as "eatouts"), among other influences. We also wanted enough data points to justify a statistical analysis of variations within a hydrologic unit. We understand that using a 7.5 minute quadrangle map size is a compromise between being practical and reaching a theoretical optimum; we chose to use them because we cannot clone this coast for regional-sized, controlled experiments. The data were available and never analyzed in this way before and the results seem informative.

At least five qualitative types of wetland changes are evident in these maps: (1) spoilbank-parallel hole formation, (2) pond formation with apparent random distribution, (3) semi- or complete impoundment and resulting open water formation, (4) cutting off of stream channels upstream of where a spoilbank crosses a natural channel, and, (5) erosion at the land-water interface. Only ponds <20 ha appear to form and disappear. This might be considered to be due to mapping errors; however, the large number (10 % of the total), a different distribution of the transient ones compared to the new ponds, and mapping of smaller features argue against accepting such biases as meaningful complications in the analysis.

New small holes are more numerous than persistent and transient water bodies of the same size. New area and number of small holes are directly related to the area and number of larger holes; if new small ponds form, it is likely that the less numerous larger ponds

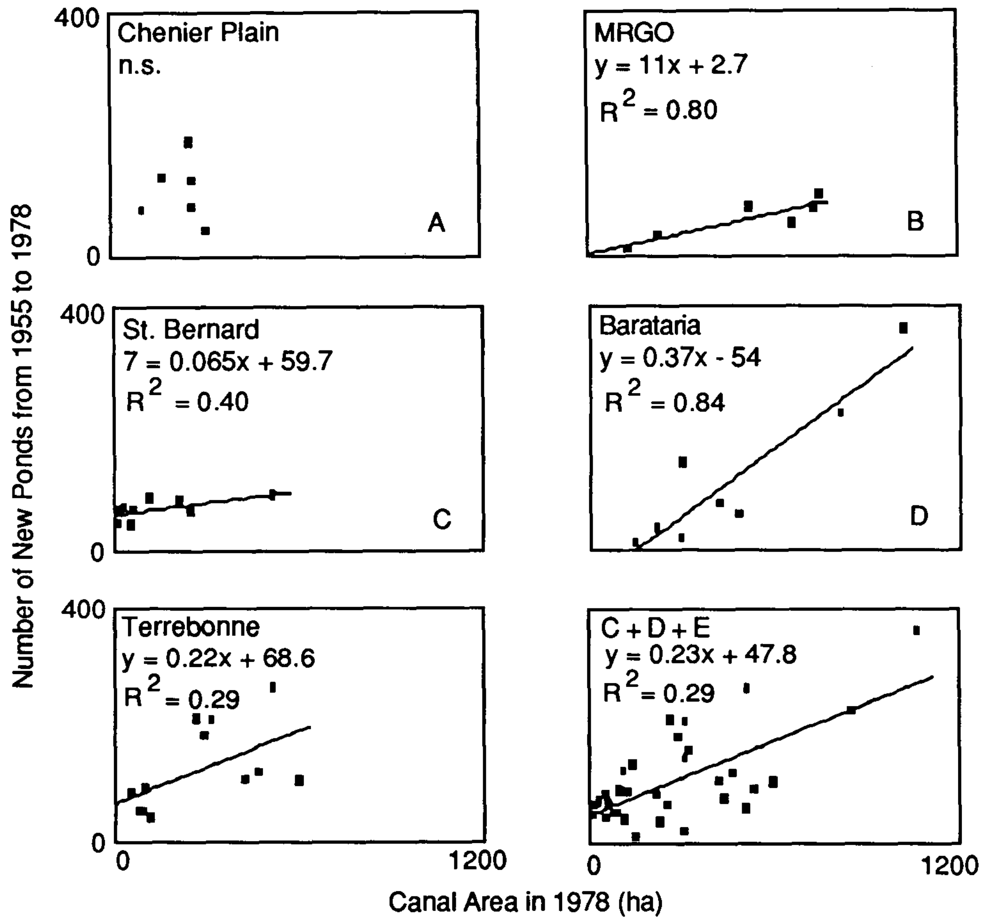

Figure 21-6. The relationship between the number of new ponds <20 ha formed between 1955/6-78 and canal surface area (ha) for four study regions: MRGO is the Mississippi River Gulf Outlet running north/south through the St. Bernard delta; St. Bernard is a former delta of the Mississippi River, on the eastern border of that river; Barataria is the hydrologic unit just west of the Mississippi River; Terrebonne is the hydrologic unit on the western border of the Barataria hydrologic unit; the Chenier Plain is the western one-third of the Louisiana coastal zone.

