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Kathryn Marie Flynn

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**Effects of hydrologic alterations on the vegetation and soils of
marshes in southern Louisiana**

Flynn, Kathryn Marie, Ph.D.

The Louisiana State University and Agricultural and Mechanical Col., 1992

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EFFECTS OF HYDROLOGIC ALTERATIONS ON THE
VEGETATION AND SOILS
OF MARSHES IN SOUTHERN LOUISIANA

A Dissertation

Submitted to the Graduate Faculty of the
Louisiana State University and
Agricultural and Mechanical College
in partial fulfillment of the
requirements for the degree of
Doctor of Philosophy

in

The Department of Oceanography and Coastal Sciences

by
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May, 1992

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ABSTRACT

Hydrology is the primary forcing function in wetlands. Therefore, alterations to hydrology, which can result from either natural or anthropogenic causes, can have dramatic effects on wetland structure and function. Structural marsh management, one type of anthropogenic alteration, employs levees and water control structures to reduce or reverse wetland loss and enhance the productivity of natural renewable resources. Its use is becoming increasingly popular in southern Louisiana as a solution to the problem of marsh deterioration despite the fact that very little comprehensive data is available on the effectiveness of this practice. Associated with the use of structural marsh management is the potential problem posed by hurricanes which can deposit large volumes of saline water inside management areas which are then retained by the encompassing levees.

The purpose of this dissertation was two-fold: first, to determine if structural marsh management improved primary productivity by reducing salinities and providing water level control; and second, to determine what factors governed the ability of an impounded freshwater marsh to recover from storm-induced exposure to high salinity waters.

In order to address the first goal, three pairs of managed and unmanaged marshes were monitored for one growing season during a drawdown year. Marsh pairs were located in the Fina LaTerre Mitigation Bank site, at Rockefeller Refuge, and at Avoca Island, Louisiana. Comparison of the three marsh pairs illustrated that while structural marsh management can effectively accomplish the goals ascribed to it, its use is not uniformly effective in improving soil and water level conditions and increasing primary productivity. It appears that the effectiveness of structural marsh management depends on location, design, marsh type, and sophistication of the plan used.

The second goal was accomplished by conducting a greenhouse experiment using mesocosms containing sods of intact freshwater marsh soil and vegetation. The mesocosms were exposed to a simulated saltwater intrusion event followed by establishment of a variety of recovery-phase salinity and water level conditions. Higher salinities during recovery exerted a strong influence on the rate and degree of vegetation recovery resulting in less regrowth and diminished species richness. This effect was exacerbated by flooding.

INTRODUCTION

Hydrology is the primary forcing function in wetlands (Mitsch and Gosselink, 1986). Alterations to hydrology, which can result from either natural or anthropogenic causes, can therefore have dramatic effects on wetland structure and function. Much of coastal Louisiana is experiencing flooding of increasing depth and duration due to a combination of natural and anthropogenic causes. Natural hydrologic alterations include marsh subsidence (Baumann, 1980) and the occurrence of eustatic sea level rise (Titus, 1986). Anthropogenic changes in hydrology have resulted from the construction of river and canal levees (which have also resulted in a sediment deficit) for flood control (Baumann et al., 1984) as well as the construction of canals, weirs, and plugs. The construction of numerous oil and gas canals and deep-water navigational channels, which can serve as conduits for saline water, may mean that many fresh and intermediate marshes will be subject to increases in salinity (Wang, 1987) and/or flooding.

According to Dunbar et al. (1990), the average land loss for the Louisiana coastal zone as of 1983 was 30.71 square miles per year. Dahl (1990), reported that Louisiana has lost approximately 46% of the wetlands present 200 years ago. Presently, large-scale human-induced modifications in the hydrology of many Louisiana marshes are occurring due to what is called structural marsh management. This is a process by which levees and water control structures are used to control hydroperiod and thereby reduce or reverse wetland loss and/or enhance the productivity of natural renewable resources.

The use of structural marsh management is becoming increasingly popular in southern Louisiana. Cowan et al. (1988) predicted that, at current rates of construction, 30-40% of Louisiana coastal wetlands will be impounded by 2020. In fact, by 1989, 9% of all Louisiana wetlands had permits issued for marsh management (Cahoon et al., 1990). Intentional alteration of hydrology via structural marsh management is in marked contrast to the occurrence of unintentional impoundments resulting from such causes as

the intersection of canals. In southern Louisiana, structural marsh management is being used to address a diverse range of objectives, including increasing waterfowl and furbearer use (Craft and Kleinpeter, 1986), excluding salt water and promoting freshwater species, and preventing marsh loss (Simmering et al., 1989).

Implementation of these structures affords a unique opportunity for the investigation of effects of large-scale hydrologic modifications on marshes. It is important to determine if structural marsh management plans can meet their intended purposes of improving primary productivity through reduced salinities and better water level control. In addition, the need for development of effective and economically realistic monitoring standards and improvements on existing management techniques are essential.

In addition to the alteration of marsh hydrology by structural marsh management, managed marshes are subject to the introduction of large volumes of saltwater during storms--especially hurricanes. The levees which enclose managed areas prevent the rapid drainage of these saline waters back into adjacent bayous and canals. The control structures can be opened to allow drainage but, because of the levees associated with marsh management, saline water will remain on the marsh for longer periods of time than would occur in an unimpounded marsh. The possibility that impounded areas will be flooded with saline waters during a storm surge is very real in Louisiana where tropical storms and hurricanes are common in the summer and early fall.

The importance of U. S. wetlands as a national and international resource has only begun to be realized by many people. Wetlands functions are diverse and vary with wetland type, quality, and location. The provision of fish and wildlife habitat, storage of floodwaters and entrapment of sediments that could potentially pollute waterways, improvement of water quality, groundwater recharge, and the protection of shorelines from erosion are just a few of the functions that wetlands perform (Dahl, 1990). The importance of these wetland functions makes it essential that we develop a comprehensive knowledge of how wetlands perform these functions and whether or not structural marsh

management can be used to enhance or preserve these functions. It is especially important to determine if structural marsh management is a viable method for alleviation of problems resulting from increasing water levels and potential salt water intrusion.

This dissertation attempts to address these needs by determining the influence of structural marsh management on the soil-plant relationships in three south Louisiana coastal marshes. In addition, it addresses the influence of post die-back salinity and water level conditions on the recovery of freshwater marsh vegetation killed by a simulated saltwater intrusion event. The long-term ability of freshwater marsh vegetation to recover from temporary flooding with high salinity water due to storm surges associated with hurricanes had not been previously determined. The increasing use of structural marsh management which would allow retention of these saline waters on the marsh surface makes the information in this dissertation very important.

Chapter 1 includes a discussion of the basic wetland types in Louisiana and their characteristics. A synopsis of the problems currently facing Louisiana's wetlands--problems which may ultimately confront coastal areas in other parts of the United States and the world--is then presented. The chapter concludes with a discussion of marsh management and the potential problem of storm surges which may transport large volumes of saline water into managed areas.

Chapters 2 and 3 document the effect of two marsh management programs, one at Rockefeller State Wildlife Refuge and Game Preserve located in southwestern Louisiana and one at the Fina LaTerre Mitigation Bank located in south-central Louisiana, on the soil-plant relationships of these brackish marshes. At both of these locations emergent primary productivity and soil conditions in a managed and unmanaged brackish marsh were compared in order to determine the effect of the management regime.

In Chapter 4, the effect of a new management area on a fresh floating marsh at Avoca Island, Louisiana is compared to an adjacent unmanaged marsh. The data in this chapter is unique in that, for the first time, the effect of management on the vegetation and soils of a floating marsh are reported.

The ability of intact freshwater marsh mesocosms to recover after emergent vegetation was killed by exposure to a simulated saltwater intrusion event is presented in Chapter 5. This information provides insight into the factors that would control recovery of a managed freshwater marsh if it were inundated by saline waters during a storm surge. Three post-die-back salinity levels and either drained or flooded conditions were imposed on the mesocosms and the extent of recovery under each set of conditions is reported.

Chapter's 2 and 3 are included as a portion of a report entitled "A Study of Marsh Management Practice in Coastal Louisiana. Vol. 3 Ecological Evaluation" edited by D. R. Cahoon and C. G. Groat and published by the U. S. Department of the Interior, Minerals Management Service in December, 1990. Chapter 4 has been submitted, with minor changes, as a report to the U. S. Environmental Protection Agency. Chapter 5 is being prepared as a report to the Sea Grant Program. All of these chapters are presented in journal format in preparation for submittal for publication in refereed journals.

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CHAPTER 1

REVIEW OF LITERATURE

Much of coastal Louisiana is experiencing flooding of increasing depth and duration due to a combination of natural and anthropogenic causes. Natural causes of this altered hydrology include world-wide sea level rise (Titus, 1986) and natural subsidence (Baumann, 1980). Anthropogenic causes of altered hydrology include the construction of river levees (which has also resulted in a sediment deficit) for flood control (Baumann et al., 1984) as well as the construction of weirs, plugs, and other structures. Alterations to hydrology, the primary forcing function in a wetland, can have dramatic effects on wetland structure and function. The construction of numerous canals and deep-water channels, which can serve as conduits for saline waters, may mean that many fresh and intermediate marshes will be subject to increases in salinity (Wang, 1987).

One method currently being used to address the problems of marsh deterioration and salt water intrusion in southern Louisiana is the use of structural marsh management, a process by which levees and water control structures are employed to reduce or reverse wetland loss and/or enhance the productivity of natural renewable resources. The use of these structures results in altered hydrology with the goal of promoting plant growth and improving soil conditions. Intentional alteration of hydrology via structural marsh management is in marked contrast to the occurrence of unintentional impoundments resulting from such causes as the intersection of canal spoil banks. In southern Louisiana, structural marsh management is being used to address a diverse range of objectives, including increasing waterfowl and furbearer use (Craft and Kleinpeter, 1986), excluding salt water and promoting freshwater species, and preventing marsh loss (Simmering et al., 1989).

Structural marsh management employs a combination of water control structures and levees in order to control water levels. Typically, water levels are manipulated in order to promote the growth and seed production of emergent vegetation. This is often done in order to provide food for waterfowl. This is accomplished by initiating a drawdown (i. e. lowering the water level below the soil surface) during the growing season approximately every third year. Water levels are maintained at or near the marsh surface during nondrawdown years.

The importance of U. S. wetlands as a national and international resource has only begun to be realized by many people. The functions of wetlands are diverse and vary with wetland type, quality, and location. These functions include provision of fish and wildlife habitat, storage of floodwaters and entrapment of sediments that could potentially pollute waterways, improvement of water quality, groundwater recharge, and the protection of shorelines from erosion (Dahl, 1990). The importance of these functions makes it essential that we develop a comprehensive knowledge of how wetlands perform these functions and whether or not structural marsh management can be used to enhance or preserve some of these functions. It is especially important to determine if structural marsh management is a viable method for alleviation of problems resulting from increasing water levels and potential salt water intrusion.

Louisiana's wetlands face a wide and complex array of problems. These problems include the occurrence of both natural and anthropogenic hydrologic alterations. Natural hydrologic alterations will result from predicted global warming, eustatic sea level rise, and rapid subsidence which will exacerbate apparent sea level rise. Anthropogenic hydrologic alterations include man-made impoundments, sediment deprivation due to levee construction, and the construction of deep-water canals and channels. Often, several of these problems will affect a single wetland area making the issue of wetland

degradation very complex. Some or all of these problems may eventually affect other wetlands in the United States or other parts of the world if predictions of sea level increases are realized.

Whether the causes of hydrologic alterations are natural or anthropogenic, the results are the same: increasing levels of inundation, the possibility that salt water intrusion will occur, and the loss of Louisiana's wetlands--an important national resource. It is extremely important that we come to a better understanding of the causes of wetland degradation and develop management procedures to reduce the rate at which loss is occurring.

TYPES OF WETLAND VEGETATION

Wetland vegetation in coastal Louisiana is classified into several different groups based on salinity tolerance, location, hydrologic regime, and growth form. This section will briefly describe the natural ecology of each group moving from saline, brackish, intermediate, and fresh marshes into cypress-tupelo swamps and bottomland hardwoods. A synopsis of data concerning the effects of different methods of impoundment--including structural marsh management--on wetland vegetation will be presented.

Characteristics of Naturally Functioning Wetland Types

Saline Marshes

Diurnal and semidiurnal tidal inundations that occur in salt marshes provide what Odum and Fanning (1973) termed a 'tidal subsidy'. This subsidy provides marsh vegetation with both a nutrient and a sediment source, removes toxic compounds from the soil, and contributes to reaeration of the soil horizon (Gosselink, 1984). The average duration of flooding in a coastal salt marsh is 12-16 hours/event. However, during

September and October, when gulf water levels are higher, the salt marsh may be inundated 80% of the time (Baumann, 1980). Salt marshes are also important sources of detritus which is exported into estuaries (Mitsch and Gosselink, 1986).

Sediment deposition is greatest along the stream banks resulting in the creation of higher ground in these areas (10-15 cm above inland marshes [Gosselink, 1984]). Streamside soils have larger sediment grain size, larger pore spaces and, therefore, better drainage and more aerated soil conditions. Because of these conditions, streamside areas, which are inundated almost as often as inland areas, are flooded for shorter periods of time (Gosselink, 1984). Growth of *Spartina alterniflora*, the dominant salt marsh species, is greatest along these banks, reaching heights of up to 1 meter. Plants inland of the streamside have reduced growth (heights of 60-80 cm) due to slower drainage and the development of more anaerobic soil conditions. Stunted *S. alterniflora* plants (height <50 cm) or dieback areas are found in depressions behind spoil banks (Smith, 1970).

Other species found in the salt marsh include *Distichlis spicata*, *Juncus roemerianus*, *Spartina patens*, and *Bacopa maritima*. These species must be able to withstand not only periodic inundation and anaerobic soils (as must all wetland vegetation), but they must also be salt tolerant. Average salinity in a salt marsh is 18 ppt (Chabreck, 1982). Although *S. alterniflora* and *D. spicata* are salt tolerant species, they do not require salt for growth. The degree of flood tolerance may be as important as salinity tolerance in the distribution of salt and brackish marsh species (Parrondo et al., 1978).

Brackish Marshes

Brackish marshes are subject to daily tidal action and exhibit a streamside effect, but since they are further inland from the Gulf, salinity averages only 8.2 ppt. The dominant brackish marsh species is *Spartina patens*. Other species include *Distichlis spicata*, *Scirpus olneyi*, *Spartina alterniflora*, *Juncus roemerianus*, *Eleocharis parvula*,

and *Scirpus robustus* (Chabreck, 1982). Inundation has been reported to occur from 75 times/year with an average duration of 50 hours/event (42% of year [Byrne et al., 1976]) up to 125 times/year with an average duration of 28 hours/event (40% of the year [Sasser, 1977]).

Intermediate Marshes

Intermediate marshes contain both halophytes and freshwater species, resulting in higher species diversity than in either brackish or saline marshes. Species include *Spartina patens*, *Phragmites australis*, *Sagittaria falcata*, *Bacopa monnieri*, *Paspalum vaginatum*, *Vigna luteola*, *Eleocharis* sp., *Scirpus olneyi*, and others. These marshes are inland of brackish marshes but do receive some tidal influence (Chabreck, 1982). Sasser (1977) reported 32 inundations/year with an average duration of 29 hours/event (26% of the year). Salinity averages 3.3 ppt and water levels are higher than in brackish marshes (Chabreck, 1982).

Freshwater Marshes

Fresh marshes are often found abutting or intermixed with forested wetlands. They are inland of intermediate marshes and normally are free of tidal influence (Chabreck, 1982). On a yearly basis, the length of time fresh marshes are flooded appears to be equal to the length of time more saline marshes are flooded. These flood events, which in fresh marshes result from wind tides and upstream runoff, are more infrequent and of longer duration lasting approximately 5 days (Gosselink, 1984). Salinity averages 1.0 ppt and mean organic content of the soil is 52% (Chabreck, 1982). Fresh marshes exhibit greater plant species diversity (without any one species being dominant) than any of the other coastal marsh types. Species include *Panicum*

hemitomon, *Sagittaria falcata*, *Eleocharis* sp., *Alternanthera philoxeroides*, *Spartina patens*, *Phragmites australis*, *Lemna minor*, *Hydrocotyle umbellata*, *Utricularia cornuta* and *Typha* spp. (Chabreck, 1982).

Floating Freshwater Marshes

A variation on the fresh marsh is the floating fresh marsh or flotant which consists of a thick, floating mat of roots and peat. The origin of this type of marsh is unknown, but two theories have been proposed: 1) the growth of emergent vegetation into an open water area and subsequent breaking-off of the buoyant mat (Russell, 1942) and 2) the disattachment of marsh from subsiding substrate due to marsh flooding and a lack of mineral sediment (O'Neil, 1949). Floating marsh is maintained by its own organic matter and the organic content of the soils may be as high 80% (Gosselink, 1984). Primary species are *Panicum hemitomon* and *Sagittaria* spp. (Bahr et al., 1983). The ability to float ensures that deep flooding never occurs and the water level is always near the marsh surface (Gosselink, 1984). The water under flotant usually averages 1 meter in depth throughout the year (Bahr et al., 1983).

Baldcypress-Tupelo Swamps

Baldcypress-tupelo swamps are characterized by extended periods of flooding and clayey soils (Bahr et al., 1983). Periodic flooding results in increased productivity in both baldcypress-tupelo swamps and bottomland hardwoods (Broadfoot, 1967; Broadfoot and Williston, 1973). The most poorly drained sites are dominated by *Taxodium distichum* with the abundance of *Nyssa aquatica* increasing on higher, better-drained sites (Bahr et al., 1983). Other species include *Salix nigra*, *Fraxinus caroliniana* and *Acer rubrum* var. *drummondii*. Understory species include herbs such as amaryllises, ferns, pennyworts and grasses (Conner and Day, 1976). Typically, baldcypress-tupelo swamps experience spring floods, a short dry period in the summer

and recurrence of flooding in the fall (Bahr et al., 1983). The presence of standing water during a large part of the year allows the growth of floating vegetation such as *Lemna minor*, *Spirodela polyrrhiza*, and *Riccia* spp. (Conner and Day, 1976). Water movement through these areas, via overland sheet flow from adjacent uplands, delivers both sediments and nutrients to the swamp. However, very little percolation of water through the saturated soil occurs and most of the water evaporates or runs off (Bahr et al., 1983). Germination of baldcypress and tupelo seed requires moist but drained soil conditions. Once germinated, the seedlings are sensitive to complete inundation.

Bottomland Hardwoods

Bottomland hardwoods undergo short annual flooding (lasting several weeks to a few months [Bahr et al., 1983]). During the times when standing water is not present, the water table is at or just below the soil surface and the soil is moist (Conner and Day, 1982). The species composition is richer than that of baldcypress-tupelo swamps and is dominated by *Acer rubrum* var. *drummondii* (red maple). Other species found in bottomland hardwood forests include *Quercus* spp., *Salix nigra*, *Ulmus americana*, *Acer negundo*, *Populus* spp., and *Cornus drummondii* (Bahr et al., 1983). Undisturbed bottomland hardwoods are more productive than undisturbed baldcypress-tupelo swamps (Conner and Day, 1976).

Types of Impoundments and Their Effects

Complete Impoundments

For the purposes of this discussion, the term 'complete impoundment' will be defined as an area completely surrounded by levees in which the only input of water is via

precipitation, the water level is raised above the marsh surface and excess water drains over a dam or spillway. Information is available on 2 swamps that are impounded--Lake Chicot Swamp and Lac des Allemands Swamp.

Lake Chicot Swamp is a two thousand acre recreational impoundment created in the winter of 1942-1943 by closing a dam across the spillway on Chicot Bayou (Penfound, 1949). The area was originally a baldcypress-tupelo swamp with water elm (*Planera aquatica*) and buttonbush (*Cephalanthus occidentalis*) also present (Eggler and Moore, 1961). The water depth after impoundment averaged 7 feet and could be closely controlled except during flood or drought periods. A drawdown of 5-7 feet was planned for each winter (Penfound, 1949). Between the spring of 1943 and February, 1945, the lake was invaded by the aquatic species *Nelumbo lutea* and *Cabomba caroliniana*, *Brasenia schreberi*, *Nymphaea odorata* and *Ceratophyllum demersum*. By 1949 tupelo had begun to disappear from the area (Penfound, 1949).

Eggler and Moore (1961) reported no reproduction of swamp trees after 18 years of impoundment. Although drawdowns occurred every year except 1952 and 1956 (in order to control aquatic vegetation), any tree seedlings that germinated during the drawdown period were killed by reflooding. In addition, the mortality of baldcypress, which was 3% during the first 4 years of impoundment, had increased to 50% after 18 years. Tupelo, with only 28% survival, had an even greater mortality rate. In contrast, the diversity of aquatic vegetation had increased to include *Elodea densa*, *Eichhornia crassipes* and *Alternanthera philoxeroides* as well as the species present in 1949 (Eggler and Moore, 1961).

The natural hydrology of the Lac des Allemands Swamp had been altered for at least 20 years when Conner et al. (1981) conducted their investigation. Water levels, which fluctuated with precipitation and drought events, had a maximum depth of approximately 1 meter. Cyclic overland flow was prevented. When compared to a natural area and an area in which water levels were seasonally manipulated (a crawfish

pond), the permanently flooded area had the fewest trees and the lowest basal area. Species, such as ash, which are intolerant to flooding, were dying out and there had been no successful recruitment of baldcypress or tupelo due to constant flooding of the substrate. The development of an open overhead canopy had provided optimum conditions for the growth of aquatics such as duckweed, *Azolla*, *Wolffiella*, *Wolffia* and *Eichornia crassipes*. The dominant shrubs were buttonbush, snowbell and red maple. Net primary productivity was reported to be 886.7 g/m²/yr as opposed to 1,166.2 g/m²/yr in a natural area and 1,779.9 g/m²/yr in the crawfish pond.

Passively Managed Semi-impoundments

Passive structural marsh management of wetland areas is accomplished through the use of non-adjustable water control structures to create semi-impoundments. Fixed-crest weirs are usually constructed such that water levels are allowed to drain no more than 6 inches below marsh level (Chabreck and Hoffpauir, 1962) although variable-crest weirs and earthen dams may also be used (Cowan et al., 1988). The purpose of these structures has traditionally been to improve wildlife habitat in coastal marsh areas i.e. promote growth of aquatic vegetation such as *Ruppia maritima* (widgeongrass) and *Myriophyllum spicatum* (Eurasian milfoil) which are excellent waterfowl food sources (Larrick, 1975). Fixed-crest weirs have also been used to create or maintain freshwater marshes in intermediate or brackish marsh areas such as in the Rockefeller State Wildlife Refuge and Game Preserve in southwest Louisiana (Joanen and Glasgow, 1965; Wicker et al., 1983).

Meeder (1987) observed that the best stands of *Ruppia* sp., *Eleocharis parvula* and *Echinochloa crusgalli* on the Rainey Wildlife Sanctuary were found in ponds with a normal water level cycle (low water in winter, dry in spring and wet in summer and fall). While ponds in semi-impounded areas did have reduced turbidity, the pond bottoms were an organic muck which was unsuitable for *Ruppia* sp. These ponds apparently did not

undergo periodic drawdown which would have allowed the soil to compact and oxidize as recommended for growth of submerged aquatics by Joanen and Glasgow (1965).

Passive structural marsh management in interior brackish marshes on Rockefeller State Wildlife Refuge and Game Preserve involves Wakefield fixed-crest weirs set 6 inches below the marsh surface (Wicker et al., 1983). These weirs were installed in order to stabilize water levels and reduce tidal exchange in these areas. Ponds located in the marsh were never completely drained which initially proved to be beneficial for the establishment of widgeongrass. However, under this type of management some areas have become open water areas with poor widgeongrass production. Wicker et al. (1983) suggested that this development of open water might be due to the levees and weirs which maintain higher than normal water levels. This type of management was rated as marginally successful for production of waterfowl, muskrat and snowgoose food.

Meeder's work (1987) has provided additional data on the effect of a fixed-crest weir. His work in intermediate-to-brackish marsh systems in the Rainey Wildlife Sanctuary compares Tom's Island Bayou, which is unaltered hydrologically, and Big Island Bayou, which has been semi-impounded by a fixed-crest weir set 9 inches below marsh surface since 1967. In addition to the weir, McIlhenny Canal has captured 10-15% of the Big Island watershed.

Comparisons of the vegetation between the upstream and downstream portions of the two bayou systems were made between May 1985 and May 1987. Tom's Bayou had slightly lower salinities and a shorter hydroperiod than the Big Island marshes. The downstream portion of Tom's Bayou had the highest standing crop. The downstream portion of Big Island Bayou, which was not well flushed--having the fewest number of flooding events and the highest mean salinity--had a lower standing crop than Tom's Bayou. The lowest measured standing crop was in the upstream portion of Big Island

Bayou. Meeder (1987) proposed that the difference in hydroperiod may account for the difference in standing crop (200 days inundation/year in the upstream portion of Big Island Bayou versus 66 days/year in the upper portion of Tom's Bayou).

Actively Managed, Gravity Drained Impoundments

Active management, involving the use of variable-crest, reversible flap-gate control structures to accomplish gravity drainage, was rated as relatively successful for waterfowl food production in Rockefeller Wildlife Refuge (Wicker et al., 1983). This type of structure allows the marsh manager to vary the water level inside the management area by changing the crest height. In addition, the reversible flap-gate makes it possible to either completely close off the management area, allow water into the management area, or allow water to drain from the management area. However, maintenance of the control structures and favorable climatic conditions (no flood or drought events) were deemed essential for meeting management goals. The degree of drawdown achieved, which is needed for induction of seed germination of plant species, depends on local rainfall and tidal amplitudes (Wicker et al., 1983).

Growth of widgeongrass is limited by wave action which results in physical injury to the plant and high turbidity levels in marsh ponds (Joanen and Glasgow, 1965). Chabreck and Hoffpauir (1962) reported that turbidity was one of the major physical factors limiting or excluding duck food plants in many areas. Growth of widgeongrass was shown to be best in ponds 24 inches deep having reduced turbidity, little wave action and stable water levels. Summer drawdown was recommended in order to let pond bottom sediments oxidize and solidify followed by reflooding for the fall growing season (Joanen and Glasgow, 1965). Ponds with water levels controlled by weirs were shown to have significantly more aquatic vegetation (30.2%) than ponds without weirs (14.4% [Larrick and Chabreck, 1976]). Ensminger (1963) reported that *Spartina patens* made up

95% of the entire plant community in an unimpounded brackish marsh. Inside the impoundment, it constituted only 40% of the plant community which was dominated by widgeongrass.

In addition to aquatic species, there are certain emergent species that are valuable as waterfowl food including wild millet (*Echinochloa walteri*) and sprangletop (*Leptochloa fascicularis*). Ensminger (1963) reported that the production of a heavy seed crop in these two species was stimulated by flooding the plants when a height of 6-10 inches was reached.

Use of impounded areas in Michigan by waterfowl was found to be excellent for several years after impoundment (Kadlec, 1962). However, the use of periodic drawdown was needed in order to maintain high marsh productivity. When drawdown was accomplished, submerged aquatics persisted only in areas where water remained. The occurrence of emergent species, such as cattail and bulrush, which require a bare mudflat for seed germination, increased. The emergent species, which are valuable as cover for waterfowl, tended to decrease as long as stable water levels were maintained. The level of soil nitrates increased due to aerobic nitrification (Kadlec, 1962).

Actively Managed, Forced Drainage Impoundments

A swamp located in the Lac des Allemands area that has been impounded and actively managed for crawfish production for at least 20 years was investigated by Conner et al. (1981). Use of pumps (forced drainage) have allowed the creation of cyclic flooding which mimics the natural pattern. The area is flooded to a depth of approximately 40 cm in the fall and drained from June through August. As a result, seed germination can occur and there is a dense overstory and understory. Net production is 1,779.9 g/m²/yr as opposed to 1,166.2 g/m²/yr in an unaltered swamp (Conner et al., 1981).

The use of forced drainage in fresh-to-intermediate marshes is used in some areas of the Rockefeller Wildlife Refuge. This type of management, while very expensive, allows for better control of the impoundment water regime (salinity and water levels) and results in excellent stands of duck foods (Wicker et al., 1983). Pumps are employed in order to drain or flood the impounded area according to the management plan.

Management and Storm Events

Another, very important, consideration when discussing structural marsh management, is the occurrence of unusual weather events such as hurricanes. Hurricanes are aperiodic, natural events which exert tremendous influence over Gulf Coast wetlands (Conner and Day, 1989). In 1985, two hurricanes struck southwestern Louisiana in the area of Rainey Wildlife Refuge. Meeder (1987) reported on the effect of these hurricanes on previously described Tom's Bayou and Big Island Bayou. The inundation of Big Island Bayou lasted 21% to 56% longer than that in Tom's Bayou probably because of the fixed crest weir on Big Island Bayou which impeded runoff of storm waters. *Scirpus olneyi* shoots were killed and, because of standing water no new shoots developed.

The tidal surge associated with hurricane Audrey flooded Rockefeller Wildlife Refuge in June 1957. Ensminger and Nichols (1957) reported that the tidal water was trapped within impoundments for several weeks resulting in a decrease in wildlife food for the next season. In addition, detritus from the floor of impoundments was swept away. They further stated that unimpounded areas exhibited very little damage from salt water.

An impoundment designed for use as a pasture and consisting of an almost pure stand of *Paspalum lividum* was also inundated by Hurricane Audrey. In this case, the tidal surge remained within the impoundment for 6 months, killing all vegetation. Reappearance of *P. lividum* did not occur until July, 1959. It did not become dominant until September, 1961 (Shiflet, 1963).

Land Loss

According to Dunbar et al. (1990) the average land loss for the Louisiana coastal zone as of 1983 was 79.54 square kilometers per year. Dahl (1990) reported that, by the 1980's, Louisiana had lost 46% of the total wetlands estimated to have been present in the 1780's. He further stated that although 52.1% of the surface area of the state could be classified as wetlands in the 1780's, this figure had dropped to only 28.3% by the 1980's. Similar trends in total wetland loss occurred in other states over that 200 year period (1780 to 1980). In fact, California lost 91% of its total wetlands during that period (dropping from 5,000,000 acres in 1780's to 454,000 acres in 1980's) and Florida lost 46% (dropping from 20,325,013 acres to 11,038,300 acres). Conversion of wetlands to agricultural land was a significant factor in these losses. If predictions of anticipated sea level rise (ranging from 0.5 to 2.0 cm yr⁻¹) are realized (Titus, 1986), the problems facing Louisiana's coastal wetlands today (which are not related to drainage for agricultural purposes) will be faced by other coastal states in the future.

Management Statistics

At the present rate of construction, 30-40% of the Louisiana coastal wetlands will be impounded by the year 2020 (Cowan et al., 1988). In fact, by 1990, permits for structural marsh management had been issued for 9% of all Louisiana wetlands (Cahoon et al., 1990). According to Cowan et al. (1988), 55% of currently existing impoundments occur in fresh and intermediate marshes which seem to respond positively to active (as opposed to passive) structural marsh management; 5% in saline marshes where the use of impoundments has not been very successful; and 10% in other types of wetlands. The most severe land losses have occurred in interior brackish marshes (Wicker et al., 1983). Thirty percent of currently existing impoundments occur in brackish marshes. Active management has been able to maintain or revegetate areas especially when drawdown occurred every 3 years (Cowan et al., 1988).

Structural Marsh Management

There are many aspects related to the effects of structural marsh management on wetland vegetation and associated environmental factors that lack quantitative information. In addition, the data that is available is primarily concerned with production of waterfowl food. As the emphasis of structural marsh management moves into the area of preservation of natural marsh communities, the need for data on how structural marsh management affects these communities becomes of vital importance.

Saltwater intrusion is often cited as one cause of marsh loss in southern Louisiana. However, there is little data on this subject available in the literature. Since structural marsh management is being proposed as a solution to the problem of salt water intrusion and associated marsh loss, it is important to know how this type of loss occurs. Recently, Mendelssohn and McKee (1987) and McKee and Mendelssohn (1989) have reported that the response of fresh marsh communities may vary depending on such factors as the initial species composition of the marsh, the level, duration, and abruptness of exposure to higher salinity waters, and the depth of flooding. More information is needed in this area before structural marsh management plans are utilized to address the problem of saltwater intrusion.

Many currently accepted monitoring programs do not appear to evaluate the vigor of marsh vegetation growing within marshes in which the hydrology is actively managed by water control structures. As a result, there has been little comprehensive, multiparameter information available on the effects of marsh management on vegetation response and associated soil parameters in comparison to nearby unmanaged marshes. It is important to determine if structural marsh management plans can meet their intended purposes of improving primary productivity through reduced salinities and better water level control. In addition, the need for development of effective and economically realistic

monitoring standards and improvements on existing management techniques are essential. This data must be made available to marsh managers to ensure that best-management practices are followed.

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CHAPTER 2

VEGETATION AND SOIL RESPONSE TO MARSH MANAGEMENT AT THE FINA LATERRE MITIGATION BANK SITE IN SOUTHERN LOUISIANA

INTRODUCTION

In southern Louisiana, marsh management is being used to address a diverse range of objectives, including increasing waterfowl and furbearer use (Craft and Kleinpeter, 1986), excluding salt water and promoting freshwater species, and preventing marsh loss (Simmering et al., 1989). Until this study, little or no comprehensive, multiparameter information was available on the effects of marsh management on vegetation response and associated soil parameters in comparison to nearby unmanaged marshes. Previous reports describing the effects of marsh management on vegetation have relied on monthly salinity measurements, water depth measurements, and vegetation transects. The vegetation transects were used to estimate percentage cover, changes in vegetation, and species diversity, which in turn were used as the basis for reporting whether marsh management improved plant productivity (Craft and Kleinpeter, 1986; Joanen and McNease, 1987; Simmering et al., 1989). The objective of this study was to determine (1) the effect of marsh management on the primary productivity and vigor of the dominant emergent marsh vegetation (*Spartina patens*) and (2) which soil variables may be controlling this vegetative response to marsh management.

Several questions concerning the effects of marsh management on vegetation within the managed marsh at the Fina LaTerre Mitigation Bank were addressed. First, what effect does marsh management have on edaphic factors (i.e., soil oxidation state and interstitial concentration of nutrients, salinity, and sulfide) as compared to nearby

unmanaged marshes? Second, what influence does marsh management have on the productivity of emergent vascular plant production as compared to nearby unmanaged marshes? And third, what conclusions can be made about the use of marsh management in light of the information this study has provided? A similar study, reported in the following chapter, was conducted concurrently at Rockefeller State Wildlife Refuge.

MATERIALS AND METHODS

The goals of the Fina LaTerre management plan, which has been in effect since 1985, are to reverse marsh loss, reverse a trend toward increased salinity, improve primary productivity, increase freshwater and sediment input, stabilize water levels, and improve water circulation. Four fixed-crest weirs (three along Marmande Canal and one along Minors Canal) in the northern part of the management area and a variable-crest flap-gated structure (with two gates) in the southern part of the management area were installed to meet these goals (Cahoon, 1990a; D. R. Clark, 1989). The 4 fixed-crest weirs were installed to allow the exchange of fresh water and sediment in the northern part of the managed area. The variable-crest flap-gate structure is used to implement a spring drawdown during which time the sill level is 24 inches below the marsh surface and the gates are flapping out. The drawdown period in 1989 began in mid-February and lasted until mid-July at which time the flooding phase was initiated by raising the sill to a level only 6 inches below the marsh surface and leaving the gate open (Cahoon, 1990a). My research was conducted in the southern portion of the management area which is intermediate marsh dominated by *Spartina patens*. This study was conducted during a drawdown year.

Sampling Schedule

Sixteen sites were located within the southern portion of Fina LaTerre Mitigation Bank management area north of Falgout Canal and 16 sites were located outside of the managed area immediately south of Falgout Canal. Within each area, sample sites were grouped according to their proximity to the major water exchange point for each area (8 sites near the water exchange point [within 0.75 km] and 8 sites far from the water exchange point [within 1 km for the unmanaged marsh and 2.5 km for the managed marsh], [Figure 1]). Sampling at Fina LaTerre was conducted four times during the course of the investigation, on May 17-18, July 18-19, September 20-21, and November 14-15, 1989. In May, September, and November all sites in the managed and unmanaged marshes were sampled. In July, all sites near the water exchange point in the unmanaged marsh were sampled, but only six sites far from the exchange point were measured because of instrumentation problems. All sites near to and far from the water exchange point were sampled within the managed area at that time.

Analysis of Vegetation Response

Net leaf CO₂ exchange rates of *S. patens*, the dominant species at the Fina LaTerre site, were measured with a portable infrared CO₂ analyser (LCA-2), an ADC air supply unit or ADC air supply unit with mass flow, and a Parkinson Leaf Chamber (Analytical Development Co., Ltd.). Two to 4 measurements (using an open gas exchange system) were made on mature, intact leaves at each sample site. A generator-powered 300W, 125V Sylvania Wide-Flood light was used when photosynthetically active radiation (PAR) was below 1300 $\mu\text{mole m}^{-2} \text{s}^{-1}$. DeJong et al. (1982) and Pezeshki et al. (1987) reported that net photosynthesis of *S. patens* was light saturated at approximately 800 $\mu\text{mole m}^{-2} \text{s}^{-1}$ under field conditions. This allowed measurement of light saturated exchange rates, which were calculated using equations adapted from von Cammerer and Farquahar (1981). Net CO₂ exchange rates were expressed as $\mu\text{moles of}$

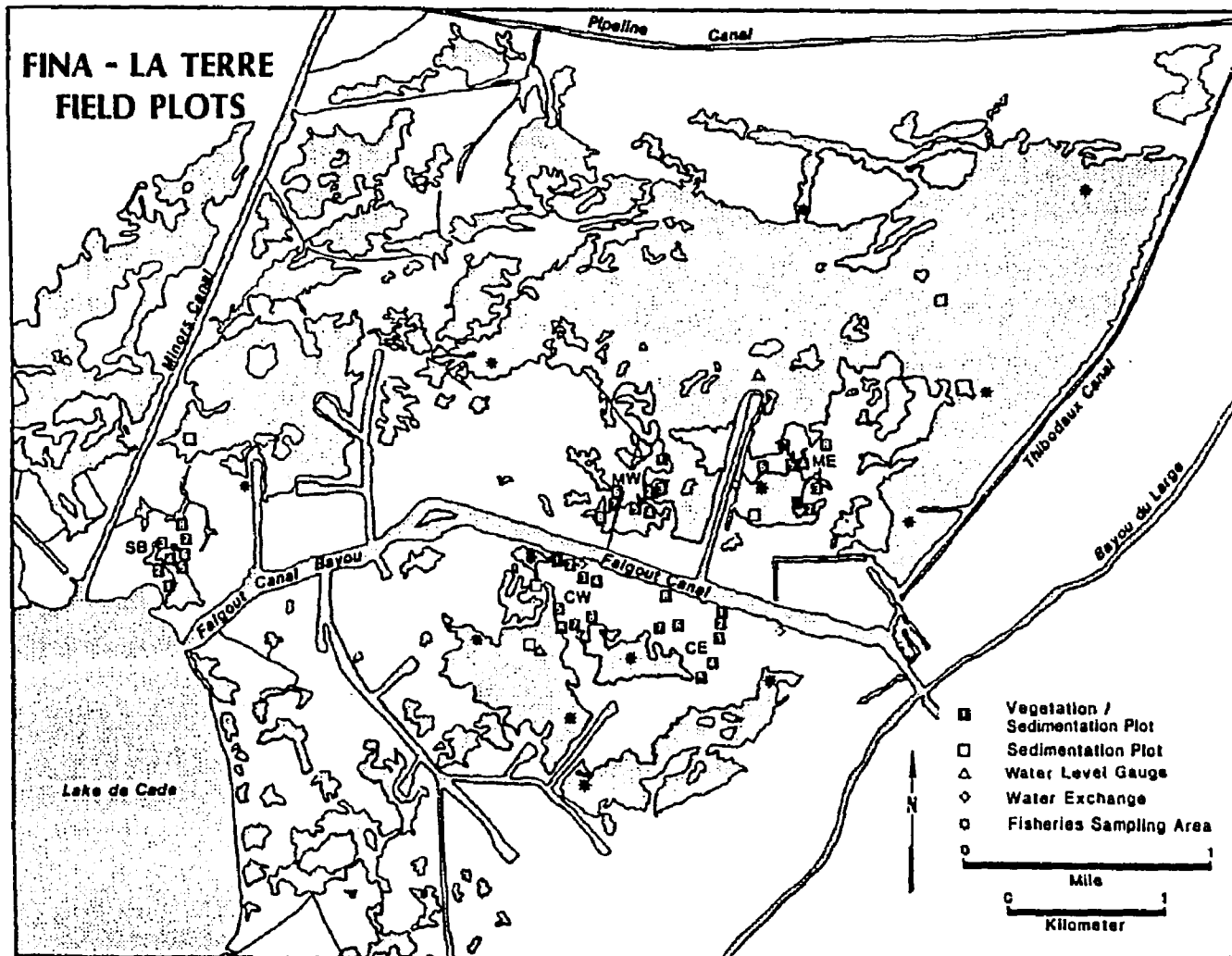


Figure 1. Map of sample sites in the managed and unmanaged marshes at the Fina LaTerre site (adapted from Cahoon and Groat, 1990)..

CO₂ exchanged per m² of leaf surface per second. Total CO₂ exchange rates (per unit area of marsh) were determined by multiplying leaf CO₂ exchange with leaf area per square meter of marsh.

Clip plots (0.1m²) were sampled at each site adjacent to where CO₂ exchange rates were measured on all four sample dates. All aboveground plant material was collected, transported to the lab, and separated by species into live and dead categories. Live stems of *S. patens* were counted to determine stem density. Samples were dried to constant weight at 65° C and weighed to the nearest 0.1 g. Plant data from clip plots were used to estimate net aboveground primary productivity (gm⁻²) of *S. patens* with three different methods, those of (1) Smalley and (2) Milner and Hughes, and (3) the maximum-minimum method as described by Kirby and Gosselink (1976). Net primary productivity is "organic matter stored in plant tissue in excess of respiration during the period of measurement" (Odum 1963:39). The maximum-minimum method estimates productivity on the basis of the difference between the maximum and minimum live aboveground standing crop. Milner and Hughes' method takes into account only changes in live standing crop between frequent harvests during the growing season. Smalley's method estimates net primary productivity on the basis of changes in both live and dead standing crops over time. None of these methods accounts for losses due to decomposition between sampling intervals or losses due to herbivory, and they therefore underestimate productivity (Kirby and Gosselink, 1976). All of these methods were used to determine whether trends in productivity calculated with different estimation techniques. No attempt was made to estimate herbivory in this study. However, very little evidence of herbivory was observed in either the managed or the unmanaged marsh.

Leaf area of *S. patens* was measured indirectly by calculating it on the basis of leaf weight. Leaf area of individual leaves (Fina LaTerre samples collected May 17-18) was measured with a Licor LI-3000 leaf area meter; the leaves were weighed, and a linear

regression ($y=0.2264+(31.27X)$, $r^2=0.93$) was generated relating leaf area (dependent variable) to leaf weight (independent variable). All subsequent measurements of leaf area were based on leaf weight using this equation. Leaf weights per 0.1 m^2 area in May samples were determined using the leaves from all stems in the plot; leaf weight for other months was computed by multiplying the leaf weight of 20 representative stems by the stem density in a plot. All weights and leaf surface areas are presented on the basis of a square meter of marsh.

Soil Parameters

Duplicate, instantaneous Eh (redox potential) readings were made in the upper one to two cm of soil (hereafter referred to as surface Eh) and at a depth of 15 cm at each site. Measurements were made using a calomel reference electrode, brightened platinum electrodes, and a portable digital pH-mV meter. In order to base the readings on a standard hydrogen electrode, 244 mV was added to each reading (Faulkner et al., 1989). Soils were classified as aerated ($>300 \text{ mV}$), moderately reduced (100 to 300 mV), reduced (-100 to 100 mV), and highly reduced ($<-100 \text{ mV}$, [Patrick, 1980]). Eh readings were not corrected for pH.

To determine water depth above or below the soil surface at each site, shallow wells were installed between *S. patens* hummocks. The wells consisted of 150 cm lengths of PVC pipes, which were buried to a depth of approximately 61 cm. Quarter-inch holes were drilled into the buried end of each pipe to allow movement of water in and out. Each pipe was covered on top with a removable cap. Water depth measurements were made at each site on all four sampling dates by removing the cap and dropping a float connected to a line into the pipe. Distance from the pipe top to the midpoint of the float was then measured. The height of each pipe was measured twice during the study. Water depth above or below the soil surface was calculated by subtracting the distance from the top of the pipe from the pipe height.

Soil cores were collected to a depth of approximately 12 cm using an aluminum corer 6 cm in diameter. Soil was extruded into 500 ml centrifuge bottles and sealed. To ensure an anaerobic environment, the samples were purged with nitrogen via a rubber septum in the bottle cap and placed on ice for transport to the laboratory. The samples were centrifuged at 10,000 g at 4° C for 10 minutes. Immediately upon opening the bottles, an aliquot of the supernatant was added to an antioxidant solution (NaOH, ascorbic acid, sodium salycilate) for sulfide determination using a Lazar ISM-146 Micro Ion sensing electrode and a portable Cole-Parmer Model 5985-80 Digi-Sense mV meter (Lazar, 1986).

Salinity of the interstitial water was measured using a Fisher Conductivity Meter Model 152 and pH was measured using an Altex Model 3560 Digital pH meter with a Corning General Purpose Combination Electrode. The water was filtered using a 0.45 micron millipore filter, and an aliquot was then removed and frozen for NH₄-N analysis (U. S. Environmental Protection Agency, 1979). The remainder of the filtered water was acidified with concentrated nitric acid and analysed for essential nutrients (P, K, Ca, Mg, Na, Fe, Mn, Cu, and Zn) using a Fisher inductively coupled argon plasma emission spectrometer (ICAP, Atom Comp Series 800).

Statistical Analysis

Statistical analyses were conducted with SAS software (SAS Institute, 1985). The effects of treatments and month on soil and plant variables were analyzed with a split-plot analysis of variance, with treatment effects (management and proximity to a water exchange point) in the main plot, and month effects in the subplot. Pairwise comparisons between months were computed with least-squares means when the overall month effect was significant. All hypotheses were tested at the 0.05 probability level unless otherwise indicated.

RESULTS

Soil Response

Water Depth and Soil Reduction

The effect of marsh management on water levels measured in the shallow wells varied with proximity to the water exchange point and with month (treatment x water x month interaction significant, $p=0.0289$, Figure 2a). Water level was higher in the unmanaged marsh (near and far) than in the managed marsh (near and far) on all sampling dates except July, when no significant differences occurred. Water depth was not different between the near and far areas within the managed marsh on any sampling date. While no significant differences between the near and far unmanaged areas occurred in July or September, the unmanaged near area had significantly greater water depth than the unmanaged far area in May and November (Figure 2a). Water depth increased significantly in the managed marsh (near and far) from May to July, but decreased in the unmanaged near area. Water depth increased significantly from May to September in the unmanaged far area and in September and November in the unmanaged near area (Figure 2a).