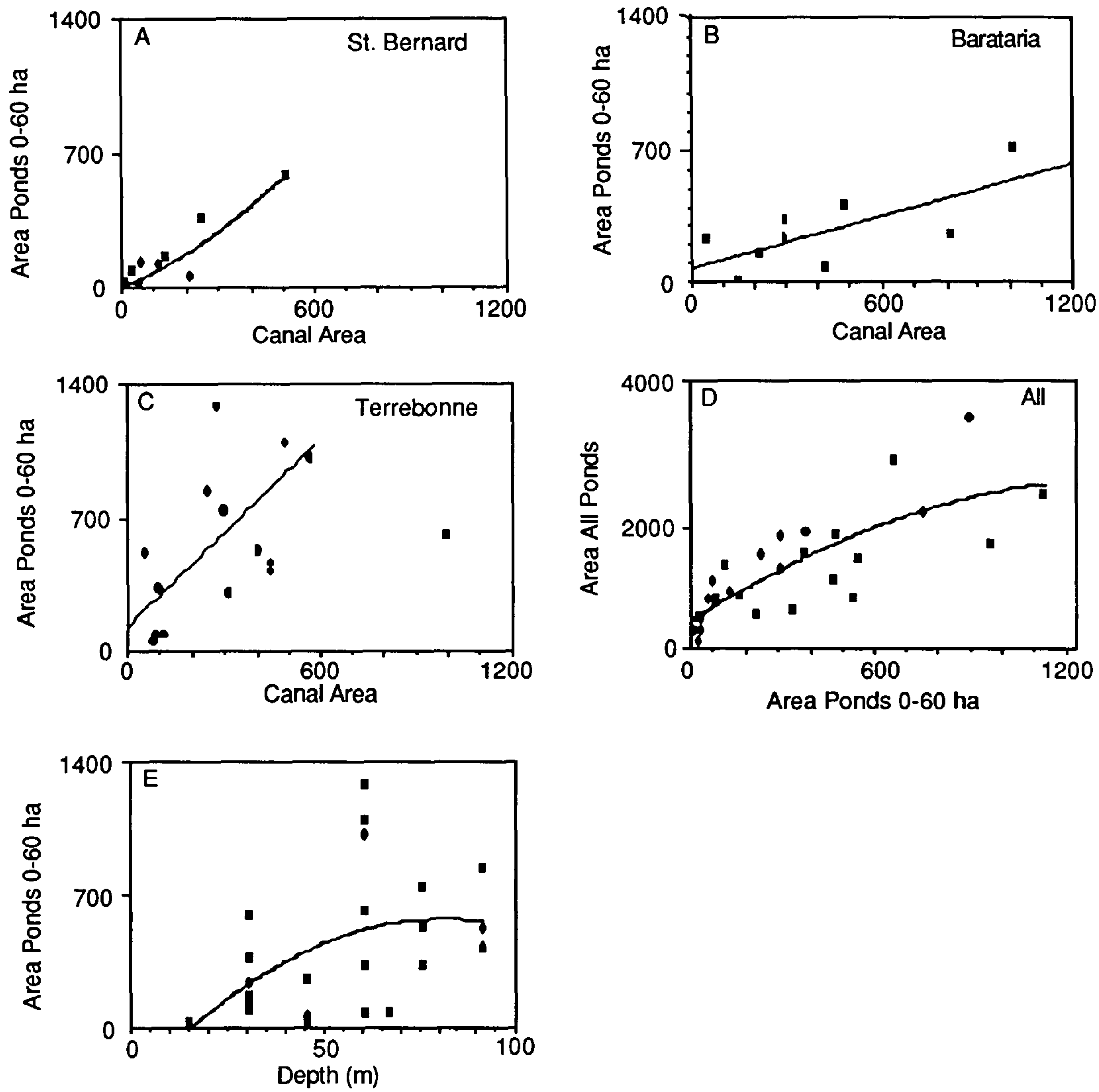

Figure 21-7. The relationship between the net gain in ponds <60 ha formed between 1955/6-1978 and canal surface area (ha) for the study region. St. Bernard is a former delta of the Mississippi River, on the eastern border of that river; Barataria is the hydrologic unit just west of the Mississippi River; Terrebonne is the hydrologic unit on the western border of the Barataria hydrologic unit. The area of this pond size is correlated with total open water gain in a curvilinear manner (21-7D) and linearly with depth to the Pleistocene Terrace (21-7E).

will also form. These patterns are consistent with the conclusion that the marsh is literally breaking up internally, rather than eroding at the edge. The relationship between small and larger pond number is less clear as pond size increases. This situation probably reflects, in part, the smaller number of the larger ponds, hence the smaller statistical base to work with. But, it also may reflect the regional (outside the quadrangle map) influence of geologic influences, canal and spoil banks, as well as agricultural impoundments (e.g., Figure 21-4), urbanization, navigation channels, and local sediment sources. These relationships complicate interpretation of wetland changes at the landscape level and an analysis of the spatial and temporal distribution of the small new ponds is probably more useful than an analysis of the few larger ponds. An analysis of areas of particularly high or low wetland loss was also useful to understand regional differences or anomalous conditions (e.g., in the Cutoff quadrangle).