Although water depth was, on average, greater in the unmanaged marsh, surface soil conditions in the managed marsh (-101 ± 13 mV) were more reduced than in the unmanaged marsh (-43 ± 15 mV, [treatment significant, $p=0.0033$]). The effect of marsh management on surface Eh varied with proximity to the water exchange point (treatment x water interaction significant, $p=0.0573$, Figure 2b). In the unmanaged marsh, surface Eh, when averaged over time, was significantly higher nearer to the water exchange point than farther from it (Figure 2b). However, in the managed marsh, surface Eh did not vary as a function of nearness to the water exchange point. In addition, the surface Eh in the unmanaged near marsh was significantly higher than that in the managed near marsh. In

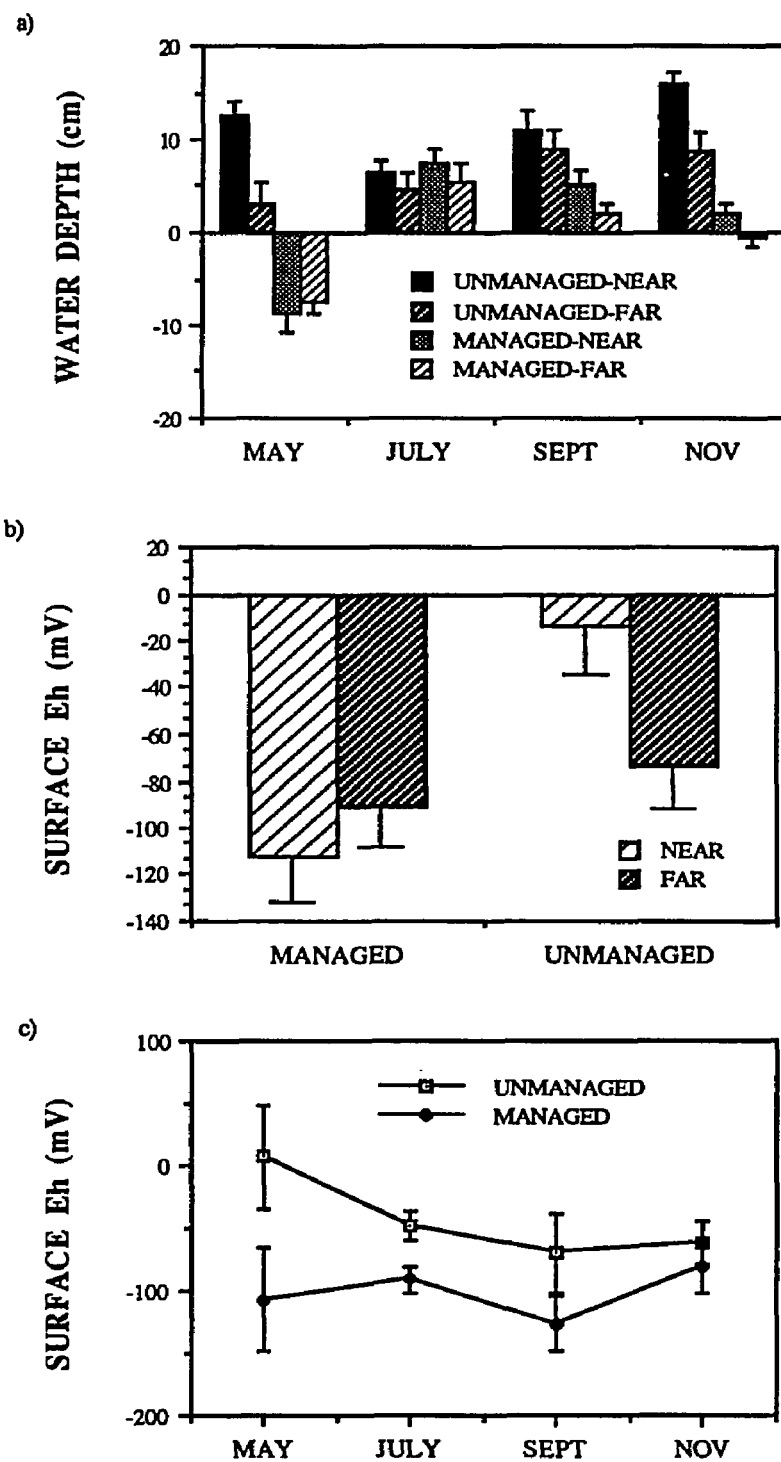


Figure 2. (a) Water depth, (b) surface Eh, and (c) surface Eh over time in an unmanaged and a managed marsh near and far from a water exchange point at the Fina LaTerre site in 1989.

the managed marsh, the average surface Eh in the near area was not significantly different from that in the far area (Figure 2b). The greater soil reduction in the managed marsh, despite lower water depth, may have been in response to: (1) the lower mineral content of the surface soils within the managed marsh (Cahoon, 1990b) and hence a lower ability to poise the Eh at higher redox potentials (Gambrell and Patrick, 1978), and (2) the presence of more labile organic matter, resulting from organic matter decomposition, in the managed marsh, which contained higher total organic matter (managed marsh: $52 \pm 4\%$ organic matter, unmanaged marsh: $75 \pm 2\%$ organic matter [Cahoon, 1990b) than the unmanaged marsh and (3) marsh vegetation can affect soil Eh (Chen and Barko, 1988; de la Cruz et al., 1989). The greater productivity of *Spartina patens* in the unmanaged marsh may have resulted in less-reduced soil conditions due to transport of oxygen via shoots to the root zone. A readily useable carbon source can stimulate bacterial activity in saturated soils and cause development of more reduced conditions (Gambrell and Patrick, 1978). Although surface Eh did not vary significantly with time within either marsh ($p=0.3614$, Figure 2c), the surface Eh of the unmanaged marsh was consistently higher than that of the managed marsh throughout the investigation (Figure 2c). It is important to remember that Eh values are useful in evaluating conditions in the soil, but these numbers are not absolute since variations in Eh can occur within small distances.

Eh of soils 15 cm deep varied with month (month significant, $p=0.0001$). However, neither management nor distance from the water exchange point had any significant effect on soil Eh at 15 cm depth (treatment, $p=0.3330$; water, $p=0.3936$). Eh at 15 cm depth was highly reduced in May (-192 ± 10 mV), reduced in July (-95 ± 8 mV), highly reduced in September (-145 ± 10 mV), and reduced in November (-103 ± 10 mV).

Interstitial pH

Averaged over time and proximity to the water exchange point, interstitial water from the unmanaged marsh had a higher pH (6.92 ± 0.02) than that from the managed marsh (6.79 ± 0.03 ; treatment significant, $p=0.0004$). The effect of marsh management on pH varied with proximity to the water exchange point (treatment x water interaction significant, $p=0.0316$, Figure 3a) and with month (treatment x month interaction significant, $p=0.0003$], Figure 3b). In the unmanaged marsh, pH did not significantly differ between the near and far areas. In the managed marsh, however, locations near the water exchange point had a significantly higher pH than the far locations (Figure 3a). The far area in the unmanaged marsh had, on average, higher pH than did the far area in the managed marsh. Near the water exchange point, however, the managed and unmanaged locations were not significantly different. The pH of both the managed and unmanaged marshes decreased from May to July, and increased in September (Figure 3b). The pH of the managed and unmanaged marshes was within the range expected for a flooded soil (Gambrell and Patrick, 1978).

Sulfide Concentration

The effect of marsh management on interstitial sulfide concentration varied with proximity to the water exchange point (treatment x water interaction significant, $p=0.0296$, Figure 4a). Higher sulfide concentrations in the unmanaged area farthest from the point of water input where Eh was lowest, indicated that, at the time of sampling, soil water drainage was impaired in the unmanaged far area. In addition, precipitation of sulfide with Fe in the managed marsh may have lowered sulfide concentrations and accounted for the lower Fe concentrations observed in the managed marsh. Sulfide concentrations were greater in the managed marsh than in the unmanaged marsh only at the near area. At the far area, the unmanaged marsh had higher sulfide concentrations than

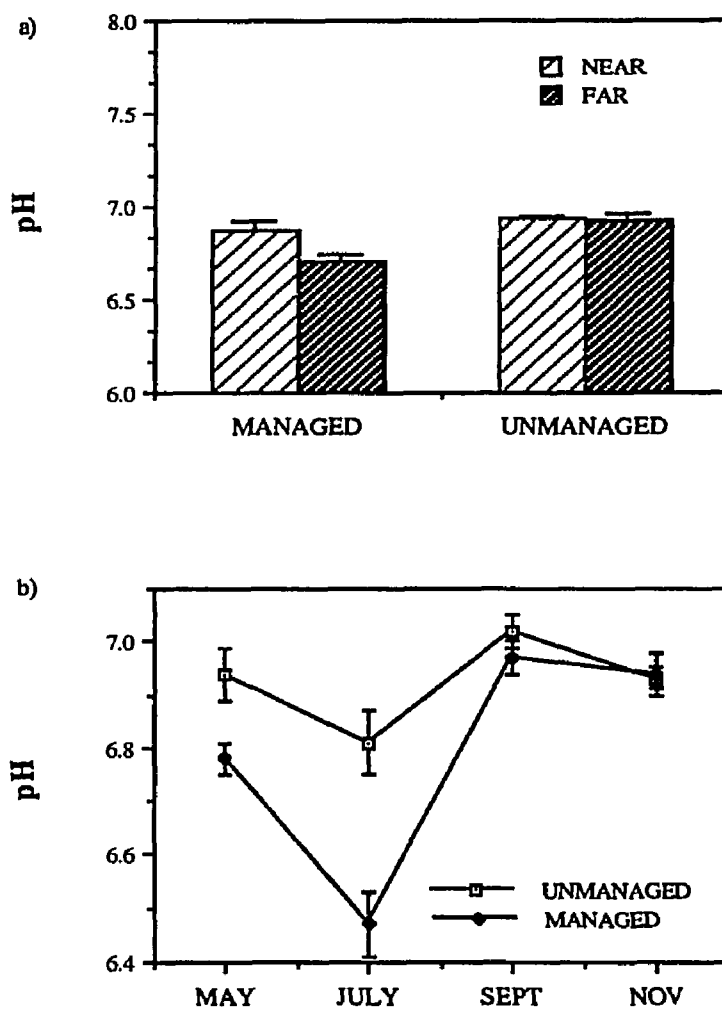


Figure 3. Interstitial water pH in (a) averaged over time in an unmanaged and a managed marsh near and far from a water exchange point, and (b) in an unmanaged and a managed marsh measured over time at the Fina LaTerre site in 1989.

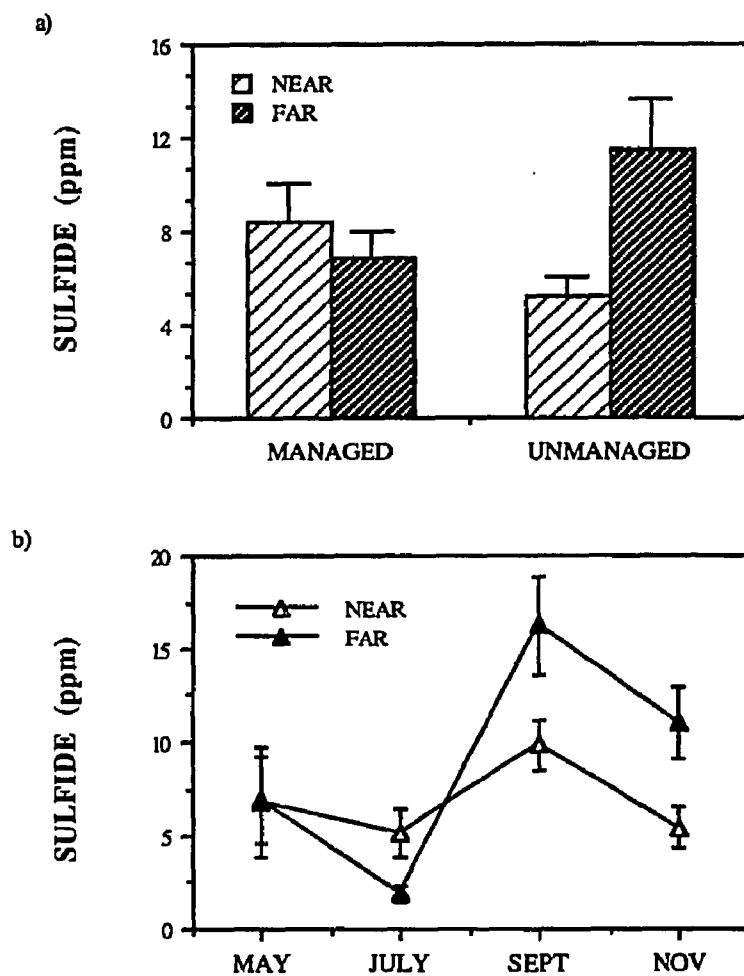


Figure 4. Interstitial water sulfide in (a) an unmanaged and a managed marsh near and far from a water exchange point, and (b) averaged over unmanaged and managed marshes near and far from a water exchange point at the Fina LaTerre site in 1989.

the managed marsh (Figure 4a). Sulfide concentrations (Figure 4a) and Eh values (Figure 2b) in the managed marsh did not vary with distance from the water exchange point.

The effect of distance from the water exchange point on sulfide concentration varied with month (water x month interaction significant, $p=0.0225$, Figure 4b). When averaged over managed and unmanaged marshes, sulfide concentrations did not significantly differ on any of the four sampling dates at the marsh area located nearest the water exchange point. At the area located farthest from the water exchange point, sulfide concentration increased sevenfold from July to September (Figure 4b). This was at a time when ambient surface water salinities increased and could have resulted in sulfate input to the marsh--especially in the unmanaged marsh. Sulfide concentrations, when averaged over managed and unmanaged marshes, were not significantly different between near and far marsh areas in May or July. In September and November, however, the area farthest from water input had significantly higher sulfide concentrations than the area near water input, indicating impaired soil-water drainage and poor circulation (Figure 4b).

Interstitial Ammonium and Phosphorus

Ammonium concentration, averaged over all treatments, was 34% greater in the managed marsh ($2.59 \pm 0.14 \text{ mg l}^{-1}$), which had more reduced soil conditions (Figure 2c) than in the unmanaged marsh ($1.70 \pm 0.19 \text{ mg l}^{-1}$; treatment significant, $p=0.0005$). Locations farther from the water exchange point had more reduced soil conditions (Figure 2b), higher sulfide concentrations (Figure 4a), and 27% more ammonium ($2.50 \pm 0.2 \text{ mg l}^{-1}$) than locations near the water exchange point ($1.83 \pm 0.14 \text{ mg l}^{-1}$; sediment significant, $p=0.0090$). Ammonium concentrations also varied with month (month significant, $p=0.0001$, Figure 5). Averaged over all treatments, May ($2.13 \pm 0.36 \text{ mg l}^{-1}$) and July ($2.14 \pm 0.16 \text{ mg l}^{-1}$) concentrations were the same; concentrations

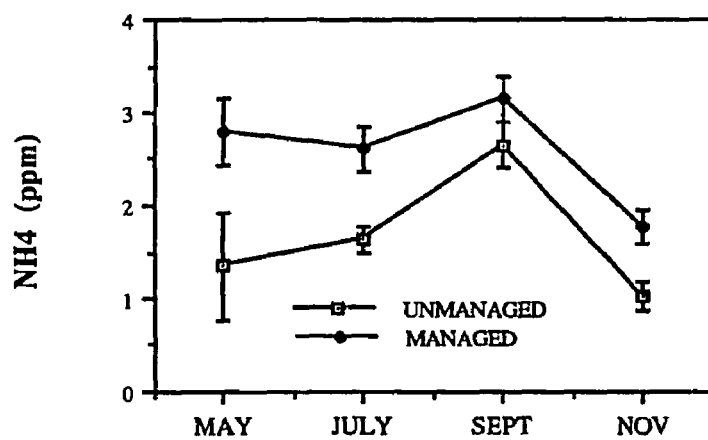


Figure 5. Interstitial water $\text{NH}_4\text{-N}$ in an unmanged and a managed marsh measured over time at the Fina LaTerre site in 1989.

peaked in September ($2.90 \pm 0.18 \text{ mg l}^{-1}$) and subsequently decreased in November ($1.39 \pm 0.14 \text{ mg l}^{-1}$). No significant differences in interstitial phosphorus were apparent during the investigation (treatment, $p=0.7109$; sediment, $p=0.8116$).

Nitrogen is considered to be the primary growth-limiting nutrient of saline marsh vegetation (Linthurst, 1979; Mendelsohn, 1979; Valiela and Teal, 1974). In submerged soils, nitrate-nitrogen is rapidly denitrified (Patrick and Wyatt, 1964). Because aerobic conditions are required for microbial oxidation of ammonium to nitrate, the breakdown of organic matter in anaerobic soils results in accumulation of ammonium-N (Redman and Patrick, 1965). Even though organic matter breakdown is slow and incomplete in anaerobic soils, anaerobic microorganisms have a low requirement for N (Patrick and Mikkelsen, 1971). As a result, soils having a high percentage of organic matter will produce more ammonium (Redman and Patrick, 1965). The higher interstitial ammonium concentrations in the managed marsh may have been due to higher organic matter and lower Eh in the managed marsh in conjunction with lower plant uptake (Koch and Mendelsohn, 1989; Mendelsohn and McKee, 1988).

Phosphorus availability is indirectly tied to Eh because of transformations of other elements with which it may precipitate. Under aerobic conditions, P exists as insoluble Fe-, Ca-, or Al-phosphate (Mahapatra and Patrick, 1969). When anaerobic conditions develop ($\text{Eh} \leq 200 \text{ mV}$), Fe, Ca, and Al become more soluble, and P compounds hydrolize (Redman and Patrick, 1965). This usually results in an increase in extractable P upon submergence of soils due to its increased solubility (Patrick, 1964). The lack of any significant difference in interstitial P between managed and unmanaged marshes may reflect the fact that P does not limit plant growth in these environments.

Salinity

Averaged over time, interstitial salinity was higher in the managed marsh (3.3 ± 0.1) than in the unmanaged marsh (2.3 ± 0.1 , treatment significant, $p=0.0001$). In

addition, the area farthest from the water exchange point had higher salinity (3.2 ± 0.1 ppt) than the area nearest the water exchange point (2.5 ± 0.1 ppt, sediment significant, $p=0.0045$). The effect of marsh management on salinity varied with proximity to the water exchange point and month (treatment x water x month interaction significant, $p=0.0013$, Figure 6a). In the managed marsh, salinity in the far area was relatively high and constant throughout the study (Figure 6a). However, although the managed near area had salinities equal to those of the managed far area in May, salinity declined significantly in July in the managed near area, which resulted in lower salinities in July, September, and November than occurred in the managed far area. Salinities at the near and far locations in the unmanaged marsh were different only in May and July (Figure 5a). In the unmanaged marsh, salinities measured in May and July were lower than in the managed marsh, but over time salinities in the unmanaged marsh increased to levels found in the managed marsh (Figure 6a). The interstitial salinities in the unmanaged marsh tracked ambient surface salinities (Rogers and Rogers, 1990) relatively closely, whereas salinities in the managed marsh did not.

Interstitial concentrations of the macronutrients Na, K, Mg, and Ca were measured in interstitial water samples from each site (Figure 6b-6e). Averaged over time, the managed marsh had greater concentrations of these four elements than the unmanaged marsh (treatment significant [Na: $p=0.0001$; K: $p=0.0383$; Mg: $p=0.0001$; Ca: $p=0.0001$], Na: managed: 1009 ± 36 ppm, unmanaged: 709 ± 54 ppm; K: managed: 30.46 ± 1.0 ppm, unmanaged: 25.89 ± 1.6 ppm; Mg: managed: 126.4 ± 4.4 ppm, unmanaged: 86.83 ± 6.7 ppm; Ca: managed: 59.15 ± 2.2 ppm, unmanaged: 39.25 ± 2.7 ppm). The effect of marsh management on Na, K, Mg, and Ca varied with proximity to the water exchange source and month (treatment x water x month interaction significant [Na: $p=0.0008$; K: $p=0.0107$; Mg: $p=0.0079$; and Ca: $p=0.0014$], Figure 6b-6e). Concentrations of Na, K, Mg, and Ca, which are the major cation components in seawater (Weyl, 1970), exhibited the same pattern as was described for salinity.

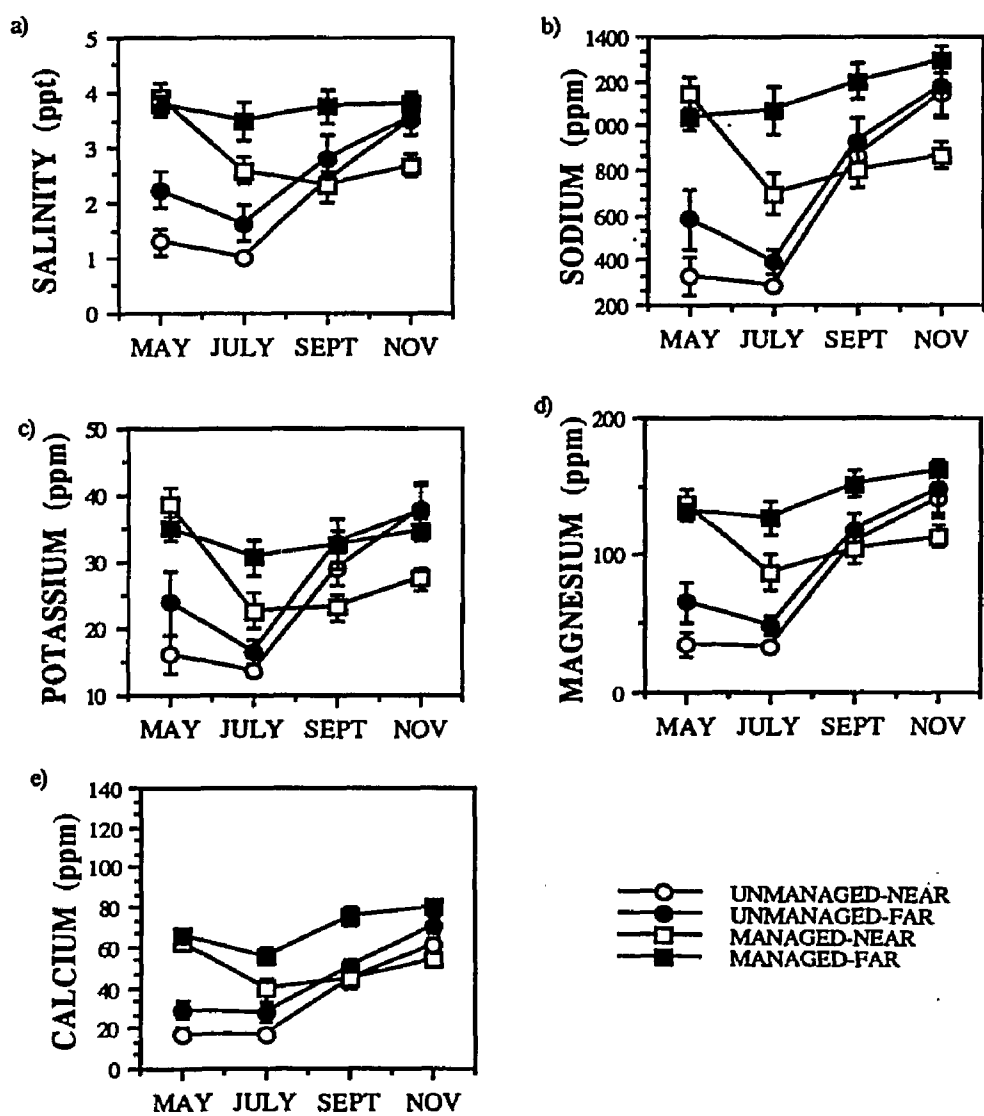


Figure 6. Interstitial water (a) salinity, (b) sodium, (c) potassium, (d) magnesium, and (e) calcium, measured over time in an unmanaged and a managed marsh near and far from a water exchange point at the Fina LaTerre site in 1989.