Given the relationships between net areal change for ponds <20, ponds <60, ha and all other pond sizes, we conclude that analyzing the pond number and area for net change in ponds < 20 ha and <60 ha, respectively, is indicative of the wetland losses in that map. This is a generalization, and local exceptions were revealed. Other, more useful combinations, also found to warrant further analyses.

Canals and their spoil banks are directly related to wetland to water conversion and it is evident about 2 kms away from those canals. This is clear from the distance versus frequency plot for two different pond sizes, by visual examination of the many maps of pond formation, and from the relationships between pond formation and canal density whose intercept is near zero.

New ponds between 0-20 ha tend to be no more numerous next to the 1955/6 canals than to 1978 canals, when compared on a number per surface area basis. We cannot explain this, but it is apparent that whatever effect the canal and spoil bank has on wetland loss, that for these maps it does not clearly diminish or gain influence with age, to date.

The implication of these relationships is that there is either a common factor relating canal area to the net change in land to open water or the relationship is spurious. We assume it is not the latter because of the high number of data points (>7,800 new ponds; 72 maps), good areal coverage, and known mechanisms to explain the apparent coupling (see other chapters in this report, including Chapter 3).

We suggest, therefore, that these results support the hypothesis that the hydrologic impacts of canals and spoil banks affect wetland to water conversion 2-3 kms away from canals, are directly related (as a causal agent) to the majority of wetland losses in the study area, and their impacts vary regionally, e.g., with sediment compaction rates that increase with increasing sediment deposition. Local influences complicate the interpretation and not all areas will be same. This is a regional study, not a local study, so the results apply only on a broad scale. This study area includes major sections of all the regional groupings described in Chapter 19 and does not differ substantially from the general results of that modeling study in that regional differences and the same influences of geological substrate are demonstrated influences on the overall regional wetland loss rates. It does not include all parts of some areas near the Atchafalaya River or the western side of Bayou Lafourche (study sites for Chapter 20). We know of no data analysis which contradicts the conclusion that the marsh is breaking up rather than eroding, except at barrier islands.

Chapter 22

LANDSCAPE PATTERNS WORKING GROUP: CONSENSUS REPORT

by
James H. Cowan, Jr.
John M. Hill
Scott G. Leibowitz
R. Eugene Turner

Interpreting the results of these three projects is complicated because we traded site-specific analyses of one process or function for broad analyses which (because of scale) is the result of the individual processes working together in all areas. We recognize that we are glossing over locally significant influences in the discussion below and that we have neglected discussing the relatively less common influences, such as agricultural and urban impoundments, refuge management and navigation channels.

Four patterns in habitat change are apparent: (1) differences in regional geology are significant influences affecting habitat changes; (2) man-made factors, including agricultural and urban development and canals and spoil banks, are spatially related to some of these changes; (3) the regional salinity changes did not influence the rate of habitat conversion to open water in some areas but may have affected the rate in other areas; and, (4) most of the widespread habitat changes are more appropriately described as inland fragmentation or loss, not lateral erosion at the shoreline.

Factors that correlated with either wetland or landloss rates include the age and thickness of previously deposited sediments, distance to sediment sources and freshwater, indicators of hydrologic change, such as canals and spoil banks, and various measures of canal and spoil bank influence (e.g. distance and density). In general, loss rates are lower where sediments are thin, the delta lobe age is high, spoil banks, canals and the seashore are far away, and rivers are close. The reverse is also generally observed: loss rates tend to be high where sediments are likely to be consolidated, where allochthonous sediments are in shortest supply, and where canals and spoil banks are dense. Loss rates in specific areas may not always follow the general tendency, especially in areas of particularly high land loss. The interpretations explaining these relationships are minerals that sustain and build wetlands come from rivers, compact under pressure and with time, and are more likely to erode where wave energy is high.

Strictly speaking, because these results are based on correlation analysis, we cannot demonstrate cause-and-effect relationships. However, based on the common appearance of all factors in the three studies, results from the scientific literature and the lack of a more efficacious explanation (the application of Occam's razor), we accept that all factors identified above are significant factors driving habitat changes from land or wetland to open water.

Because of the interactions between factors, regions differ. In terms of loss rates and the relationship between these factors, the Chenier Plain is distinct from the Deltaic Plain. Similarly, the Deltaic Plain can be subdivided into smaller regions. Boundary definition is important in analyzing landscape changes, and fortunately all three studies analyzed areas at appropriate scales, which are here identified for the first time.

The quantification of the influence of these factors on wetland loss was a goal of these three studies. The working group accepts the role of geologic factors as primary. Whatever effect sediment supply and distribution or canals and spoil banks have, it happens within a regional framework largely defined by geomorphology. We found evidence to support the hypothesis that man-made factors, as modified by the geologic factors, are also significant (>10%); but, the evidence was not evenly developed so that differences of opinion developed when quantifying these impacts, and these differences vary depending on how much reliance is put in analyses of partial data sets, qualified assumptions, correlation as a substitute for experimentation and incomplete analyses.

What is the percent of wetland loss caused by canals and spoil banks? The amount from the direct result of dredging, as well as agricultural and urban projects is discussed in Chapter 4. It is the indirect impacts with which we are concerned. We are not willing to give a definitive reply but can discuss a range of possibilities based on only the results in these three studies. The stronger statements are in individual chapters.

The statistical analysis of quadrangle maps, by region, resulted in a projected decrease in wetland loss by 15%, if no canals were present (the range for hydrologic units = 4 to 51 %). The analysis of pond formation within selected regional groupings of quadrangle maps had loss rates near zero at zero canal area. These two results are not precise estimates, however, and are based on hindcasting with considerable variation about the estimate. Other results indicate that 40 to 80% of gross land loss within selected regions can occur in areas without canals or spoil banks.

Chapter 23

PROJECT CONSENSUS

The major questions upon which the project was organized (see Chapter 1) provide the focus for project consensus. Consensus opinion was derived from the orderly synthesis of data and ideas, beginning with the individual Technical Approaches and proceeding to discussion and integration at the working group level. Project consensus was achieved through the input of all working groups and project members.

Question 1

If land is sinking more quickly than land is building and the rates of each process are changing, to what extent is this disparity caused by changes in (1) sediment supply reaching the marshes; (2) organic matter accumulation; (3) subsidence rates; and, (4) water level rise?

Answer 1

In the face of regionally high but historically (80 years BP) non-accelerating rates of relative water level rise, low rates of inorganic sediment accumulation coupled with annual fluctuations in basin water levels have led to biologically significant periodic (20 to 25 years), and perhaps longer-term, disparities between land building and water level rise processes.

Relative water level rise (RWLR) results from geologic subsidence (compaction) and water level rise (sea level and basin water level changes). Land building occurs through the aggradation and accretion of matter, both organic and inorganic. Disparity can be defined as the difference between accretion and RWLR and has recently increased within the Louisiana coastal zone. This study is concerned with cases in which accretion is less than RWLR. An example of a surface disparity between RWLR and accretion is described in Eqn 17.1 below.

R = relative water level rise = 1.2 cm/yr
A = vertical marsh accretion = 0.7 cm/yr

and

$$\text{Disparity} = \quad \begin{aligned} Dis &= R - A \\ Dis &= 1.2 - 0.7 \\ Dis &= 0.5 \text{ cm/yr} \end{aligned} \qquad \text{Eqn 17.1}$$

The four factors mentioned above have different relationships with each other. Geologic subsidence is independent of marsh processes as is water level rise. Subsidence and water level rise affect the marsh through their sum, i.e., RWLR. The organic component of marsh accretion is strongly related to and controlled by the inorganic component. In other words, the marsh standing crop is dependent upon the bulk density of soil, given as g/cm^3. For example, salt marsh plant growth has been shown to become very stressed at a minimum bulk density, i.e., ≈ 0.2 g/cm^3. The standing crop increases proportionately as bulk density increases so that a maximum bulk density of approximately 0.4 g/cm^3 is reached. The salt marsh is viable when the bulk density is between 0.2 to 0.4 g/cm^3 and when its surface elevation is in the optimum flooding range of *Spartina alterniflora*. The salt marshes we investigated are at a minimum bulk density needed to support marsh vegetation because of the increase in the rate of water level rise and because the aggradation rate appears too low to maintain an adequate bulk density, at least during the 1986-87 time period.

The relationship between the organic and inorganic components of brackish and fresh marshes is less clearly understood, although these marsh types require certainly no more and most likely less inorganic sediment to maintain plant production than salt marsh systems. Therefore, the low rates of mineral accumulation and bulk densities measured in these marsh types are probably more adequate at maintaining plant standing crop. How adequate is not known, however.

Vertical marsh accretion is related to marsh aggradation (mineral and organic matter accumulation) because the inorganic and organic components of the marsh are correlated. The relationship can be expressed as:

$$S = \rho \cdot A \qquad \text{Eqn 23.1}$$

where S is the marsh aggradation rate (mineral and organic accumulation) in g/cm^2/yr, A is vertical marsh accretion in cm/yr, and ρ is bulk density, with values 0.2 to 0.4 g/cm^3. Therefore, disparity also can be described in terms of S (marsh aggradation) if the bulk density of the soil is known.