Micronutrients--Fe, Zn, Mn, and Cu

The micronutrients Fe, Zn, Mn, and Cu were also measured in interstitial water samples. Averaged over time and proximity to the water exchange point, the unmanaged marsh had approximately three times the level of Fe (0.1845 ± 0.0458 ppm) as the managed marsh (0.0685 ± 0.0030 ppm, treatment significant, $p=0.0385$, Figure 7a). There were no differences in Fe concentrations associated with proximity to the water exchange point ($p=0.3873$) or with sampling time ($p=0.1838$). The lower concentration of soluble Fe in the managed marsh compared to the unmanaged marsh could be due to the formation and precipitation of FeS caused by the higher Eh (Figure 2c), which would result in higher concentrations of soluble interstitial Fe and sulfide (Gambrell and Patrick, 1978).

The effect of marsh management on Mn concentrations varied with month (treatment x month interaction significant, $p=0.0216$, Figure 7b). The managed marsh had greater concentrations of Mn than the unmanaged marsh in May and July. Concentrations of Mn were not significantly different in September and November. Although Mn concentrations in the managed marsh did not change over time, concentrations of interstitial Mn in the unmanaged marsh were significantly higher in September and November than in July. Distance from the water exchange point did not affect concentrations of Mn ($p=0.0926$).

Zinc concentrations, which were the same in the unmanaged and managed marshes (treatment: $p=0.4688$), varied with month (month significant [$p=0.0001$]). Zn levels were very low in May and July, but increased fivefold from July to November (Figure 7c). Cu was not different between treatments ($p=0.4959$) or with time ($p=0.2179$).

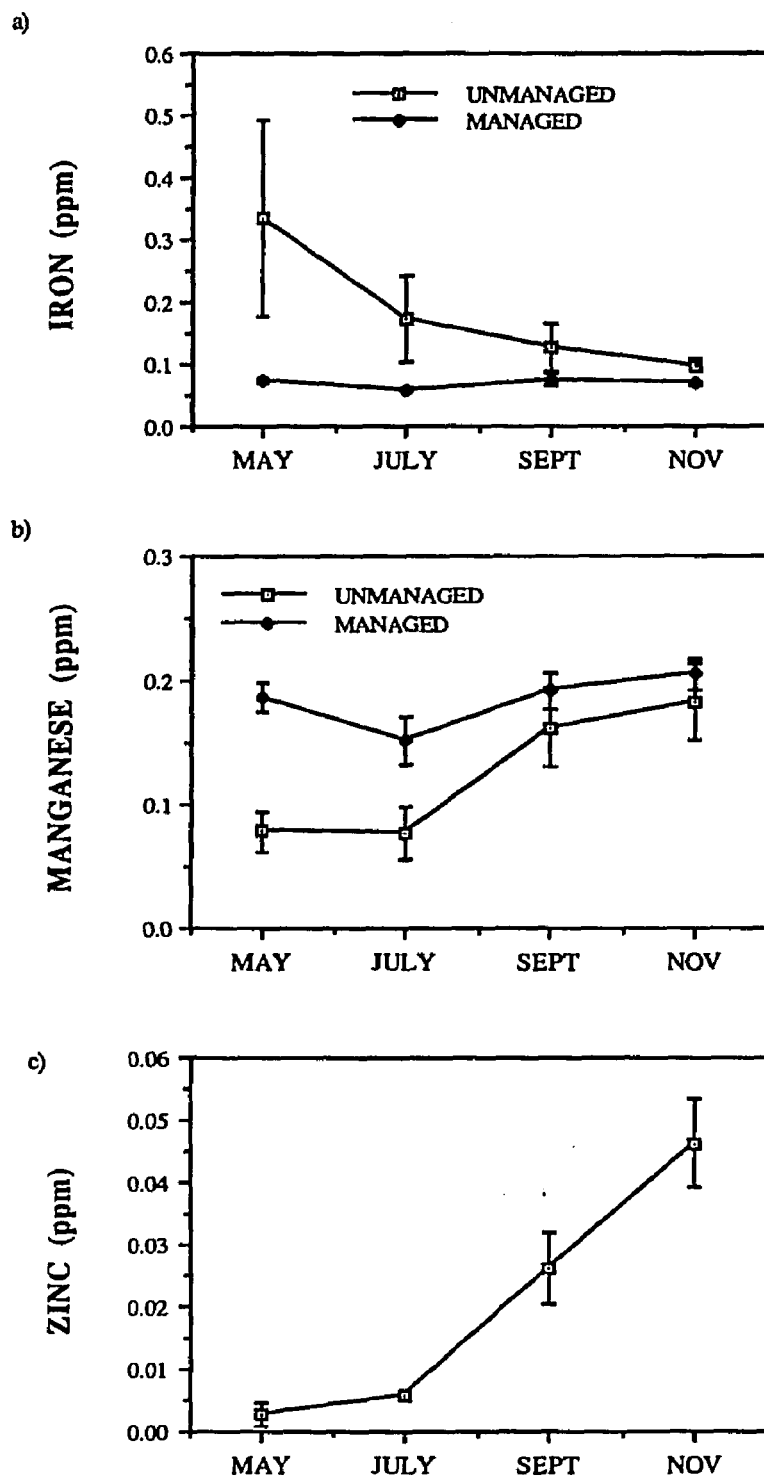


Figure 7. Interstitial water (a) iron, and (b) manganese, measured over time in an unmanaged and a managed marsh, and (c) zinc, averaged over managed and unmanaged marshes at the Fina LaTerre site in 1989.

Vegetation Response

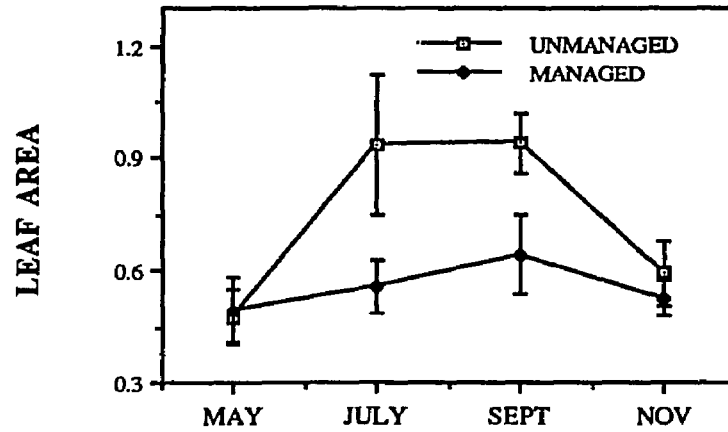
Stem Density and Leaf Area

Stem density and leaf area of *S. patens* were significantly greater (treatment significant; stem density: $p=0.0564$; leaf area: $p=0.0512$) in the unmanaged marsh (99 ± 6 stems m^{-2} marsh; leaf area: 0.75 ± 0.06 m^2 m^{-2} marsh) than in the managed marsh (81 ± 4 stems m^{-2} marsh, leaf area: 0.56 ± 0.04 m^2 m^{-2} marsh). Stem density did not significantly change over time. Leaf area, averaged over treatments, exhibited a seasonal pattern; it increased from May (0.49 ± 0.06 m^2 m^{-2} marsh) to July (0.74 ± 0.10 m^2 m^{-2} marsh) and September (0.79 ± 0.07 m^2 m^{-2} marsh), and declined in November (0.56 ± 0.05 m^2 m^{-2} marsh) (month significant, $p=0.0034$). The seasonal increase in leaf area, however, was more evident in the unmanaged marsh (Figure 8a). The treatment effect upon stem density and leaf area did not differ with month (treatment x month interaction; stem density, $p=0.6178$; leaf area, $p=0.0950$). Also, proximity to the water exchange point did not affect stem density ($p=0.6717$) or leaf area ($p=0.3743$).

Live Aboveground Biomass

Live aboveground biomass of *Spartina patens*, averaged over time and proximity to the water exchange point, was significantly greater in the unmanaged marsh (1356.5 ± 97.2 g m^{-2}) than in the managed marsh (897.2 ± 63.7 g m^{-2} [treatment significant, $p=0.0068$]). Live biomass, averaged over treatments, varied with month (month significant, $p=0.0192$]), though the effect was more evident in the unmanaged marsh (Figure 8b). The effect of marsh management on live biomass, however, did not differ with month (treatment x month interaction, $p=0.2579$) or proximity to the water exchange point ($p=0.7225$). *S. patens* made up 99-100% of the live vegetation in the unmanaged clip plots and 92-100% of the live vegetation in the managed clip pots at the Fina LaTerre site.

a)



b)

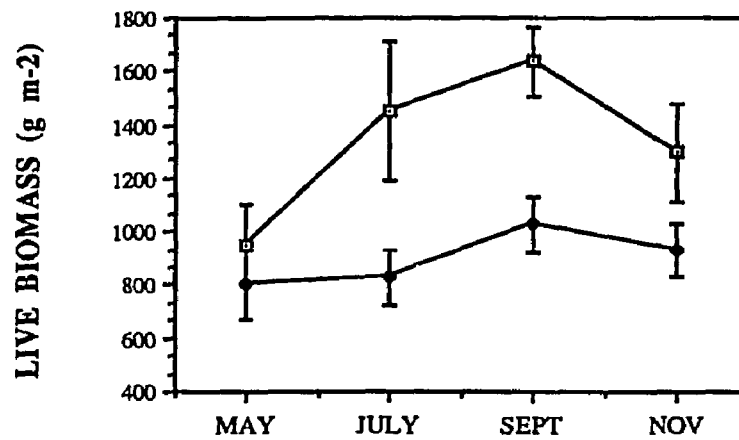


Figure 8. *Spartina patens* (a) leaf area and (b) live biomass measured over time in an unmanaged and a managed marsh at the Fina LaTerre site in 1989.

Dead and Total Aboveground Biomass

Both dead and total aboveground biomass varied with sampling date (month significant [dead, $p=0.0003$; total, $p=0.0046$]), but were not affected by treatment (dead, $p=0.2097$; total, $p=0.8586$). Dead biomass increased from May ($1757.7 \pm 223.9 \text{ g m}^{-2}$) to July ($2188.5 \pm 218.1 \text{ g m}^{-2}$). It decreased in September ($1649.2 \pm 151.5 \text{ g m}^{-2}$) after which no significant change in dead biomass occurred (November $1234.7 \pm 132.1 \text{ g m}^{-2}$). Total biomass exhibited the same pattern as leaf area--a low in May ($2648.9 \pm 279.8 \text{ g m}^{-2}$), and an increase in July ($3316.3 \pm 309.6 \text{ g m}^{-2}$). No significant differences in total biomass were measured in July ($3316.3 \pm 309.6 \text{ g m}^{-2}$) and September ($2981.5 \pm 211.3 \text{ g m}^{-2}$). Total biomass decreased to initial levels in November ($2345.4 \pm 195.9 \text{ g m}^{-2}$). Proximity to the water source did not affect either dead ($p=0.3540$) or total ($p=0.6459$) biomass.

Net Primary Productivity

Regardless of the method used, estimates of net primary productivity of *S. patens* in the unmanaged marsh were always approximately twice as great as those for the managed marsh (treatment significant [maximum-minimum, $p=0.0289$; Smalley, $p=0.0134$; Milner and Hughes, $p=0.0495$], Table 1). The Smalley method, which accounts for plant mortality but not decomposition between sampling intervals (Kirby and Gosselink, 1976; Shew et al., 1981), generated the highest estimate of net primary productivity, whereas the maximum-minimum and the Milner and Hughes methods, which do not account for either of these losses (Kirby and Gosselink, 1976), gave much lower estimates. The greater net primary productivity of the unmanaged marsh compared to the managed marsh is consistent with differences between the two marshes in stem density, leaf area (Figure 8a), and live biomass (Figure 8b). Distance from the water

Table 1. Estimates of net primary productivity (g m^{-2}) for *Spartina patens* in managed and unmanaged marshes at the Fina LaTerre site during 1989 (n=number of sites within each marsh for which biomass data was obtained on all 4 sample dates).

SITE	n	MAX-MIN	SMALLEY	MILNER-HUGHES
Managed	15	945.0 \pm 143.4 ^a	1486.6 \pm 285.9 ^a	860.0 \pm 142.3 ^a
Unmanaged	10	1541.4 \pm 213.7 ^b	3036.2 \pm 662.0 ^b	1424.1 \pm 270.5 ^b

exchange point did not significantly affect net primary productivity rates within the managed or unmanaged marsh (maximum-minimum, $p=0.6871$; Smalley, $p=0.0881$; Milner and Hughes, $p=0.2954$).

Leaf CO₂ Exchange Rate

Averaged over time and proximity to the water exchange point, the leaf CO₂ exchange rate of *S. patens* was higher in the unmanaged marsh ($19.0 \pm 1.15 \mu\text{mole m}^{-2} \text{s}^{-1}$) than in the managed marsh ($15.0 \pm 1.06 \mu\text{mole m}^{-2} \text{s}^{-1}$) (treatment significant, $p=0.0102$). The effect of marsh management on leaf CO₂ exchange rate varied with month (treatment x month interaction significant, $p=0.0349$, Figure 9a). Since the time of day when samples were collected significantly affected the CO₂ exchange rate measured (hour² significant, $p=0.0002$, accounted for 7.4% of the variation in CO₂ exchange rate, Table 2), the CO₂ exchange rates were adjusted for the quadratic covariable, hour. Leaf CO₂ exchange rate was greater in the unmanaged marsh than in the managed marsh in May (Figure 9a). Although leaf CO₂ exchange rate was not significantly different between the two marshes on any other sample date, values tended to be higher in the unmanaged marsh than in the managed marsh (Figure 9a). The rate of CO₂ exchange did not significantly differ with month in the managed marsh. In the unmanaged marsh, the leaf CO₂ exchange rate was significantly lower in November than in May (Figure 9a). Although the effect of marsh management on the leaf CO₂ exchange rate varied with month and proximity to the water exchange point (treatment x water x month interaction significant, $p=0.0462$), the trends were not consistent with respect to the water exchange point.

Total CO₂ Exchange Rate

Total CO₂ exchange rate was also adjusted for the covariable, hour, because of the significant effect of time on total CO₂ exchange rate (hour significant, $p=0.0297$, hour

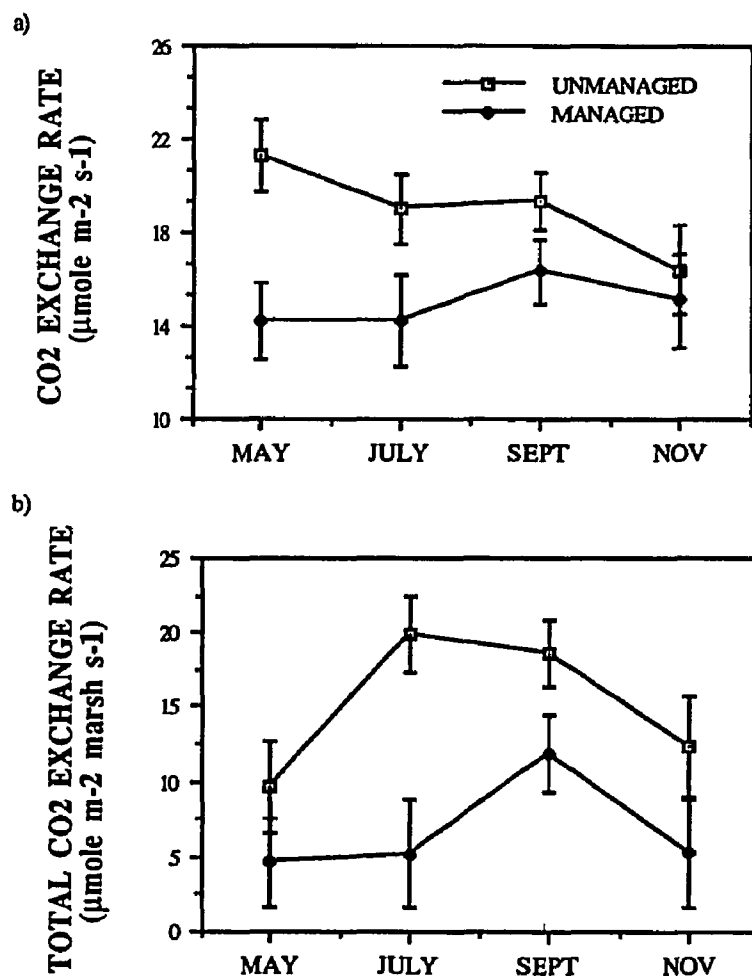


Figure 34. Leaf CO₂ exchange rates of *Sacciolepis striata* and *Hydrocotyle umbellata* (a) in a managed and unmanaged marsh averaged over all sample dates, and (b) in a managed and unmanaged marsh over time at Avoca Island, Louisiana, 1990.

Table 2. *Spartina patens* leaf CO₂ exchange rates ($\mu\text{mole m}^{-2}$ leaf area s⁻¹; adjusted and non-adjusted for time of day) in a managed and unmanaged marsh, near and far from a sediment exchange point, at Fina LaTerre, Louisiana, 1989.

	n	Adjusted	Non-adjusted
<hr/>			
Managed-near			
May	8	16.4±1.6	17.6±1.8
July	8	13.2±1.8	14.1±1.0
September	8	14.8±1.5	15.8±2.2
November	8	15.9±2.3	16.1±1.1
Managed-far			
May	8	12.1±2.4	12.2±1.8
July	8	15.3±2.8	14.6±1.7
September	8	17.9±2.2	17.3±1.3
November	8	14.3±2.2	14.6±1.5
Unmanaged-near			
May	8	19.9±1.5	21.3±2.5
July	8	16.2±1.5	17.7±0.6
September	8	19.8±2.3	19.9±1.5
November	8	16.9±2.0	17.1±0.7
Unmanaged-far			
May	8	22.8±2.4	22.6±2.2
July	6	21.8±2.4	21.0±2.0
September	8	18.8±3.2	15.0±2.0
November	8	16.0±2.3	15.0±1.2
<hr/>			

Table 3. *Spartina patens* total CO₂ exchange rates ($\mu\text{mole m}^{-2}$ marsh s⁻¹; adjusted and non-adjusted for time of day) in a managed and unmanaged marsh, near and far from a sediment exchange point, at Fina LaTerre, Louisiana, 1989.

	n	Adjusted	Non-adjusted
Managed-near			
May	8	3.6 \pm 2.8	5.9 \pm 1.1
July	8	5.3 \pm 3.2	7.9 \pm 1.8
September	8	11.9 \pm 2.6	12.7 \pm 3.7
November	8	6.2 \pm 4.1	9.1 \pm 1.4
Managed-far			
May	8	5.7 \pm 4.4	8.9 \pm 2.2
July	8	5.0 \pm 5.0	7.8 \pm 1.5
September	8	11.8 \pm 4.0	8.7 \pm 1.6
November	8	4.4 \pm 4.0	7.3 \pm 1.2
Unmanaged-near			
May	8	4.0 \pm 3.9	7.9 \pm 1.7
July	8	5.3 \pm 3.2	14.9 \pm 4.8
September	8	12.4 \pm 4.1	15.2 \pm 2.4
November	8	13.6 \pm 3.6	11.8 \pm 2.7
Unmanaged-far			
May	8	15.3 \pm 4.3	11.6 \pm 3.2
July	6	26.1 \pm 4.3	23.6 \pm 4.2
September	8	24.8 \pm 5.7	16.9 \pm 3.0
November	8	11.0 \pm 4.1	7.4 \pm 1.3

accounted for 3.2% of the overall variation in CO₂ exchange rates, Table 3). Averaged over the four sample dates and proximity to the water exchange point, total CO₂ exchange rate of *S. patens* was 125% higher in the unmanaged marsh ($15.1 \pm 2.1 \mu\text{mole m}^{-2} \text{ marsh s}^{-1}$) than in the managed marsh ($6.7 \pm 0.1.9 \mu\text{mole m}^{-2} \text{ marsh s}^{-1}$) (treatment significant, $p=0.0020$). The average total CO₂ exchange rate also varied with month (month significant, $p=0.0012$). In the unmanaged marsh, the total CO₂ exchange rate, which was low in May, increased significantly in July (Figure 9b). No significant difference in total CO₂ exchange rate occurred between July and September, but a significant decrease occurred from July to November in the unmanaged marsh (Figure 9b). No differences in the rate of total CO₂ exchange occurred in the managed marsh on any of the four sampling periods. Proximity to the water exchange point did not affect the total CO₂ exchange rate ($p=0.9578$).

The growth response of *S. patens* within the managed marsh at Fina LaTerre was consistently poorer than that of *S. patens* in the unmanaged marsh. Stem density, leaf area, live biomass and net aboveground primary productivity, as well as the leaf and total CO₂ exchange rates were all greater in the unmanaged marsh. Apparently, the more reduced soil conditions in the managed marsh (Figure 2c) were responsible for decreased growth. Anaerobic soil conditions have been demonstrated to cause reduced plant nitrogen uptake and reduced growth in *Spartina alterniflora* (Koch et al., 1990; Morris and Dacey, 1984). The higher concentrations of interstitial ammonium in the managed marsh compared to the unmanaged (Figure 5) suggest that nitrogen uptake may be lower in the managed marsh. Koch and Mendelssohn (1989), in a greenhouse experiment, and Mendelssohn and McKee (1988), under field conditions, found that interstitial ammonium can accumulate when *S. alterniflora* is exposed to more anaerobic conditions, especially in the presence of sulfide. Since nitrogen is important for photosynthesis (Sage and Pearcy, 1987), reduced ammonium uptake may play a role in the lower leaf and total CO₂ exchange rates (Figure 9a-b) observed in the managed marsh. Lower net photosynthesis

rates measured in *S. alterniflora* under extremely anaerobic conditions have been attributed to the potential for inadequate aeration of the root rhizosphere via aerenchyma (Pezeshki et al., 1989).

SUMMARY

Drawdowns have been implemented at the Fina LaTerre site from the time management began in 1985 through 1989. The following summary is based on data collected during a drawdown year only and pertains only to the southern portion of the managed area and the unmanaged reference area south of Falgout Canal.

1. At Fina LaTerre the managed marsh had more organic soils, more reduced soil conditions, and higher interstitial salinities than in the adjacent unmanaged marsh in the 1989 growing season.
2. Net aboveground primary productivity of *Spartina patens* at Fina LaTerre was lower in the managed marsh, which had more reduced soil conditions and higher salinities, than in the adjacent unmanaged marsh during the 1989 growing season. In addition, stem density, leaf area, and live biomass of *S. patens* were greater in the unmanaged marsh during the 1989 growing season.
3. Leaf CO₂ exchange rate (per unit area of leaf surface) and total CO₂ exchange rate (per unit area of marsh surface) of *S. patens* were greater in the unmanaged marsh than in the managed marsh at Fina LaTerre during the 1989 growing season.
4. Although I cannot unequivocally state the reason for the lower primary productivity in the managed marsh compared to the unmanaged marsh, the more reduced soil conditions

in the managed marsh is a likely factor. In addition, although interstitial salinity levels were, on average, statistically significantly higher in the managed marsh, the difference was not great enough to be ecologically significant. It is impossible to state, unequivocally, that the differences observed between the managed and unmanaged marshes were due to the management plan employed at the Fina LaTerre Mitigation Bank because there is no data available describing the two marshes prior to instigation of management in 1985.

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CHAPTER 3

VEGETATION AND SOIL RESPONSE TO MARSH MANAGEMENT AT ROCKEFELLER WILDLIFE REFUGE, LOUISIANA

INTRODUCTION

An investigation into the effects of marsh management on both edaphic factors and the productivity of emergent macrophyte vegetation was conducted at Rockefeller State Wildlife Refuge during 1989 which was a drawdown year. The management goals at Rockefeller Wildlife Refuge are to reduce salinity, increase marsh-to-water ratios, and enhance the production of aquatic vegetation thereby improving waterfowl habitat. In addition, these goals must be met while accommodating the production of shrimp and other aquatic species. Overall, this management plan is attempting to restore conditions present in the area prior to hydrologic alterations by navigation canals.