R = relative water level rise = 1.2 cm/yr
ρ = soil bulk density = 0.3 g/cm^3
S = marsh aggradation = 0.1 g/cm^2/yr

and

$$DIS = R - \frac{S}{\rho} \qquad \text{Eqn 23.2}$$

$$DIS = 1.2 \text{ cm/yr} - \frac{0.1 \text{ g/cm}^2\text{/yr}}{0.3 \text{ g/cm}^3}$$

$$DIS = 0.87 \text{ cm/yr}$$

For example, given a bulk density of 0.3 g/cm^3 with a submergence of 1.2 cm/yr and an aggradation rate of 0.1 g/cm^2/yr, then the vertical accretion rate is 0.33 cm/yr and the disparity is 1.2 - 0.33 = 0.87 cm/yr. The minimum aggradation rate (mineral and organic accumulation) needed to produce a vertical marsh accretion rate of 1.2 cm/yr at a density of 0.3 g/cm^3 would be 0.36 g/cm^2/yr.

$$S = 0.3 \text{ g/cm}^3 \cdot 1.2 \text{ cm/yr}$$
$$S = 0.36 \text{ g/cm}^2\text{/yr}$$

Therefore, the disparity expressed in terms of marsh aggradation is 0.36 - 0.1 = 0.26 g/cm^2/yr.

A schematic representation of these concepts is presented in Figures 23-1 and 23-2. Relative water level rise is defined in Figure 23-1 as the difference between the water level and subsidence. In Figure 23-2 accretion and disparity are shown compared with RWLR. The accretion rate increases after the water level rise rate increases but at a rate less than that needed to equal RWLR.

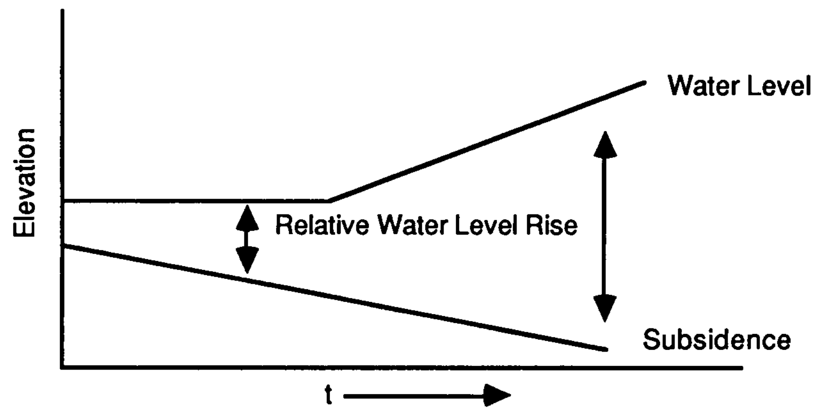

Figure 23-1. Relative water level rise defined as the difference between water level rise and subsidence.

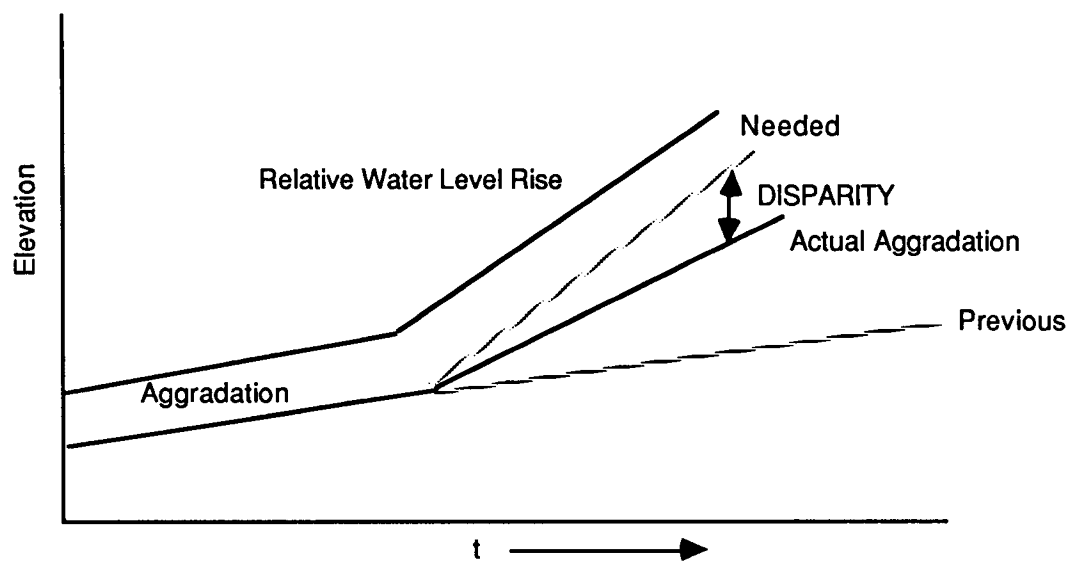

Figure 23-2. Relative water level rise, aggradation, and disparity for constant inorganic sedimentation.

Question 2

Do levee construction, canal dredging, and oil and gas production influence the rates of sedimentation, organic matter accumulation, and subsidence in coastal Louisiana? If so, do these impacts contribute to the high rate of coastal submergence?

Answer 2

The flood control levees of the Mississippi River have reduced the supply of sediment directly available to the marshes by over-bank flooding (a quantity equal to, on average, 3% of the total annual suspended sediment load). Spoil banks associated with man-made canals have a clearly defined direct effect on compaction of the marsh surface but a far less clearly defined influence on marsh aggradation (mineral and organic matter accumulation) caused indirectly by changes to local hydrology. Oil and gas production (i.e., fluid withdrawal) have a direct, potentially significant but local effect on subsidence in coastal wetland habitats. All of these factors contribute, in varying degrees, to submergence of the wetlands.

Flood control levees on the lower Mississippi River restrict the supply of sediment to the marsh through overbank flooding. Not only have the sites at which overbank flooding occurs been eliminated, but damming and flood control structures in the upper river basin have trapped sediment. The levels of suspended sediment and the size of the suspended particles carried by the river have decreased during this century. Thus, both the quantity and quality of sediments carried by overbank flooding to the marsh, where such events are allowed to occur, have been altered. The result is decreased mineral sediment supply to the marsh, less aggradation directly caused by this sediment reduction, an altered nutritional value of the resultant marsh substrate, and the potential for reduced plant growth and organic sedimentation.

Spoil banks for canals and navigation channels cause the underlying marsh to compact. This lowers the marsh level on the marsh side of the spoil bank, and often results in pond formation. Those ponds may enlarge with time in a region that does not have an alternate hydrologic connection into the marsh.

Spoil banks may alter the hydrology of the marsh. Partially or fully impounded portions of the marsh are often flooded less frequently than nearby natural marsh, but the flooding events are of longer duration. The marsh vegetation in salt, brackish, intermediate, and fresh marsh are all known to be sensitive to waterlogging stress. Higher mineral sediment content and bulk density immediately behind the levee (edge effects) and lower values farther into the marsh were found at the natural bayou study sites but not at the spoil bank sites. Aggradation rates in the marsh behind natural levees were higher, on average, than behind spoil banks. However, this difference is not statistically significant. In some cases lack of replicates precluded a statistical test (stable isotope technique), while in other cases low sample size and high variance resulted in a test with low power. If the difference in sedimentation rates is real, it would be botanically significant. However, the present analysis could not demonstrate that canals significantly affect sediment aggradation and accumulation.

At one OCS pipeline spoil bank site, notably high sedimentation rates (comparable to earlier published estimates from the immediate vicinity) were observed on the southern side of the bank. The spoil bank at this one site was extensive in the east/west direction. The observed sedimentation rates were much higher than those measured at a nearby north/south trending bayou site and several shorter east/west-trending canal sites. We interpret these results to indicate a blockage of the wind-driven storm surge flow associated with winter frontal passages.

The extra water introduced into bayous and canals during winter storm surges is derived from the lower estuaries and near-shore waters of the Gulf of Mexico. Those waters are, therefore, often saltier than those to which they are added. Thus, while spoil banks may reduce the occurrence of flooding, when winter, wind-driven storm surges do cause flooding, the associated water is of relatively high salinity, thus potentially increasing the resultant stress on the marsh vegetation. After the flood waters drain off, the remaining interstitial waters are expected to increase in salinity because of evapotranspiration unless they are diluted by rainfall or a freshwater flooding event. Unfortunately, we have no direct data to confirm this conjecture.

Local subsidence greater than 10 cm potentially can occur over shallow reservoirs because of fluid withdrawal and occurs over the lifetime of the reservoir. The coastal area of known shallow reservoirs with such potential is approximately 51,000 hectares.

Question 3

Are there spatial patterns of land loss and, if so, can these patterns be interpreted?

Answer 3

Yes, there are clearly discernible patterns of land loss in the coastal landscape. Even though there is no one primary factor highly correlated with the spatial patterns identified, analysis of several factors in combination does allow for interpretation of these patterns.

The deltaic and chenier plains of southern Louisiana are the result of the shifting position of the Mississippi River and the construction and subsequent abandonment of a series of deltaic lobes. The southwestern coast was constructed from fluvial sediments and organic materials of the Mississippi River and other local rivers reworked by marine processes. Rates of erosion and subsidence within the deltaic plain are correlative with the time of abandonment of the delta lobe. During the last decade, the major area of new land formation has shifted from the main Mississippi River channel and delta front to the Atchafalaya River distributary. The decline in new land build-up along the Mississippi River and around the delta front may be related to the 60% decline in suspended sediment load carried by the river. A similar decline in bedload has not been documented but is highly likely.

There are numerous other examples documenting differences in the spatial pattern of land loss in coastal Louisiana. From a coastwide perspective, there is no one primary factor which is highly correlated with all the spatial patterns identified. There is a combination of primary factors, which include man-induced alterations (MIA) and geology (age and depth of soils). There are three distinct regions (clusters) where these factors interact differently. South central Louisiana has young, thick sediments with relatively low canal densities and high land loss (36%). This area is perhaps most sensitive to new MIAs. The Chenier Plain has older, relatively thin sediments with many MIAs and moderately low land loss (20%). It does not appear to be very sensitive to the indirect effect of MIAs. The region east of the bird-foot delta has moderately old, thick sediments with moderate land loss (22%) and moderately high canal densities.