The results of management on edaphic factors and emergent macrophyte vegetation was compared to conditions occurring in a nearby unmanaged marsh. This study was conducted concurrently with a similar investigation (reported in Chapter 2) at Finia LaTerre, LA. These two studies are unique because the conditions within the managed marsh are compared to those of adjacent or nearby unmanaged marshes. The objective of this study was to determine (1) the effect of marsh management on the primary productivity and vigor of the dominant emergent marsh vegetation (*Spartina patens*) and (2) which soil variables may be controlling this vegetative response to marsh management.

In order to meet these objectives the following questions were addressed. First, what effect does marsh management have on edaphic factors (i.e., soil oxidation state and interstitial concentration of nutrients, salinity, and sulfide) as compared to nearby

unmanaged marshes? Second, what influence does marsh management have on the productivity of emergent vascular plant production as compared to nearby unmanaged marshes? And third, what conclusions can be made about the use of marsh management in light of the information this study has provided?

MATERIALS AND METHODS

Unit 4 of Rockefeller Wildlife Refuge, where this study was conducted, has been managed since 1958 in such a way that a drawdown occurs every 3 to 4 years via a variable-crest flap-gated structure having seven gates. During nondrawdown years, water levels are maintained at or near the marsh surface by setting the stoplogs at a level 6 inches below the marsh surface in order to encourage the growth of aquatic plant species. I studied this marsh in 1989, during a drawdown year. The stoplogs were set 18 inches below the marsh surface and the gates were down and flapping out (meaning that water inside the management area could push against the gates and drain out but if water levels in the adjacent canal rose to levels higher than in the management area this water could not enter through the gates). The drawdown occurred from mid-February to mid-June except for short periods in May and June when the gates were opened to allow larval shrimp to enter the management area. Unit 4 is a brackish marsh dominated by *Spartina patens*.

Sampling Schedule

In the spring of 1989, twenty randomly chosen sample sites, of which 10 were burned and 10 unburned, were located within a managed marsh (Rockefeller Unit 4, Figure 10), and within an unmanaged marsh (located at the northern end of East Little Constance Bayou, Figure 11) in the Rockefeller State Wildlife Refuge and Game Preserve. Since the unmanaged marsh was located along a Bayou which exhibited a

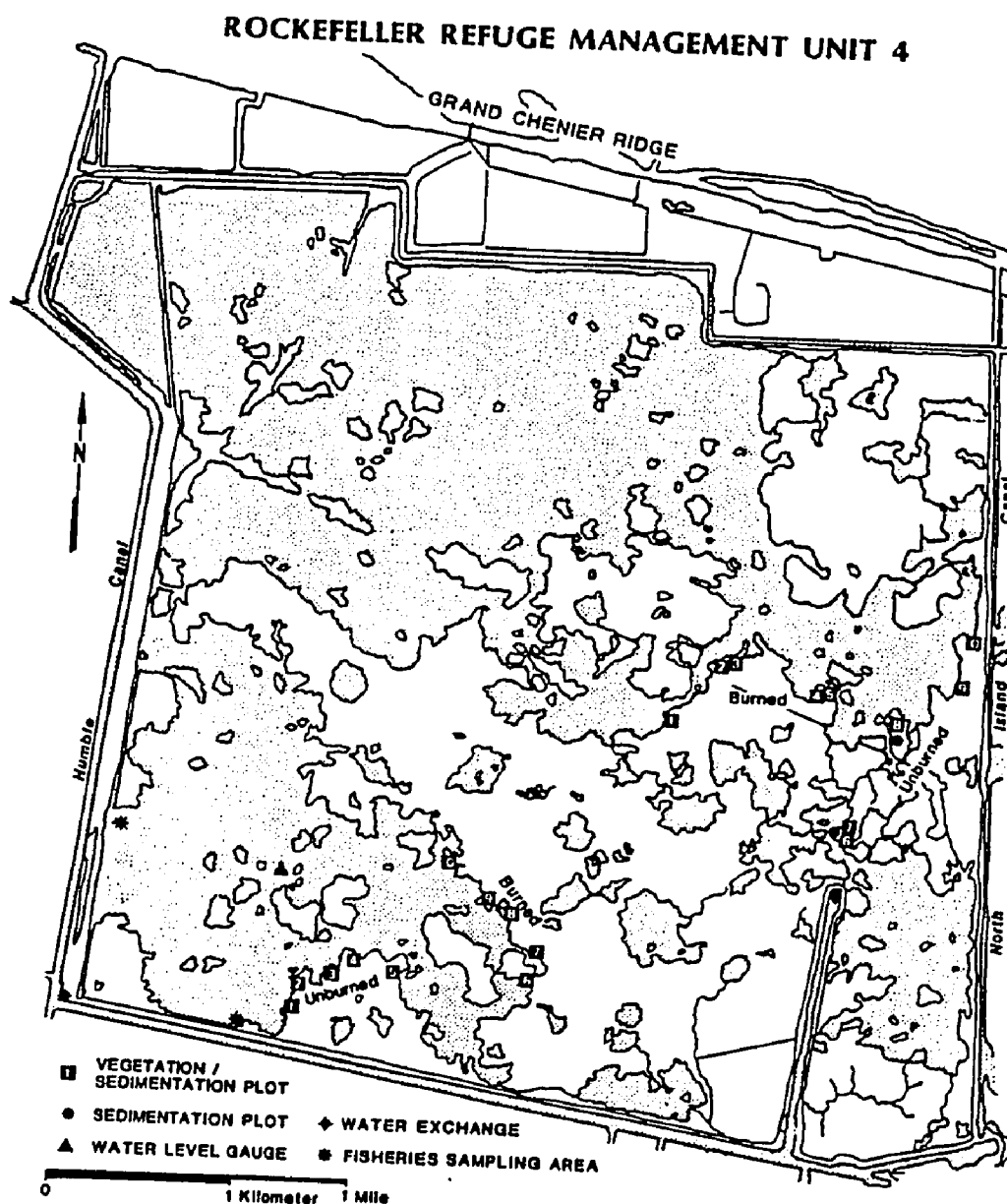


Figure 10. Map of the sample sites in the Unit 4 management area at Rockefeller Wildlife Refuge (adapted from Cahoon and Groat, 1990).

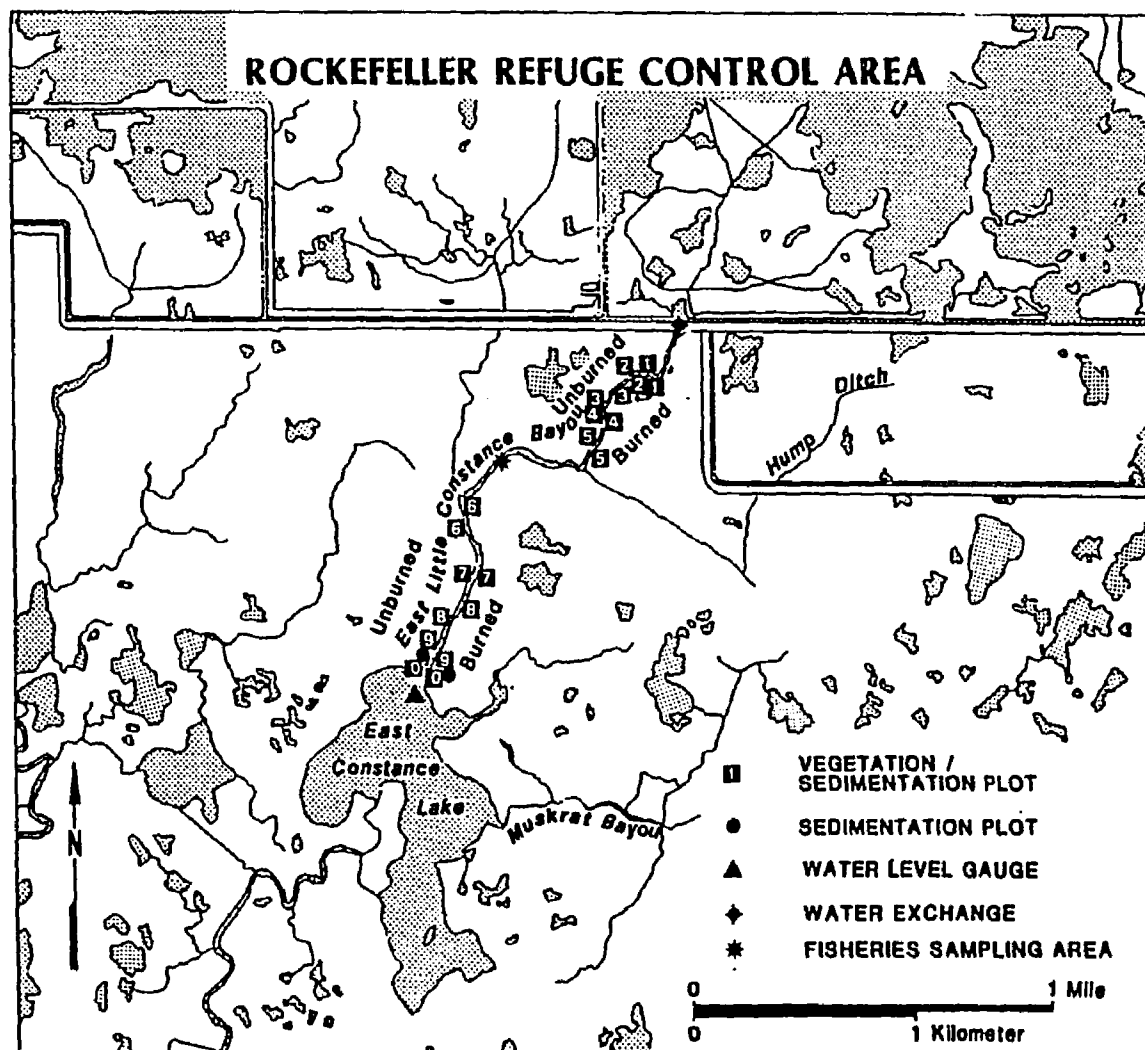


Figure 11. Map of sample sites in the unmanaged marsh at Rockefeller Wildlife Refuge (adapted from Cahoon and Groat, 1990).

typical streamside effect, plots were placed 10m beyond the inland edge of streamside vegetation. Sample plots in the managed marsh were likewise placed 10m in from the marsh-water interface even though a typical streamside effect was not evident.

Sampling was conducted four times during the investigation, on May 31 through June 1, August 21-22, October 4-6, and November 29-December 1, 1989. Because of instrumentation problems in May and August, and the shorter days in October and November, samples were not collected from every site during every sampling period. In May, nine burned and ten unburned sites were sampled in the managed marsh. However, only one burned and six unburned sites were sampled in the unmanaged marsh. In August, five sites within the burned and five sites within the unburned areas of both the managed and unmanaged marshes were sampled. In October and November, the shorter days prevented sampling at two of the burned sites in the managed marsh. However, all sites were sampled in the managed unburned area and in the burned and unburned areas of the unmanaged marsh.

Analysis of Vegetation Response

Net leaf CO₂ exchange rates of *S. patens*, the dominant species at Rockefeller State Wildlife Refuge and Game Preserve, were measured with a portable infrared CO₂ analyser (LCA-2), an ADC air supply unit or ADC air supply unit with mass flow, and a Parkinson Leaf Chamber (Analytical Development Co., Ltd.). Two to 4 measurements (using an open gas exchange system) were made on mature, intact leaves at each sample site. A generator-powered 300W, 125V Sylvania Wide-Flood light was used when photosynthetically active radiation (PAR) was below 1300 $\mu\text{mole m}^{-2} \text{s}^{-1}$. DeJong et al. (1982) and Pezeshki et al. (1987) reported that net photosynthesis of *S. patens* was light saturated at approximately 800 $\mu\text{mole m}^{-2} \text{s}^{-1}$ under field conditions. This allowed measurement of light saturated exchange rates, which were calculated using equations adapted from von Cammerer and Farquahar (1981). Net CO₂ exchange rates were

expressed as μmoles of CO_2 exchanged per m^2 of leaf surface per second. Total CO_2 exchange rates (per unit area of marsh) were determined by multiplying leaf CO_2 exchange with leaf area per square meter of marsh.

Clip plots (0.1m^2) were sampled at each site adjacent to where CO_2 exchange rates were measured. All aboveground plant material was collected, transported to the lab, and separated by species into live and dead categories. Live stems of *S. patens* were counted to determine stem density. Samples were dried to constant weight at 65°C and weighed to the nearest 0.1 g. Plant data from clip plots were used to estimate net aboveground primary productivity (gm^{-2}) of *S. patens* with three different methods, those of (1) Smalley and (2) Milner and Hughes, and (3) the maximum-minimum method as described by Kirby and Gosselink (1976). Net primary productivity is "organic matter stored in plant tissue in excess of respiration during the period of measurement" (Odum 1963:39). The maximum-minimum method estimates productivity on the basis of the difference between the maximum and minimum live aboveground standing crop. Milner and Hughes' method takes into account only changes in live standing crop between frequent harvests during the growing season. Smalley's method estimates net primary productivity on the basis of changes in both live and dead standing crops over time. None of these methods accounts for losses due to decomposition between sampling intervals or losses due to herbivory, and they therefore underestimate productivity (Kirby and Gosselink, 1976). All of these methods were used to determine whether trends in productivity calculated with different estimation techniques.

Leaf area of *S. patens* was measured indirectly by calculating it on the basis of leaf weight. Leaf area of individual leaves (Fina LaTerre samples collected May 17-18) was measured with a Licor LI-3000 leaf area meter; the leaves were weighed, and a linear regression ($y=0.2264+(31.27X)$, $r^2=0.93$) was generated relating leaf area (dependent variable) to leaf weight (independent variable). All subsequent measurements of leaf area

were based on leaf weight using this equation. Leaf weights per 0.1 m² area in May samples were determined using the leaves from all stems in the plot; leaf weight for other months was computed by multiplying the leaf weight of 20 representative stems by the stem density in a plot. All weights and leaf surface areas are presented on the basis of a square meter of marsh.

Soil Parameters

Duplicate, instantaneous Eh (redox potential) readings were made at in the upper 1-2 cm of soil (hereafter referred to as the soil surface) and at a depth of 15 cm at each site. Measurements were made using a calomel reference electrode, brightened platinum electrodes, and a portable digital pH-mV meter. In order to base the readings on a standard hydrogen electrode, 244 mV was added to each reading (Faulkner et al., 1989). Soils were classified as aerated (>300 mV), moderately reduced (100 to 300 mV), reduced (-100 to 100 mV), and highly reduced (<-100mV, [Patrick, 1980]). Eh readings were not corrected for pH.

To determine water depth above or below the soil surface at each site, shallow wells were installed between *S. patens* hummocks. The wells consisted of 150 cm lengths of PVC pipes, which were buried to a depth of approximately 61 cm. Quarter-inch holes were drilled into the buried end of each pipe to allow movement of water in and out. Each pipe was covered on top with a removable cap. Water depth measurements were made at each site on all four sampling dates by removing the cap and dropping a float connected to a line into the pipe. Distance from the pipe top to the midpoint of the float was then measured. The height of each pipe was measured twice during the study. Water depth above or below the soil surface was calculated by subtracting the distance from the top of the pipe from the pipe height.

Soil cores were collected to a depth of approximately 12 cm using an aluminum corer 6 cm in diameter. Soil was extruded into 500 ml centrifuge bottles and sealed. To ensure an anaerobic environment, the samples were purged with nitrogen via a rubber septum in the bottle cap and placed on ice for transport to the laboratory. The samples were centrifuged at 10,000 g at 4° C for 10 minutes. Immediately upon opening the bottles, an aliquot of the supernatant was added to an antioxidant solution (NaOH, ascorbic acid, sodium salicylate) for sulfide determination using a Lazar ISM-146 Micro Ion sensing electrode and a portable Cole-Parmer Model 5985-80 Digi-Sense mV meter (Lazar, 1986).

Salinity of the interstitial water was measured using a Fisher Conductivity Meter Model 152 and pH was measured using an Altex Model 3560 Digital pH meter with a Corning General Purpose Combination Electrode. The water was filtered using a 0.45 micron millipore filter, and an aliquot was then removed and frozen for NH₄-N analysis (U. S. Environmental Protection Agency, 1979). The remainder of the filtered water was acidified with concentrated nitric acid and analysed for essential nutrients (P, K, Ca, Mg, Na, Fe, Mn, Cu, and Zn) using a Fisher inductively coupled plasma argon emission spectrometer (ICAP, Atom Comp Series 800).

Statistical Analyses

Statistical analyses were conducted with SAS software (SAS Institute, 1985). The effects of treatments and month on soil and plant variables were analyzed with a split-plot analysis of variance, with treatment effects (management and burning) in the main plot, and month effects in the subplot. Pairwise comparisons between months were computed with least-squares means when the overall month effect was significant. All hypotheses were tested at the 0.05 probability level unless otherwise indicated.

RESULTS

Soil Response

Water Depth and Soil Reduction

Averaged over time and burning, both surface Eh and 15-cm Eh were less reduced in the managed marsh (surface= 69 ± 24 mV, 15-cm Eh= -5 ± 16 mV) than in the unmanaged marsh (surface= -92 ± 23 mV, 15-cm Eh= -162 ± 9 mV) (treatment significant [surface Eh, $p=0.0030$; 15-cm Eh, $p=0.0001$]) and water level was higher in the unmanaged marsh (treatment significant, $p=0.0208$). The effect of marsh management on water level, surface Eh, and 15-cm Eh varied with month (treatment x month interaction significant [Water level, $p=0.0001$; Surface Eh, $p=0.0001$; and 15-cm Eh, $p=0.0023$], Figure 12a-c). Although the effect of burning on water level varied with month (burn x month interaction significant, $p=0.0006$), this difference may be due more to possible elevational differences between the burned and unburned areas than to any real effect of burning on water level.

In May, when the water level was 22 ± 1.2 cm below the surface of the managed marsh (Figure 12a), surface soils were aerobic (Figure 12b) and soils 15-cm deep were moderately reduced (Figure 12c). Soils in the unmanaged marsh, however, where the water level was only -0.2 ± 1.4 cm from the surface, had highly reduced conditions at the surface and at a depth of 15 cm (Figure 12b and 12c). Since flooding reduces the rate of gas exchange between the atmosphere and soil by a factor of 10,000 (Greenwood, 1961), the respiration of vegetation and soil microorganisms quickly results in the development of reduced soil conditions (Ponnamperuma, 1972).

In August, water levels in the managed marsh had risen to an average depth of 5 ± 1.1 cm above the soil surface (Figure 12a). This resulted in the development of moderately reduced conditions at the surface and at a depth of 15 cm (Figure 12b and 12c). Although Eh was not significantly different in the managed and unmanaged surface

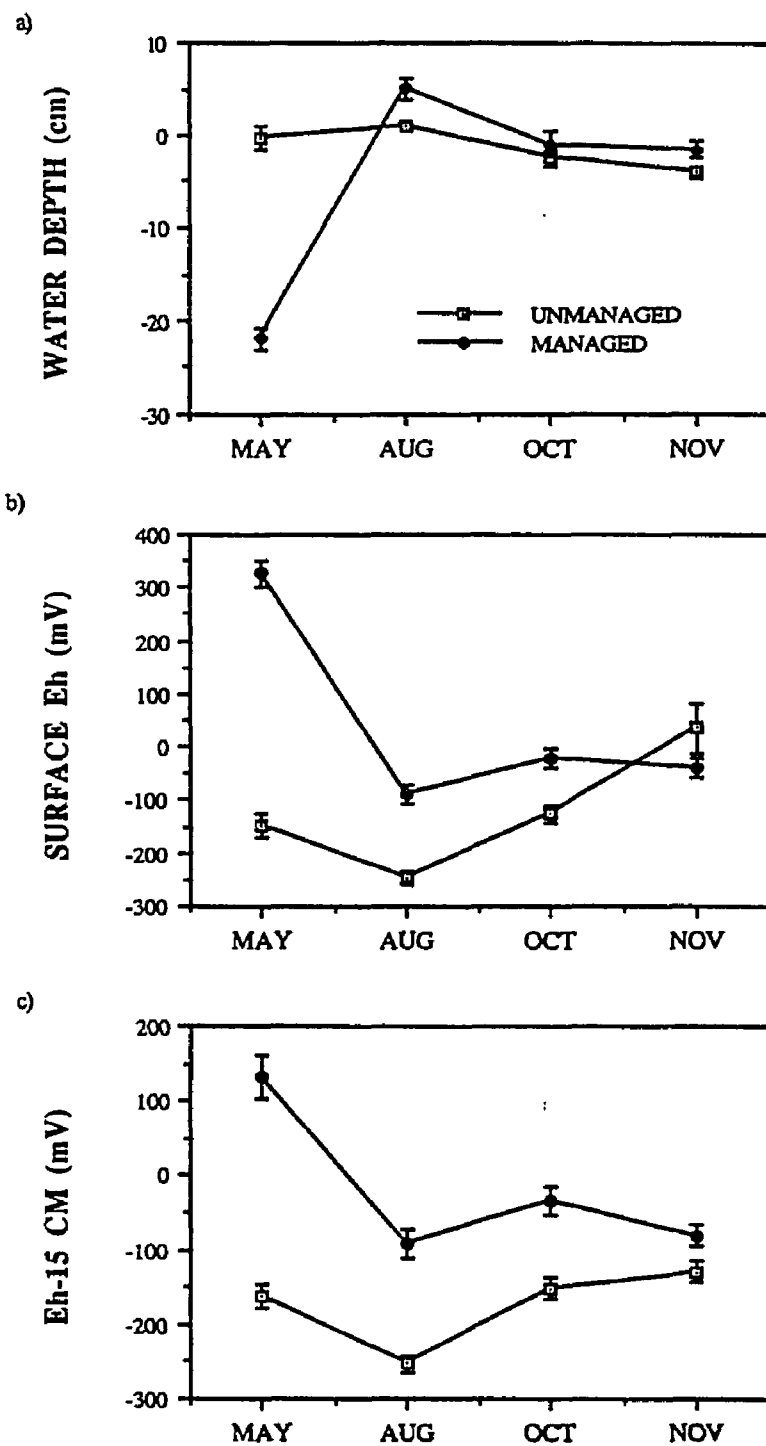


Figure 12. (a) Water depth, (b) surface Eh, and (c) Eh at 15 cm depth in an unmanaged and a managed marsh at Rockefeller Refuge in Cameron-Vermilion parishes, Louisiana, 1989.

soils in August, the unmanaged soils were classified as highly reduced. Eh at 15 cm was significantly lower in the unmanaged marsh than in the managed marsh in August and October. Water level dropped slightly below the soil surface in both the managed and unmanaged marshes in October (Figure 12a), however, soil conditions remained significantly more reduced in the unmanaged marsh than in the managed marsh (Figure 12b-c). Water level, which remained below the soil surface in November (Figure 12a), was similar in the managed and unmanaged marshes. Even so, surface soil conditions were significantly less reduced in the unmanaged marsh (35 ± 47 mV) than in the managed marsh (-38 ± 18 mV, Figure 12b). There were no differences in 15-cm Eh between the managed and unmanaged marshes in November (Figure 12c).

Interstitial pH

Averaged over all treatments, interstitial water pH of the managed marsh was significantly lower than that of the unmanaged marsh (treatment significant, $p=0.0001$). The effect of marsh management on interstitial water pH varied with month (month x treatment interaction significant, $p=0.0001$, Figure 13). The pH of the unmanaged marsh remained between 6.5 and 6.9 on all four sampling dates. In the managed marsh, however, the pH in May (5.1 ± 0.1) was significantly lower than in the other months (Figure 13). This low pH during drawdown was probably a result of the oxidation of sulfide, Fe, and/or Mn which can increase soil acidity (Ponnamperuma, 1972). The increase in water level (Figure 12a) and subsequent development of reduced soil conditions (Figure 12b) was responsible for the increase in pH observed in the managed marsh in August. When acidic soils are flooded, a series of reduction reactions which require H^+ occur, resulting in a trend towards neutral pH (Ponnamperuma, 1972). Burning did not affect pH ($p=0.8308$).