One cannot accurately predict spatial patterns of coastwide land loss because they are dependent on the interactions of geology and MIAs. These factors are multiplicative, nonlinear, and the coefficients vary geographically in magnitude. The interaction of these factors within hydrologic units, however, does seem to be moderately predictable.

It appears that the spatial data we examined exhibit significant spatial patterns at any scale. Some of these patterns are summarized in Table 23-1. Direct impacts are well documented, and many other causes of land loss are inferred. Other spatial patterns are, however, unexplained, perhaps because of a lack of measurements or a lack of understanding of the cumulative relationships. These particular areas are "hot spots" of contiguous land loss that are small in aerial extent but account for up to 40% of regional loss.

Table 23-1. Patterns of land loss in the Louisiana coastal zone by spatial unit, scale, and potential mechanisms.

Spatial Unit	Scale	Potential Mechanism
Coastal Zone	10,000s km^2	Delta decay, major MIAs[a], cumulative impacts of minor MIAs
Regional/Hydrologic Unit	10,0002 km^2	Subsidence, seal level rise, distance to sediment sources, sediment depth and age,damming of distributary channgels, major MIAs
Sub-delta Lobes	100s km^2	Proximity to coast, sediment age, sediment depth, distance to sediment sources, major MIAs
Quadrangle Maps	10s km^2	Local geology, distance to channels and canals
"Hot Spots"	10-100s km^2	Local geology, new pond formation, impounding, canal density, shoreline erosion, fluid withdrawal, point source erosion
Pixel	<1 km^2	Variation in marsh accretion, minor topographic effects

[a] Man-induced alterations (i.e., canals, agriculture, spoil banks).

There are many generalized, preconceived patterns of land loss that this project has helped clarify or refute. Three major generalizations, contrasted with our findings, are presented here:

	Hypothesis	Finding
1.	The majority of land loss is from the coast inward and uniform.	Land loss is actually internal to the marsh and its rate is highly dependent on site-specific factors.
2.	Saltwater intrusion is a major contributing contiguous factor coastwide.	Saltwater intrusion is (at most) localized.
3.	Man-induced alterations are a major contributor to coastwide land loss.	The impact of man-induced alterations is not uniform.

In summary, it appears as if man-induced factors are operating on the scale of geologic processes coastwide, and their influences can occur on a time scale of decades, a time considerably shorter than the normal geologic period. Consequently, land loss in geologically eroding areas is accelerated by man's activities. For example, the rapidly constructed levees along the Mississippi River have contributed to land loss on a geologic scale in a matter of 30 to 40 years.

Question 4

How long does it take for a change in subsidence, sedimentation or accumulation, i.e., surface disparity, to be expressed as land loss?

Answer 4

An annual surface disparity rate of 0.5 cm/yr could result in a surface disparity great enough to significantly reduce plant growth in salt marsh systems to the point where the viability of the marsh is in jeopardy in 20 years.

The time interval needed for the occurence of a significant effect on the marsh vegetative community caused by a surface disparity depends on the rate of relative water level rise, marsh vertical accretion rate, marsh type and species composition, and interacting local abiotic and biotic variables. An analysis of this surface disparity for a specific time interval was conducted. We selected the period from 1962 to 1980 for this analysis because (1) it is a period of high observed land loss; and, (2) data for marsh aggradation by ^{137}Cs dating and estimates of water level rise from tide gauge records were available for this interval.

Although relative water level rise exhibited variable rates during the past 80 years, the period from 1962 to 1980 showed a rate of 1.2 cm/yr (see Table 23-2). This value is a representative average for salt marshes across the coastal plain of Louisiana. The average aggradation rate in this area is 0.7 cm/yr, resulting in a surface disparity of 0.5 cm/yr. Over a 21-year period, this will produce a 10-cm water level/marsh surface disparity, which can significantly reduce plant growth. This statement is supported by the salinity/submergence experiments reported in this study that showed decreases in biomass of 75% following a rapid 10-cm decrease in marsh surface elevation in salt marshes over one growing season. Based on this vegetative response to increased flooding level, it is probable that some marsh deterioration would occur during this period and most certainly if the surface disparity continued. If one assumes a similar situation in fresh and brackish marshes, these habitats would be similarly affected. However, because of the greater species diversity in fresh marshes and the variable flood tolerance of fresh marsh plant species, the effect of increased flooding level on this habitat will be modified by species composition. If the surface disparity is also accompanied by a biologically significant rise in salinity, then the rate of marsh loss will likely be accelerated in fresh and brackish, but not salt, marshes.

We do not know just how well a marsh can adapt to decade-long disparities between water level rise and sedimentation. Also, a ten-year disparity may be temporary when viewed over 20 years or more. Water level rise during 1963 to 1982 was higher (two times) than that recorded during this century, but within the observed amount of variation; sedimentation rates for 100 years BP are nearly equal to that for 1963 to present. If (1) the marsh can adjust to temporary disparities or (2) longer-term water level rise records are more appropriate to use in these comparisons, then the surface disparity may be less severe than implied in Table 23-2. Future measurements of water level rise and marsh aggradation should clarify this issue.