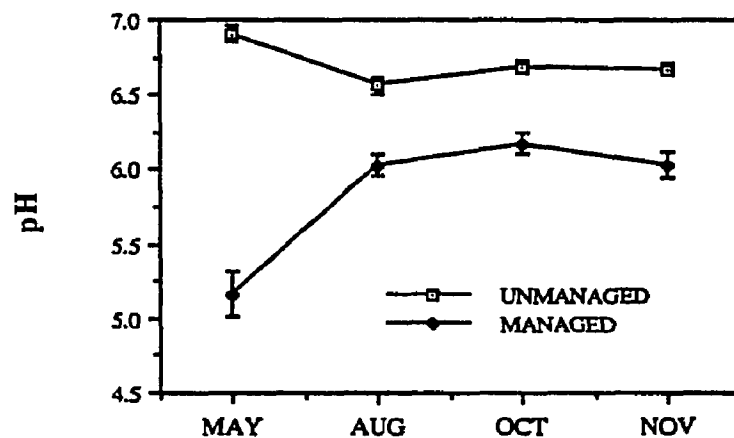


Figure 13. Interstitial water pH in an unmanaged and a managed marsh at Rockefeller Refuge in Cameron-Vermilion parishes, Louisiana, in 1989.

Sulfide Concentration

Interstitial sulfide concentration, averaged over all treatments, was 25 times greater in the unmanaged marsh (11.7 ± 1.7 ppm) than in the managed marsh (0.5 ± 0.1 ppm, treatment significant [$p=0.0001$]). Sulfide was higher in the unmanaged marsh (Figure 14) on all sample dates. No differences in sulfide concentration were found with respect to burning ($p=0.0685$). Sulfide concentrations are so much lower in the managed marsh for several reasons --lower water levels, more soil oxidation, and lower input of sulfate. Flap gates are used to reduce the tidal inflow into the managed marsh that regularly occurs in the unmanaged marsh. This restriction of saline Gulf water from the managed marsh means that sulfate, which is present at high levels in seawater (Weyl, 1970), is not readily available for reduction to sulfide when anaerobic soil conditions develop. Unless the flap gates are opened and tidal waters allowed into the managed marsh, the only source of water for flooding is precipitation. In addition, Connell and Patrick (1968) have reported that Eh must be at or below -150 mV, and pH must be between 6.5 and 8.5, for sulfate to be reduced to sulfide. These conditions were present in soils of the unmanaged marsh on all sample dates at the 15-cm depth, but were not present in the managed marsh. Sulfide, which can be toxic to vegetation at high concentrations (Ponnamperuma, 1972), has been shown to reduce plant growth (Koch and Mendelssohn, 1989; Linthurst, 1979; Mendelssohn and McKee, 1988). The susceptibility of vegetation to sulfide toxicity has been shown to vary with species. The biomass production of *Panicum hemitomon*, a dominant fresh marsh species in Louisiana, was affected by addition of sulfide to hydroponic culture more than was *S. alterniflora*, a dominant salt marsh species (Koch and Mendelssohn, 1989). Mendelssohn and McKee (1989) found reduced aboveground growth of *S. patens* with sulfide accumulation of 85 ppm.

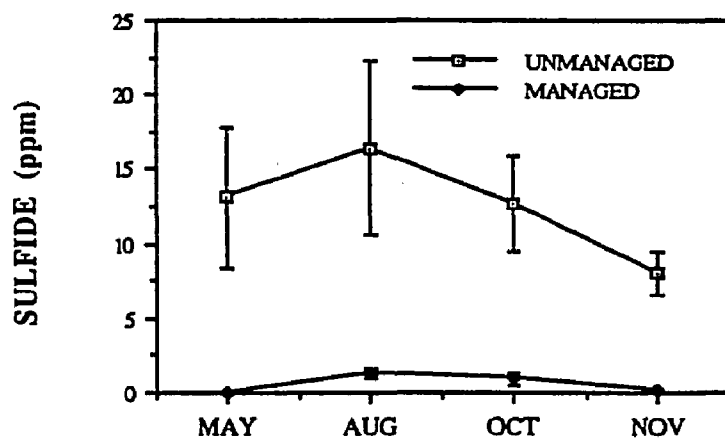


Figure 14. Interstitial water sulfide concentrations over time in an unmanaged and a managed marsh at Rockefeller Refuge in Cameron-Vermilion parishes, Louisiana, in 1989.

Interstitial Ammonium and Phosphorus

Ammonium concentration, averaged over all treatments, was greater in the unmanaged marsh (2.22 ± 0.23 ppm) than in the managed marsh (0.56 ± 0.08 ppm, treatment significant [$p=0.0001$]). The effect of marsh management on ammonium concentration, however, varied with month and with burning (treatment x burn x month interaction significant, $p=0.0486$, Figure 15). In the unmanaged marsh, ammonium concentrations were highest with burning during all four sample periods. In the managed marsh, ammonium concentration was not significantly affected by burning except in May, though levels tended to be higher in the unburned area. In May, the managed unburned marsh had a higher ammonium concentration than the unmanaged unburned marsh, but concentrations were not significantly different in August. In October and November, however, the unmanaged unburned marsh had a higher concentration of ammonium (Figure 15). The unmanaged marsh (burned and unburned) had a seasonal pattern of low ammonium concentration in May and November, and a peak in October (Figure 15). Ammonium concentrations decreased in the managed unburned marsh from May to November (Figure 15).

Phosphorus concentrations, averaged over time and burning, were greater in the unmanaged marsh (2.24 ± 0.3 ppm) than in the managed marsh (0.71 ± 0.2 ppm [treatment significant, $p=0.0016$]). Phosphorus concentrations were not significantly affected by burning ($p=0.1075$) or month ($p=0.1486$). In addition, the effect of marsh management on interstitial P concentrations did not vary with burning (treatment x burn, $p=0.0723$), or with month (treatment x month, $p=0.9867$).

Nitrogen is considered to be the primary growth-limiting nutrient in saline marshes (Linthurst, 1979; Mendelssohn, 1979; Valiela and Teal, 1974). Ammonium accumulates in soils at an Eh of approximately 220 mV or less (Mitsch and Gosselink, 1986). Soil conditions at the surface and 15 cm depth were within this range on all sample dates in the unmanaged marsh (Figure 12b-c). The surface soils of the managed

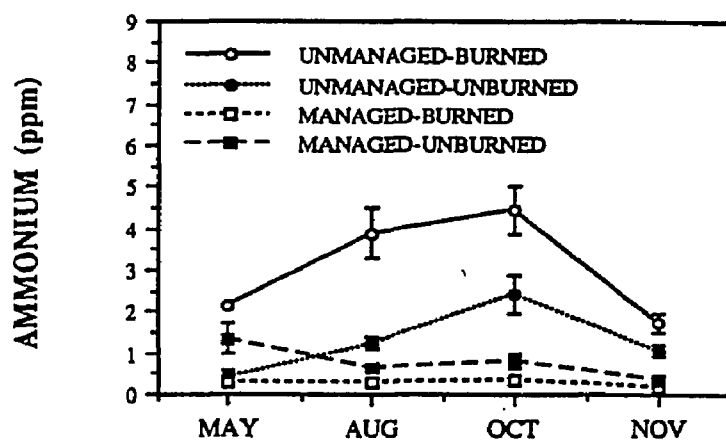


Figure 15. Interstitial water $\text{NH}_4\text{-N}$ in an unmanaged (burned and unburned) and a managed (burned and unburned) marsh at Rockefeller Refuge in Cameron-Vermilion parishes, Louisiana, in 1989.

marsh were also at or below 220 mV in August, October, and November (Figure 12b). Koch and Mendelssohn (1989) reported increased interstitial ammonium in sods of *S. alterniflora* with the addition of sulfides and attributed the increase to reduced plant uptake. Higher P concentrations may be attributed to the increased solubility of P under anaerobic conditions (Redman and Patrick, 1965) and low concentrations of iron for precipitation of P in the unmanaged marsh. Our data support these conclusions since we observed lower ammonium and P in the managed marsh, which had less reduced conditions, virtually no sulfide, and greater biomass production.

Salinity

Averaged over all treatments, interstitial water in the unmanaged marsh was more saline than that in the managed marsh (treatment significant, $p=0.0001$). The effect of marsh management on interstitial water salinity varied with month (treatment x month interaction significant, $p=0.0001$, Figure 16a) and with burning (burn x month interaction significant, $p=0.0041$, Figure 17a). The highest salinities detected in either marsh were measured in May (managed: 5 ± 0.5 ppt; unmanaged 10 ± 1.0 ppt). Salinities decreased significantly in August in both marshes (Figure 16a). Although salinity remained unchanged in the managed marsh on the last two sample dates, it increased in October in the unmanaged marsh (Figure 16a). Burned areas had significantly higher salinities than unburned areas in August and October, but did not significantly differ in May and November (Figure 17a). Salinity levels were lowest in the unburned areas in August. Salinity levels were not significantly different in the unburned areas in May, October, and November (Figure 17a). Salinity levels did not differ significantly within the burned areas on any of the sampling dates.

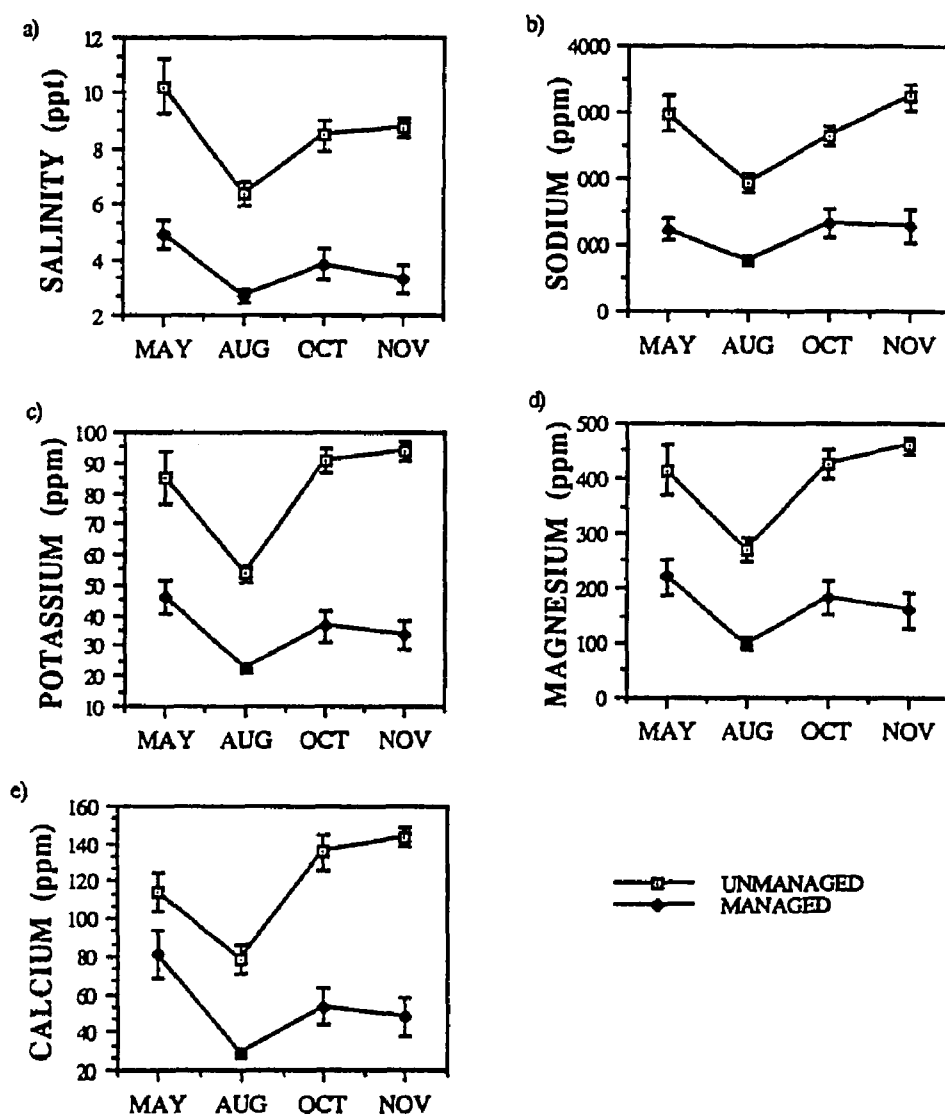


Figure 16. Interstitial water (a) salinity, (b) sodium, (c) potassium, (d) magnesium, and (e) calcium, measured over time in an unmanaged and a managed marsh at Rockefeller Refuge in Cameron-Vermilion parishes, Louisiana, in 1989.

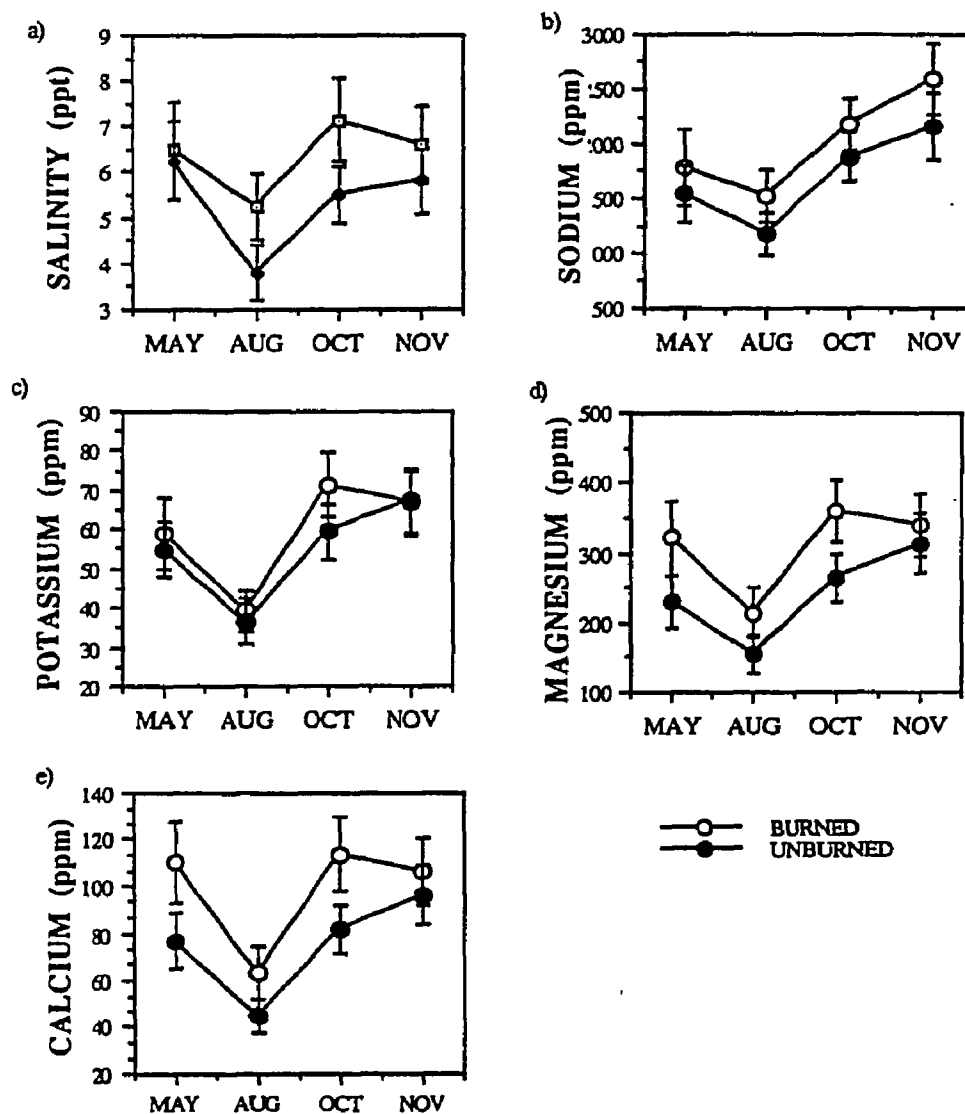


Figure 17. Interstitial water (a) salinity, (b) sodium, (c) potassium, (d) magnesium, and (e) calcium, measured over time in burned and unburned marshes at Rockefeller Refuge in Cameron-Vermilion parishes, Louisiana, in 1989.

Macronutrients--Na, K, Ca, and Mg

The primary cations (Na, K, Ca, Mg) of sea water, exhibited similar responses to the treatments as salinity (Figures 16 a-e, Figure 17 a-e) Averaged over time, the unmanaged marsh had higher concentrations of K, Na, Ca, and Mg (treatment significant, $p=0.0001$ for all four macronutrients). The effect of marsh management on K, Na, Ca, and Mg varied with month (treatment x month interaction significant, $p=0.0001$ for all four macronutrients, Figure 16b-e) and with burning (burn x month interaction significant [K, $p=0.0008$; Na, $p=0.0281$; Ca, $p=0.0001$; Mg, $p=0.0001$], Figure 17b-e). On average, burned areas had higher concentrations of Na, P, Ca, and Mg than unburned areas.

Salinity, Na, K, Mg, and Ca behave similarly within the managed and unmanaged marshes (Figure 16 and 17). Restricting tidal waters from the managed marsh resulted in a less saline environment. The decrease in the concentration of all five of these variables in both the managed and unmanaged marshes in August was related to the heavy precipitation prior to the August sampling period (Richard, 1990). Subsequent increases in salinity, Na, K, Mg, and Ca in the unmanaged marsh were probably due to tidal inundation with more saline water. Salinity increases in the managed marsh were due to a decrease in water level (Figure 12a) via drainage through opened flap gates and evapotranspiration. Sodium, K, Mg, and Ca are all components of seawater and, as such, were more highly concentrated in the tidally influenced unmanaged marsh than in the more tidally restricted managed marsh. Differences in Na, Mg, and Ca associated with burning (Figure 17b-e) may be due to the release of these elements from burned plant tissue.

Iron

Averaged over all treatments, the managed marsh had higher concentrations of Fe (9.386 ± 1.57 ppm) than did the unmanaged marsh (0.5022 ± 0.0731 ppm), (treatment

significant, $p=0.0001$). The effect of marsh management on Fe concentration varied with month (month x treatment interaction significant, $p=0.0006$, Figure 18a) and with burning (treatment x burn interaction significant, $p=0.0506$, Figure 18b). Iron concentrations were low in both the managed and unmanaged marshes in May (Figure 18b). While no significant changes in Fe concentrations occurred in the unmanaged marsh during the investigation, concentrations of Fe increased twelvefold in the managed marsh from May to August (Figure 18b). No significant differences in Fe concentrations were detected between August and October in the managed marsh; however, Fe concentrations decreased by 64% from October to November. Iron concentrations were not different between the burned and unburned areas in the unmanaged marsh. However, in the managed marsh, Fe concentrations were greater in the burned area than in the unburned area (Figure 18a).

Iron is reduced from the ferric to the ferrous form at an Eh of approximately 120 mV or less. At this level of soil reduction, both Fe and Mn act as electron acceptors. Soils which initially have lower pH, such as those in the managed marshes, tend to have a greater release of ferrous Fe when flooded than do soils with higher initial pH (Redman and Patrick, 1965). When redox potentials are low enough for sulfide to form, reducible Fe compounds in the soil will precipitate with sulfide to form insoluble ferrous sulfide (FeS) (Gambrell and Patrick, 1978; Patrick and Mikkelsen, 1971). In the unmanaged marsh, soil conditions are reduced enough for the formation of ferrous Fe and sulfide. However, while sulfide concentrations are, on average, 11.7 ppm in the unmanaged marsh, the scarcity of Fe indicates that Fe was precipitated from solution as insoluble FeS. Ferrous sulfide gives soils a gray or black color (Ponnampura, 1972), such as was observed in the unmanaged marsh. In the managed marsh, there was virtually no sulfide even though soil conditions were reduced enough for sulfide formation at the 15 cm depth. As a result, iron levels were relatively high (Figure 18b).

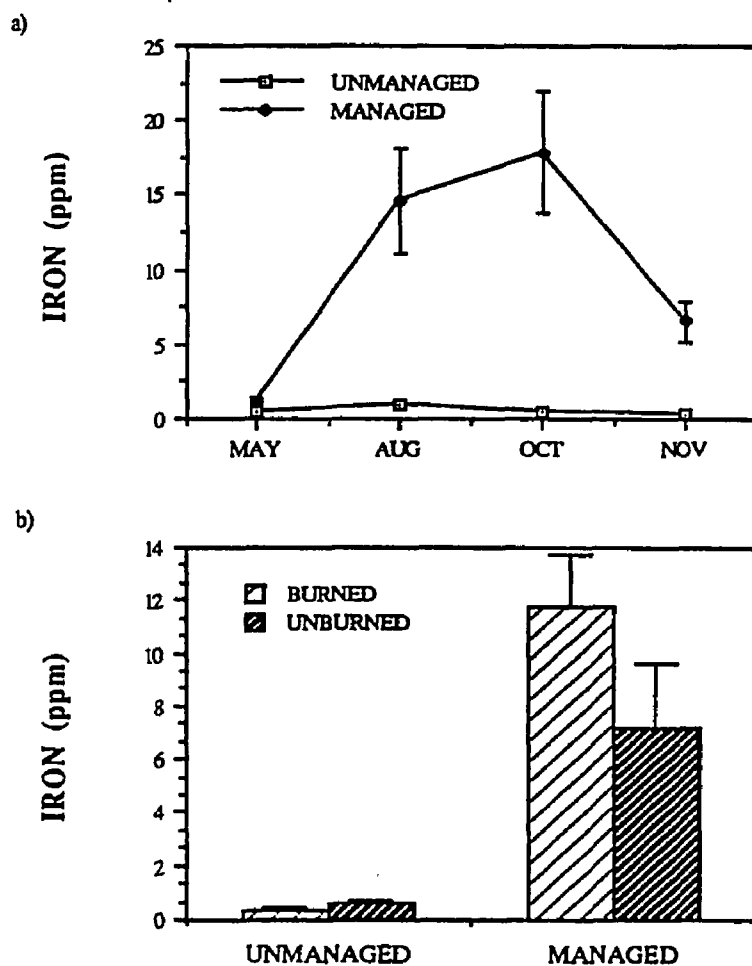


Figure 18. (a) Interstitial water iron in an unmanaged and a managed marsh over time and (b) in an unmanaged (burned and unburned) and a managed (burned and unburned) marsh at Rockefeller Refuge in Cameron-Vermilion parishes, Louisiana, in 1989.

Manganese

The effect of marsh management on Mn varied with month (treatment x month interaction significant, $p=0.0232$, Figure 19). Whereas the managed marsh had higher concentrations of Mn than the unmanaged marsh in May, the reverse was true in August, October, and November. Concentrations of Mn in both marshes decreased in November. Burning did not affect Mn ($p=0.7452$). Manganese can, in the presence of sulfide under anaerobic conditions, precipitate as insoluble MnS (Engler and Patrick, 1975).

The managed marsh had greater concentrations of Zn (0.0642 ± 0.0093 ppm) than the unmanaged (0.0223 ± 0.0055 ppm) marsh (treatment significant, $p=0.0054$) and this response did not differ with month. Overall, significant differences in Zn concentrations occurred with month (month significant, $p=0.0022$). The highest concentrations of Zn were detected in May, and these steadily decreased through October (Figure 19b). Burned and unburned areas had similar Zn concentrations ($p=0.2451$). Concentrations of Cu ($p=0.1075$) were not significantly different between the managed and unmanaged marshes. The overall effect of flooding upon Zn and Cu is a decrease in solubility (Ponnamperuma, 1972). The presence of sulfide could have affected Zn and Cu concentrations through formation of ZnS and CuS (Engler and Patrick, 1975).

Vegetation Response

Stem Density and Leaf Area

The effect of marsh management on stem density varied with month (month x treatment interaction significant, $p=0.0006$). Stem density was significantly greater in the unmanaged marsh in May, but no significant differences between marshes occurred in the other months (Table 4). Burning did not affect stem density ($p=0.2180$). Leaf area was greater in the managed marsh ($1.33 \pm 0.08 \text{ m}^2 \text{ m}^{-2}$ of marsh) than the unmanaged marsh ($0.33 \pm 0.03 \text{ m}^2 \text{ m}^{-2}$ of marsh) throughout the investigation (treatment significant,

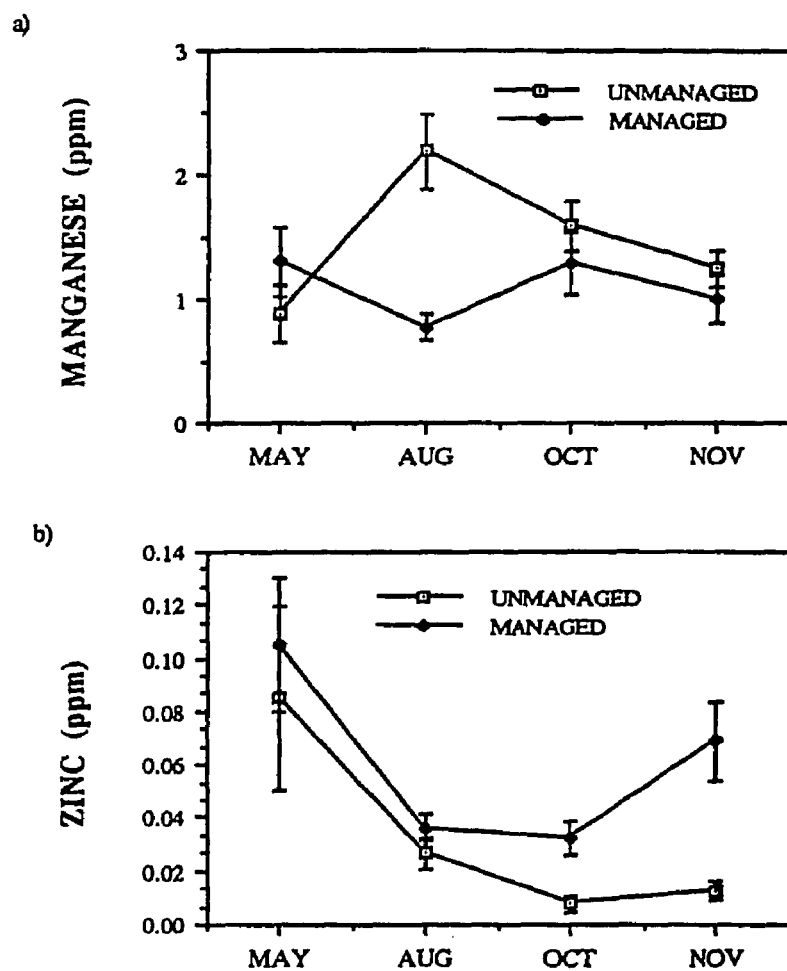


Figure 19. Interstitial water (a) manganese and (b) zinc measured over time in an unmanaged and a managed marsh at Rockefeller Refuge in Cameron-Vermilion parishes, Louisiana, in 1989.

Table 4. Stem density (g m^{-2}) for *Spartina patens* in the managed and unmanaged marshes at Rockefeller State Wildlife Refuge and Game Preserve, Cameron-Vermilion Parishes, Louisiana, during 1989.

MONTH	UNMANAGED	MANAGED
May	187 ± 34	105 ± 10
August	107 ± 18	94 ± 11
October	111 ± 7	128 ± 10
November	117 ± 14	136 ± 12

$p=0.0001$, Figure 20a). No significant differences in leaf area occurred with time in either marsh during the investigation ($p=0.6086$), and burning had no effect on leaf area ($p=0.7809$).

Live Aboveground Biomass

When averaged over all treatments, live aboveground biomass was three times greater in the managed marsh ($2040.0 \pm 135.4 \text{ g m}^{-2}$) than in the unmanaged marsh ($687.7 \pm 56.3 \text{ g m}^{-2}$) (treatment significant, $p=0.0001$). The effect of marsh management on live biomass, however, varied with month and burning (month x treatment x burn interaction significant, $p=0.0298$, Figure 20b). In May, live biomass was similar in all areas except in the unmanaged burned marsh. However, this point was based on only one sample (see methods). Live biomass increased in the managed marsh from May to August in both the burned and unburned areas (Figure 20b). Live biomass decreased from October to November in the managed burned marsh, and biomass in the unburned area was therefore significantly greater in November. Live biomass did not significantly differ over time or between burned and unburned areas in the unmanaged marsh. *S. patens* made up 82-99% of the live vegetation in the unmanaged clip plots and 92-97% of the live vegetation in the managed clip plots at Rockefeller.

Dead and Total Aboveground Biomass

More dead aboveground biomass was present in the managed marsh ($1325.6 \pm 178.6 \text{ g m}^{-2}$) than in the unmanaged marsh ($590.2 \pm 68.1 \text{ g m}^{-2}$) (treatment significant, $p=0.0348$). Dead biomass was significantly greater in November than in May (month significant, $p=0.0348$) and was greater when burning did not occur (burn significant, $p=0.0001$, Figure 21a-b). The unburned managed marsh had greater dead biomass than the burned managed marsh or the unmanaged marsh (burned and unburned, treatment x burn significant, $p=0.0485$, Figure 21b).

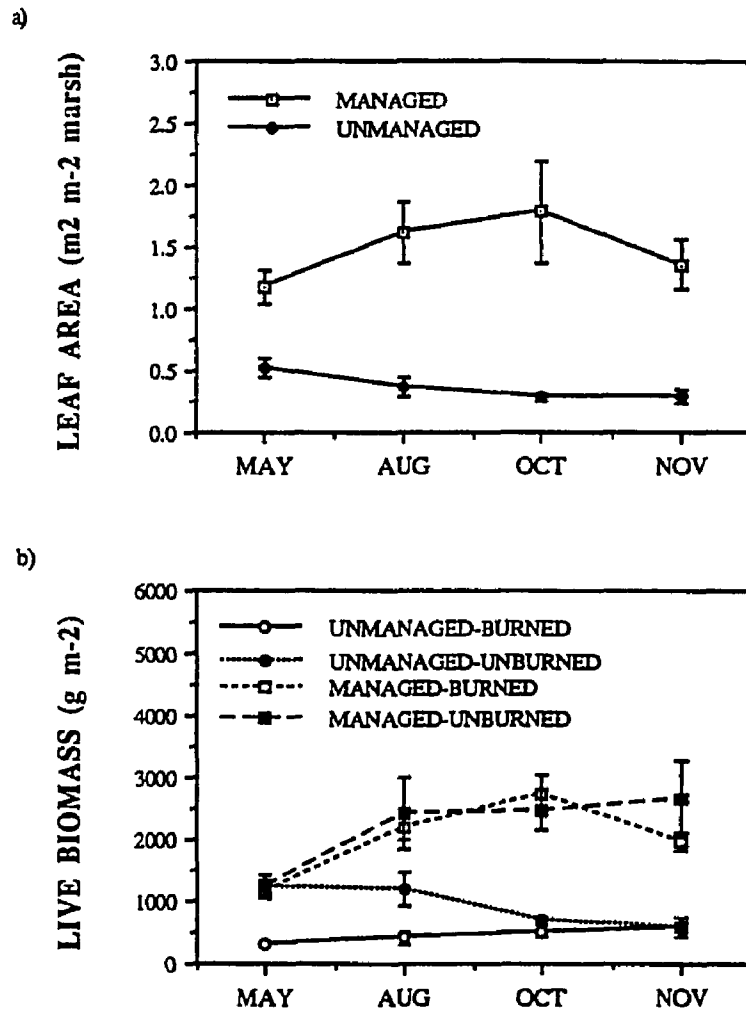


Figure 20. *Spartina patens* (a) leaf area measured over time in an unmanaged and a managed marsh and (b) live biomass measured over time in an unmanaged (burned and unburned) and a managed marsh (burned and unburned) at Rockefeller Refuge in Cameron-Vermilion parishes, Louisiana, in 1989.

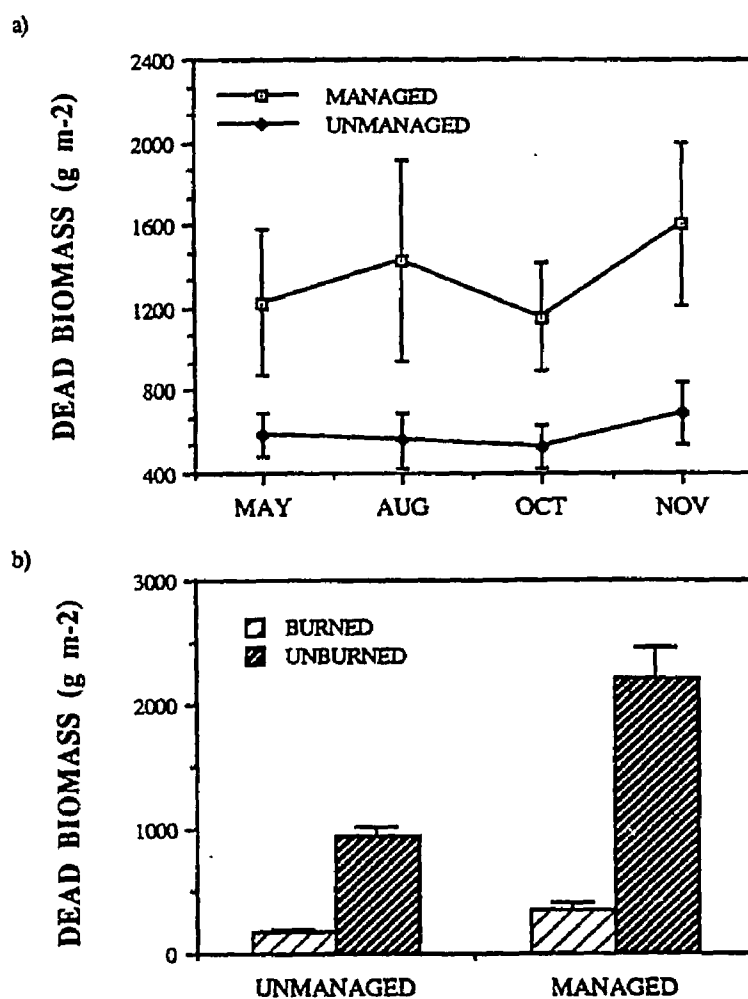


Figure 21. *Spartina patens* dead aboveground biomass measured in (a) an unmanaged and a managed marsh over time and (b) in an unmanaged (burned and unburned) and a managed (burned and unburned) marsh at Rockefeller Refuge in Cameron-Vermilion parishes, Louisiana, in 1989.

Total aboveground biomass, when averaged over all sample dates and burning, was 2.5 times greater in the managed marsh than in the unmanaged marsh (treatment significant, $p=0.0001$). Burning resulted in an average of 51% less total biomass (burn significant, $p=0.0002$), primarily due to its effect on the dead component. Total biomass exhibited a seasonal trend; the lowest amounts occurred at the beginning and end of the growing season (month significant, $p=0.0094$, Figure 22).

Net Primary Productivity

The managed marsh had greater net primary productivity than the unmanaged marsh (treatment significant [maximum-minimum, $p=0.0053$; Smalley, $p=0.0422$; Milner and Hughes, $p=0.0049$], Table 5). Estimates of total net primary production differed with the method used. The Smalley method, which accounts for plant mortality between sampling intervals, yielded a higher estimate than the maximum-minimum method or the Milner and Hughes method which do not account for plant mortality between sampling intervals. No significant differences in productivity were associated with burning (maximum-minimum, $p=0.2145$; Smalley, $p=0.4060$; Milner and Hughes, $p=0.6708$).

Leaf CO₂ Exchange Rate

The effect of marsh management on leaf CO₂ exchange rates varied with month (month x treatment interaction significant, $p=0.0005$, Figure 23a). Since the covariable, hour, did not significantly affect leaf CO₂ exchange rates ($p=0.1515$), values are not adjusted for the time of day when measurements were made. The relatively high leaf CO₂ exchange rates measured in both the managed and unmanaged marshes in May were not significantly different. In August, the rate of leaf CO₂ exchange in the managed marsh was less than the rate in the unmanaged marsh. In October, however, the exchange rate was greater in the managed marsh. Both marshes had the same rate of leaf CO₂ exchange in November (Figure 23a). Whereas leaf CO₂ exchange rates in the unmanaged marsh

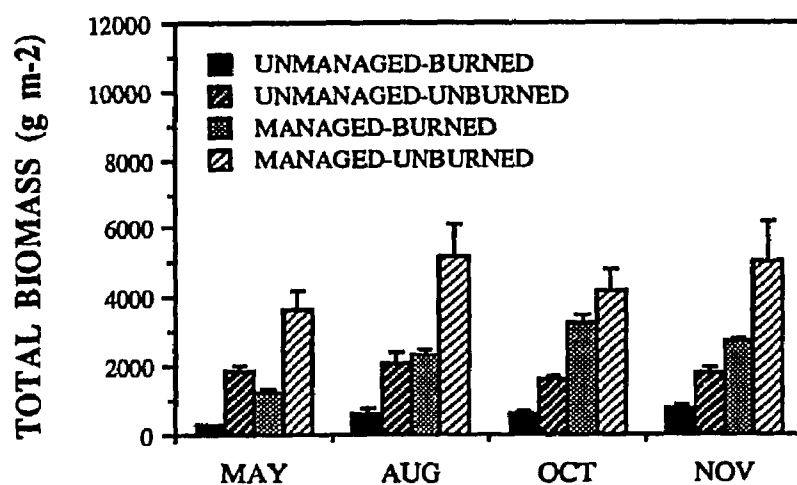


Figure 22. *Spartina patens* total aboveground biomass in an unmanaged (burned and unburned) and in a managed (burned and unburned) marsh at Rockefeller Refuge in Cameron-Vermilion parishes, Louisiana, in 1989.

Table 5. Estimates of net primary productivity (g m^{-2}) for *Spartina patens* in managed and unmanaged marshes at Rockefeller State Wildlife Refuge and Game Preserve, Cameron-Vermilion Parishes, Louisiana, during 1989.

SITE	N	MAX-MIN	SMALLEY	MILNER-HUGHES
Managed	8	2373.5 \pm 319.7 a	4252.7 \pm 1135.1 a	2414.7 \pm 310.4 a
Unmanaged	6	905.9 \pm 158.3 b	921.5 \pm 355.9 b	403.7 \pm 284.7 b

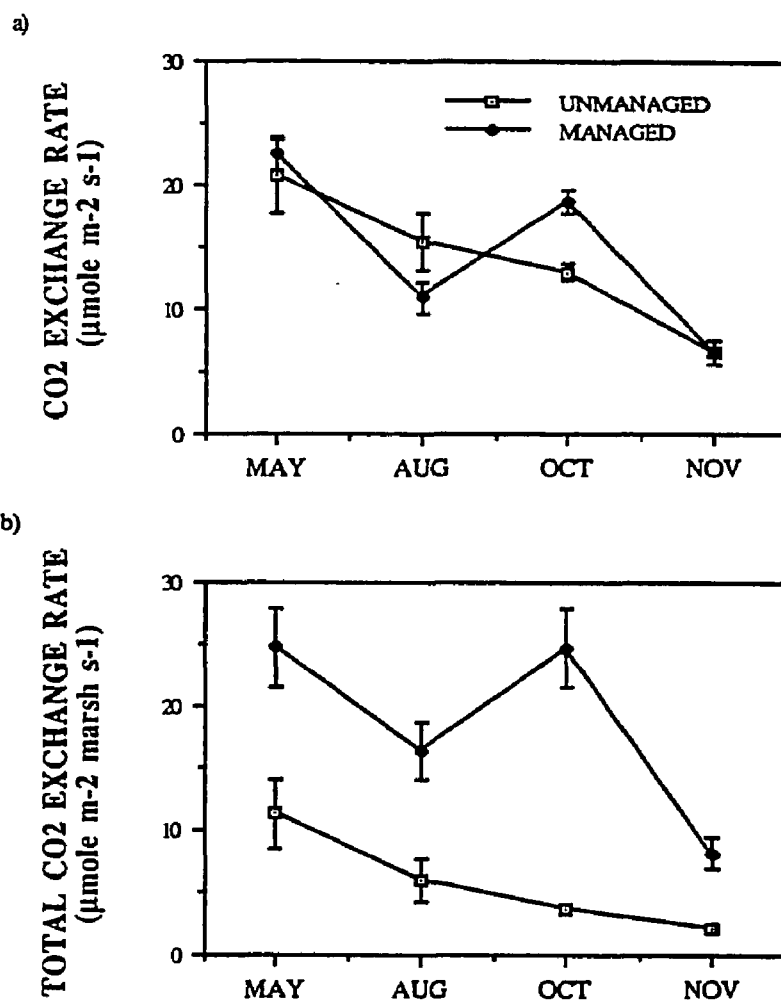


Figure 23. (a) Leaf CO₂ exchange rates and (b) total CO₂ exchange rates for *Spartina patens* in a managed and an unmanaged marsh at Rockefeller Refuge in Cameron-Vermilion parishes, Louisiana, in 1989.

decreased steadily from May to November, the leaf CO₂ exchange rates measured in the managed marsh fluctuated from month to month with a significant overall decreasing trend (Figure 23a). Burning had no significant effect on leaf CO₂ exchange rates in either the managed or unmanaged marsh ($p=0.9704$).

Total CO₂ Exchange Rate

Total CO₂ exchange rates, when averaged over all sampling dates, were significantly higher in the managed marsh than in the unmanaged marsh (treatment significant, $p=0.0001$, Figure 23b). Since the covariable, hour, did not significantly affect total CO₂ exchange rates ($p=0.9133$), values were not adjusted for the variation associated with time of day. Total CO₂ exchange rates decreased over time in both the managed and unmanaged marshes (treatment x month interaction significant, $p=0.0018$). As with leaf CO₂ exchange rates, this decline was steady in the unmanaged marsh but fluctuated in the managed marsh (Figure 23b). In the managed marsh, the highest total CO₂ exchange rates were measured in May and October and the lowest were measured in November (Figure 23b). Burning did not affect total CO₂ exchange rates ($p=0.2800$).

The observed differences in total CO₂ exchange rate were due not to higher rates of CO₂ exchange per unit area of leaf, but to the greater leaf area in the managed marsh as compared to the unmanaged marsh (Figure 20a). Interestingly, burning the marsh did not result in differences in total CO₂ exchange rate or leaf area between the burned and unburned areas in either the managed or unmanaged marshes. The 50% decline in leaf (Figure 23a) and total (Figure 23b) CO₂ exchange rates in the managed marsh from May to August may have been the result of the large increase in water level (Figure 12a) and the associated decrease in surface and 15-cm Eh between May and August. Pezeshki et al. (1989) reported a temporary decrease in photosynthesis of *Spartina alterniflora* with development of reduced soil conditions.

Turitzan and Drake (1981) reported a seasonal decline in the photosynthetic efficiency of *S. patens* and *Distichlis spicata* in a Chesapeake Bay salt marsh. They attributed part of this decline to natural seasonal changes in the canopy structure. As fall approached, the canopy changed from an erect to a horizontal orientation, which resulted in a change in light distribution within the canopy. De Jong et al. (1982) also observed a seasonal decline in the photosynthesis of *S. patens*. They reported a stronger seasonal decline in *S. patens* than in either *Scirpus olneyi* or *D. spicata* although all three species had similar *in situ* CO₂ exchange rates on a leaf area basis.

The soil parameters monitored during this investigation (Eh, interstitial water salinity, sulfide concentrations, and water level) indicate that soil conditions were more favorable for vegetation in the managed marsh than in the unmanaged marsh. Water levels were lower and conditions less reduced at the surface and at a depth of 15 cm in the managed marsh than in the unmanaged marsh (Figure 12a-c). Because of the more aerobic soil conditions in the managed marsh, plant roots should have sufficient oxygen for aerobic respiration. However, the more reduced conditions in the unmanaged marsh could result in root oxygen deficiencies, as demonstrated by Burdick et al. (1989) for *S. patens*. Root oxygen deficiencies result in alcoholic fermentation, which is less energy efficient than aerobic respiration, as the primary pathway for carbon metabolism.

Salinity (Figure 16a) and sulfide concentrations (Figure 14) were lower in the managed marsh than in the unmanaged marsh. This was due to the unrestricted tidal flooding of Gulf waters into the managed marsh, which introduced both salt and sulfate into the marsh. The management plan used at Unit 4 reduces this free influx of Gulf waters. In addition, precipitation can be retained within Unit 4, diluting Gulf waters introduced when the management area is flooded. The reduced conditions in the unmanaged marsh allow the reduction of sulfate to hydrogen sulfide. Although sulfide

concentration in the unmanaged marsh was not exceptionally high, a potential for sulfide toxicity in the unmanaged marsh compared to the managed marsh cannot be completely ruled out.

High interstitial salinity has been associated with decreases in net photosynthesis and productivity in *S. patens* (Pezeshki et al., 1988a). In addition, sulfide has been linked to decreased rates of photosynthesis in *S. alterniflora* (Pezeshki et al., 1988b) as well as decreased growth in this species (King et al., 1982; Koch and Mendelssohn, 1989; Mendelssohn and McKee, 1988). Sulfide-induced inhibition of photosynthesis and reduced growth are related to decreased nitrogen uptake (King et al., 1982; Koch and Mendelssohn, 1989; Mendelssohn et al., 1982). The higher interstitial ammonium and sulfide concentrations along with the lower leaf and total CO₂ exchange rates measured in the unmanaged marsh as compared to the managed marsh agree with these conclusions. In addition, Koch et al. (1990) reported that sulfide inhibited the activity of alcohol dehydrogenase, an enzyme important in alcoholic fermentation, in *S. alterniflora* and *P. hemitomon*, which resulted in a reduced capacity for the anaerobic generation of energy needed for nutrient uptake under reduced soil conditions.

The management regime at unit 4 at Rockefeller Refuge has produced an environment more favorable to emergent vegetation than that of the unmanaged marsh bordering Little Constance Bayou. Results reported herein support this conclusion on several levels--total CO₂ exchange rates, leaf area m⁻² marsh, biomass measurements (live, dead, and total biomass), and three methods of estimating net primary productivity. The favorable environment is apparently the result of the interaction of several edaphic factors ultimately controlled by water level: soil Eh, interstitial sulfide concentration and salinity.

SUMMARY

Drawdowns have usually occurred every fourth year at Rockefeller Refuge. The following observations are based on data collected during a drawdown year.

1. The managed marsh at Rockefeller had less reduced soil conditions, lower interstitial salinity, and lower interstitial sulfide concentrations than did the unmanaged marsh during the growing season of 1989.
- 2 Net aboveground primary productivity of *Spartina patens* was three times greater in the managed area at Rockefeller than in the unmanaged area, probably because of the more favorable edaphic factors in the managed area during the 1989 growing season. In addition, leaf area and live biomass of *S. patens* were greater in the managed marsh than in the unmanaged marsh at Rockefeller during the 1989 growing season.
3. Total CO₂ exchange rates of *S. patens* were higher in the managed marsh than in the unmanaged marsh at Rockefeller Refuge during the 1989 growing season.