Table 23-2. Comparison of apparent water level rise and marsh aggradation rate for various locations in Louisiana.

Area	Apparent Water Level Rise 1962-1983[a]	Aggradation 1963-present[b,c]	21-year Disparity
Coastal Louisiana	1.2 cm/yr	0.7 cm/yr	10.5 cm
Eastern Pontchartrain	0.15 to 2.16	0.54 to 1.07	none to 22.9
Mississippi Delta	1.8	1.8>2.0	none
Lower Barataria	1.8 to 1.91	0.68 to 1.2	12.6 to 25.6
Terrebonne Delta Plain	1.64 to 2.11	0.65 to 0.99	13.6 to 30.6
Vermilion	0.78 to 0.93	0.69 to 0.86	none to 5.0
Cameron	1.17	0.57 to 0.70	9.9 to 12.6

[a] From Penland et al., 1987.

[b] From DeLaune et al. (1987) and this study; based on ^{137}Cs horizon technique.

[c] We assume that the aggradation rate/year is relatively uniform with time during the period from 1963 to 1987.

Question 5

What are the direct and indirect impacts of OCS activities on wetland losses in coastal Louisiana?

Answer 5

The direct impacts of OCS activities account for 4 to 5% of the total Louisiana wetland loss from 1955 to 1978 (≈ 11,000 to 14,000 ha). Indirect impacts from OCS activities are estimated to account for 4 to 13% of all indirect impacts.

The total direct impact of OCS activities is documented in Chapter 4. The study team estimates that these direct impacts are approximately 4 to 5% of the total Louisiana wetland loss from 1955 to 1978, or approximately 11,000 to 14,000 ha. The vast majority of these direct impacts is caused by the conversion of wetlands into spoil bank levees and canals. A minor amount (38.5 ha) is caused by construction of facilities and less than 20% (up to 2,900 ha) is caused by navigation channels. In general, the impact per length of pipeline is highest in wetlands and lowest in non-wetlands. The total net change in canal and spoil area from 1955/6 to 1978 was 46,000 ha, equivalent to 16% of the total net wetland loss for the same interval. OCS pipeline and navigation channel direct impacts are in the range of 12,000 to 15,000 ha, or as much as 32% of the total net increase in spoil and canal area for the same period. The direct impacts of OCS construction are therefore a small percentage of the total direct impacts, but may represent a change of local importance or encompass a significant total area.

The indirect impacts of OCS pipelines, facilities, and navigation channels are more difficult to assess, but it is possible to establish ranges of impacts in relation to the total net change in wetlands. The amount of indirect changes caused by facilities is relatively unimportant because of the relatively small direct impact (38.5 ha). The indirect impacts by pipeline canals and navigation channels on wetland losses are considered the more important issues.

Navigation channels, especially OCS channels, are normally deeper, straighter, and wider than natural channels and encourage more frequent and severe saltwater intrusion events. We could not quantify the impact of increased salinity intrusion on wetland loss, even though it is thought to be locally significant in some cases. Consequently, we could not demonstrate whether navigation channels have a major indirect impact on wetlands on a coastwide basis.

Significant indirect impact can be attributed to the impacts of spoil bank levee and canal area. The rationale and mechanism for these indirect impacts are generally related to the hydrologic changes resulting from the spoil bank. We can pro-rate the indirect impacts of OCS spoil banks based on the minimum total indirect impacts of spoil bank levees of all kinds and the relevant percentage of OCS-related spoil bank levees. The OCS and non-OCS channels, canals, and spoil banks are not necessarily equal, however. For example, we know that their depth-to-width relationships, alignment, density, and uses are not exactly the same. Further, because of mapping scale issues, OCS pipeline canals and spoil bank levees may be more accurately mapped than the non-OCS equivalents and therefore be relatively over-represented. The numbers are estimated based on the best available data.

Correlative models of wetland loss and spatial analyses of land loss resulted in a consensus loss range of 20 to 60% for indirect impacts associated with all spoil banks and canals. We recognize that other geological and biological factors may contribute to these losses. We used spoil bank and canal area as a surrogate for indirect impacts and pro-rated their influence among OCS and non-OCS pipeline and navigation channels. Indirect impacts were assumed to be equal to the net change in wetlands from 1955/6 to 1978 minus net changes in that interval caused by (1) agricultural and urban development and (2) spoil and canal construction. Indirect impacts from OCS activities were thus estimated to be 4 to 13% of all indirect impacts. These numbers are necessarily based on interpretation of limited data and correlation, as opposed to cause-and-effect experimentation, and should therefore be used with caution to indicate only the relative magnitude of possible indirect impacts, and only on a regional, not local, basis.

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