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CHAPTER 4

VEGETATION AND SOIL RESPONSE OF A FRESHWATER FLOATING MARSH AT AVOCA ISLAND, LOUISIANA, TO MARSH MANAGEMENT

INTRODUCTION

Water level (structural) marsh management is being used in many areas of south Louisiana to address a diverse range of objectives which include increasing waterfowl and furbearer use (Craft and Kleinpeter, 1986), excluding salt water and promoting freshwater species, and preventing marsh loss (Simmering et al., 1989). Some monitoring (including monthly salinity measurements, water depth measurements, and vegetation composition) has been conducted in many of these management areas.

A recent study of marsh management (Cahoon and Groat, 1990) included a comprehensive study of two brackish/intermediate marshes. In the present study, every attempt was made to collect vegetation and soil data comparable to that reported in the Cahoon and Groat (1990) report in order to add to the growing body of information available on marsh management. This study is unique because Avoca Island is the only fresh marsh which has been scientifically studied for management effects along the Louisiana coast. In addition, the area consists of floating marsh, defined by O'Neil (1949) as a mat of vegetation floating on a layer of water and freely oscillating with changes in the underlying water column.

The management plan utilized at Avoca Island since 1987, was designed to increase primary productivity by using protective levees to control erosion and water control structures for water level management (SCS, 1988). There are two water control structures at Avoca Island. One is a 10 foot wide variable-crest weir in the northeast and

the other is a 24 inch diameter double-flapgated aluminum culvert in the southeast corner of the management area. The managers plan to lower water levels in the late winter through spring, allow free exchange of water in the summer and fall, and increase water levels inside the management area in late fall and winter to create waterfowl habitat. The objectives of this study were to (1) compare primary productivity of the emergent herbaceous community in a managed and adjacent unmanaged marsh, (2) compare the CO₂ exchange rate of two plant species (*Sacciolepis striata* (L.) Nash and *Hydrocotyle umbellata* L.) present in both the managed and unmanaged marshes and (3) describe those soil variables that may be controlling the vegetative responses to marsh management.

Several questions concerning the effects of structural marsh management on the emergent fresh marsh vegetation at Avoca Island were addressed. First, what effect does marsh management at Avoca Island have on edaphic factors (i.e. soil oxidation state, interstitial conductivity and pH, and interstitial concentration of nutrients) as compared to a nearby unmanaged marsh? Second, what influence does marsh management at Avoca Island have on primary productivity, species composition, species dominance, and CO₂ exchange rates of selected emergent vascular plants as compared to a nearby unmanaged marsh? Third, what conclusions can be made about the use of marsh management in a fresh and/or floating marsh in light of the information this study has provided?

MATERIALS AND METHODS

Experimental Design

The Avoca Island management area is approximately 1.0 mile south of Morgan City, Louisiana in the lower portion of the Atchafalaya Basin approximately 10 miles north of Atchafalaya Bay (29° 37' N Latitude, 91° 8' W Longitude). Sixteen sample sites

were randomly selected from the herbaceous emergent habitat in both the Avoca Island management area and in an adjacent unmanaged marsh in the spring of 1990 (Figure 24). Sampling was conducted 3 times during the course of the investigation: May 24-25, July 12-13, and September 16-17 and 26, 1990. In addition, on August 17, 1990, leaf CO₂ exchange rates were measured on two important species (see below) at one site throughout the day in order to characterize the pattern of CO₂ fixation under ambient light conditions.

Analysis of Vegetation Response

Net leaf CO₂ exchange rates of *Sacciolepis striata* and *Hydrocotyle umbellata* were measured with a portable infrared CO₂ analyser (LCA-2), and ADC air supply unit or ADC air supply unit with mass flow, and a Parkinson Leaf Chamber (Analytical Development Co., Ltd.). Two to 4 measurements (using an open gas exchange system) were made on mature, intact leaves of each species at each sample site. Measurements of CO₂ exchange rate were made at or above 1000 $\mu\text{mol m}^{-2} \text{s}^{-1}$ PPFD (photosynthetic photon flux density) for *S. striata* and above 550 $\mu\text{mol m}^{-2} \text{s}^{-1}$ PPFD for *H. umbellata*. Net CO₂ exchange rates were calculated using equations adapted from von Caemmerer and Farquhar (1981) and expressed as $\mu\text{moles of CO}_2$ exchanged per m^2 of leaf surface s^{-1} . On August 17, 1990, a diel characterization of the pattern of CO₂ fixation over time for *S. striata* and *H. umbellata* was determined by measuring CO₂ exchange rates on 3-6 plants of each species at roughly 1-1/2 hour intervals from 8:00 am until 7:00 pm.

Clip plots (0.1 m^2) for estimates of primary productivity were collected at the 16 sampling sites within both the managed and unmanaged marshes. All aboveground plant material was collected, transported to the lab, and separated by species into live and dead categories. Samples were dried to constant weight at 65 degrees C and weighed to the nearest 0.1 g. Plant data from clip plots were used to estimate net aboveground primary productivity with 3 different methods: the Smalley method and the Milner and Hughes

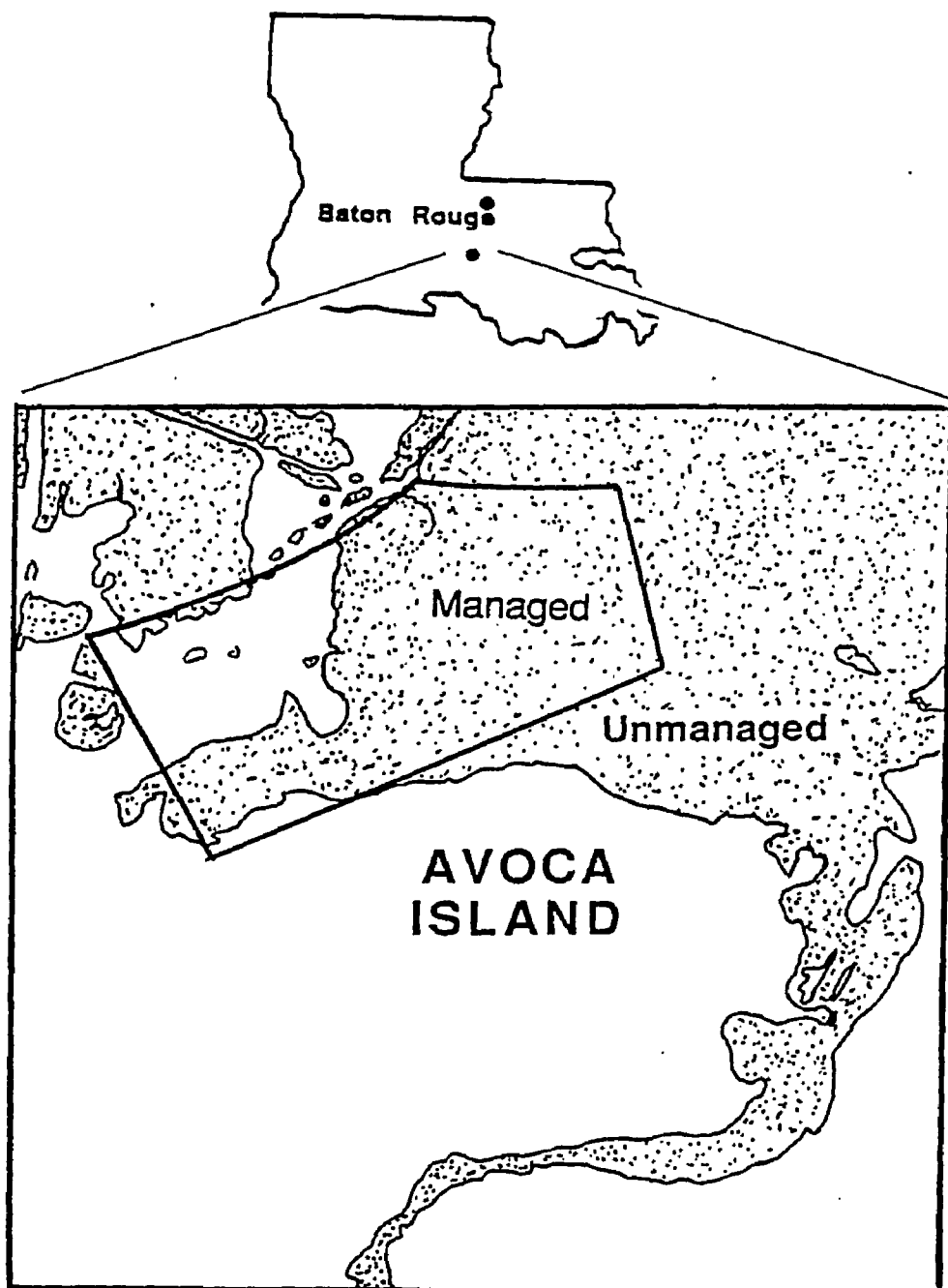


Figure 24. Location of the managed and unmanaged marshes at Avoca Island, Louisiana which were monitored from May through September, 1990.

method as described by Kirby and Gosselink (1976) and Kaswadji (1982), and by calculating peak live + dead biomass. Initial biomass was considered to be zero because little or no vegetation was observed in the study area prior to the first sample date.

The Milner and Hughes method only considers changes in live standing crop during the growing season, while the Smalley method utilizes changes in both live and dead standing crops over time to estimate net primary productivity. None of these methods account for losses due to decomposition or herbivory occurring between sampling intervals and are therefore underestimates of productivity (Kirby and Gosselink, 1976). I used all of these methods in order to determine if trends in productivity agreed using different estimation techniques.

Species composition lists comparing the managed and unmanaged marshes were compiled for each sample date individually and for all sample dates combined. In addition, Species Dominance Values (SDV) were calculated, as described by Mendelssohn and Marcellus (1976), by the formula:

$$SDV = RF + RSC$$

where:

- | | |
|-------|--|
| SDV = | Species Dominance Values |
| RF = | Relative Frequency, which is equal to the number of times the species occurred per the number of times all species occurred. |
| RSC = | Relative Standing Crop, which is equal to the mean standing crop of the species per the sum of the mean standing crops of all species. |

Since RF and RSC are percentage values, the largest possible SDV is 200 which would only occur in monospecific stands. We calculated SDV's for each sample date (May, July, and September) at each marsh (managed and unmanaged), and for the composite species list (from all three sample dates) in each marsh.

Soil Parameters

Depth of the water column from the surface of the floating mat to the underlying clay bottom was measured at each site on each sample date. Duplicate, instantaneous Eh (redox potential) readings were made in the upper 1-2 cm of soil (hereafter referred to as surface Eh) and at a depth of 15 cm. Measurements were made using a calomel reference electrode and a portable digital pH-mV meter. In order to base the readings on a standard hydrogen electrode, 244 mV was added to each reading (Faulkner et al., 1989). Soils were classified as aerated (>300 mV), moderately reduced (100 to 300 mV), reduced (-100 to 100 mV), and highly reduced (<-100 mV, [Patrick, 1980]). Eh readings were not corrected for pH since pH did not vary greatly (see below).

Interstitial water samples were collected with a syringe, put on ice, and transported back to the lab. Conductivity of the water samples (corrected to 25° C) was measured using a Fisher Conductivity Meter Model 152 and pH was measured using an Altex Model 3560 Digital pH meter with a Corning General Purpose Combination Electrode. The water was filtered using a 0.45 micron Millipore filter after which an aliquot was removed and frozen for $\text{NH}_4\text{-N}$ analysis (U. S. Environmental Protection Agency, 1979). The remainder of the filtered water was acidified with concentrated nitric acid and analysed for essential nutrients (P, K, Ca, Mg, Na, Fe, Mn, Cu, and Zn) using a Fisher inductively coupled argon plasma emission spectrometer (ICP, Atom Comp Series 800).

Interstitial water samples were collected for sulfide analysis in May. The samples were added to an antioxidant solution (NaOH, ascorbic acid, sodium salicylate) for sulfide determination using a Lazar ISM-146 Micro Ion sensing electrode and a portable Cole Parmer Model 5985-80 Digi-Sense mV meter (Lazar, 1986).

Soil cores were collected in July using a cylinder (diameter: 10 cm; height: 10 cm) having a volume of 785 cm³. The soil cores were dried at 65 degrees C and weighed to the nearest gram. Bulk density was calculated by dividing the dry weight of the soil by the volume of the coring device. After determining the bulk density, the soil samples were analyzed for pH (with a pH meter and electrode) and extractable nutrients (P [extracted with 0.03 M NH₄F + 0.1 M HCl], Na, K, Ca, and Mg [extracted at pH 7 with 1 M NH₄OAc], Cu, Fe, Mn, and Zn [extracted at pH 7.3 with 0.0005 M DTPA]) with an ICP. Results are reported as the average of duplicate analyses that are within a 10% confidence interval and are based on weight after oven drying. The Wilson Soil Testing Laboratory, Louisiana State University, performed the analyses.

Statistical Analyses

Statistical analyses were conducted with SAS (Statistical Analysis Systems, 1985). The effect of treatments and month on soil and plant variables were analyzed with a split-plot analysis of variance, with treatment effects (management) in the main plot, and month effects in the subplot. When analyzing the CO₂ assimilation data, light was used as a covariable, the treatment effect was in the main plot and the species and month effects were in the subplot. Pairwise comparisons among months were computed with least-squares means when the overall month effect was significant. All hypotheses were tested at the 0.05 probability level unless otherwise indicated. Figures presented in the text are based on the main effects or interaction effects that were statistically significant.

RESULTS AND DISCUSSION

Soil Response

Water Depth and Soil Reduction

Distance from the surface of the floating mat, through the water column, to the underlying clay bottom (hereafter referred to as water depth) was not significantly affected by management ($p=0.3803$). However, water depth decreased significantly ($p=0.0001$) from May to July and from July to September in both the managed and unmanaged marshes (Figure 25a). This decrease in water depth agrees with the description of floating marshes as oscillating with changes in the underlying water column (O'Neil, 1949; Swarzenski, 1987), thus substantiating the floating nature of these marshes. The increasingly shallow water column resulted in less reduced conditions at the 15 cm depth over time (see below). The interaction between management and time did not significantly affect water depth ($p=0.3676$).

Surface soils of both the managed and unmanaged marshes were reduced on all three sample dates (Figure 25b). The reduced condition of the surface soils was probably due to continuous saturation by the underlying water column. However, there were no significant differences in surface Eh due to management ($p=0.1795$), between months ($p=0.4888$), or with the interaction of management with month ($p=0.3911$). Although it appeared that in July and September, the surface Eh of the managed marsh was higher than that in the unmanaged marsh, the large variation in the surface Eh measurements precluded finding statistically significant differences.

However, at the 15 cm depth, the managed marsh had significantly less reduced soils than did the unmanaged marsh when averaged over all three months ($p=0.0101$, [Figure 26a]). Averaged over treatments (managed and unmanaged) Eh at the 15 cm depth varied with month ($p=0.0045$ [Figure 26b]). The interaction of month with management was not significant at the 15 cm depth ($p=0.5063$).

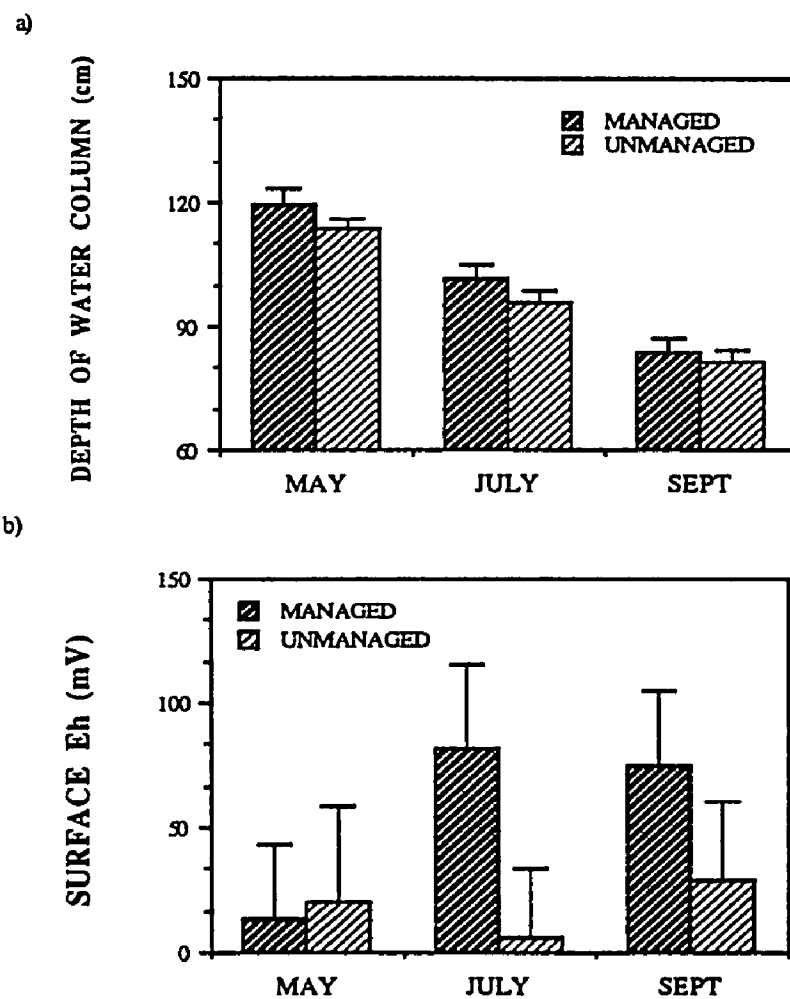
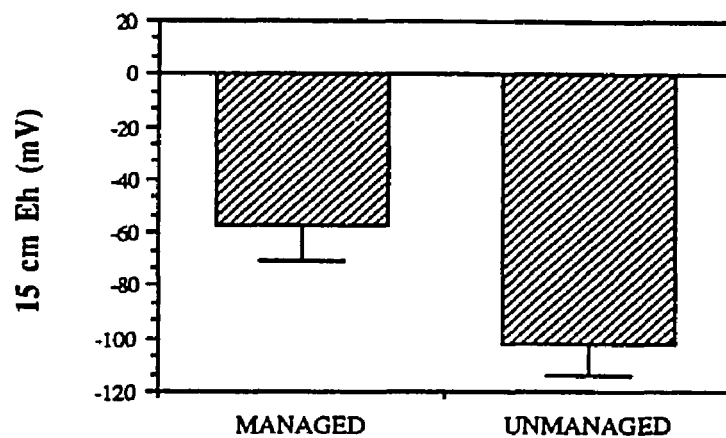


Figure 25. (a) Distance from the surface of the floating mat to the underlying clay bottom, and (b) surface Eh, over time in a managed and an unmanaged marsh at Avoca Island, Louisiana, 1990.

a)



b)

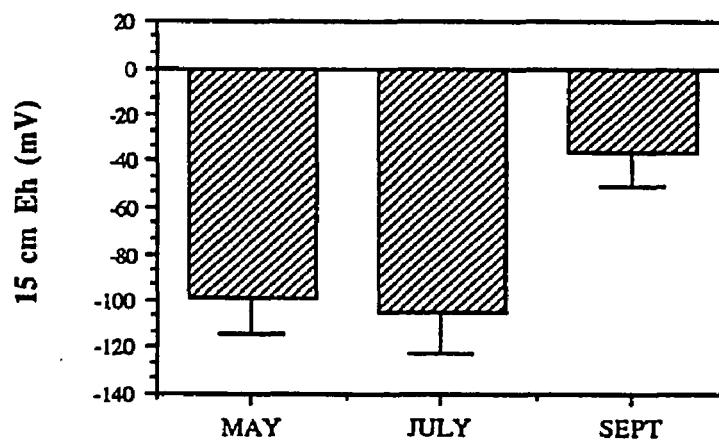


Figure 26. 15 cm Eh averaged (a) over time and (b) over treatments for Avoca Island, Louisiana in 1990.

Interstitial Components

Conductivity and Sulfide

Conductivity, the measure of the capacity of water to convey an electric current, is related to the total concentration of ionized substances in the water sample (American Public Health Association, Inc., 1960). Averaged over time, interstitial water conductivity (Figure 27a) was significantly lower ($p=0.0009$) in the managed marsh than in the unmanaged marsh. Averaged over treatments, interstitial water conductivity varied significantly with month ($p=0.0001$ [Figure 27b]). Conductivity was higher in September than in May or July. The month x treatment interaction was not significant ($p=0.3458$). The observed interstitial water conductivity levels represent salinity values of < 0.5 ppt (i. e. fresh water). The difference between the conductivity of the managed and unmanaged marshes (0.04 umhos/cm) was unlikely to be of biological significance.

Interstitial sulfide concentrations, measured in May, were at undetectable levels in both the managed and unmanaged marshes. Avoca Island is not impacted by saltwater intrusion (SCS, 1988) which means that there is relatively little sulfate input to this area, and hence little sulfide is produced via sulfate reduction, compared to salinity influenced marshes. Interstitial sulfide concentrations were not measured in July or September.

Interstitial pH

Interstitial water pH, when averaged over treatments, varied with month ($p=0.0059$ [Figure 28]). A statistically significant (although not biologically significant) decrease in the interstitial water pH occurred from May to July followed by an increase to the initial level in September. The pH of the interstitial water did not significantly differ between the managed and unmanaged marshes ($p=0.5419$). The interstitial water pH levels observed at Avoca Island are within the range of pH values reported for fresh marshes (range: 4.30-6.30, mean: 5.24) by Chabreck (1972).

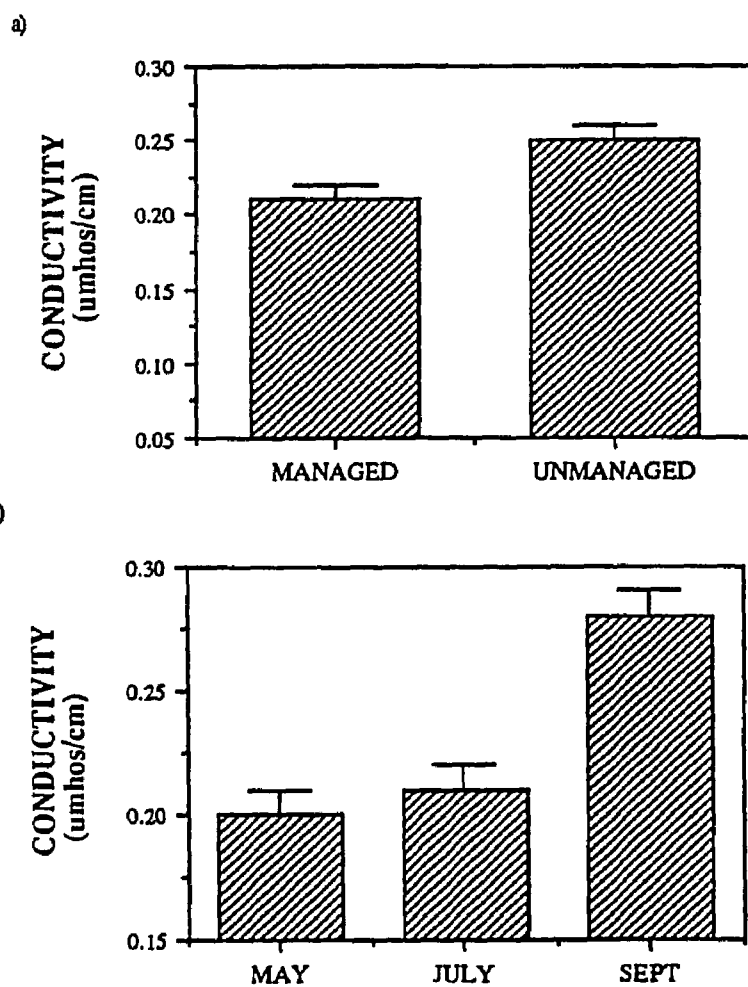


Figure 27. Interstitial water conductivity (a) averaged over time, and (b) averaged over the managed and unmanaged marshes at Avoca Island, Louisiana, 1990.

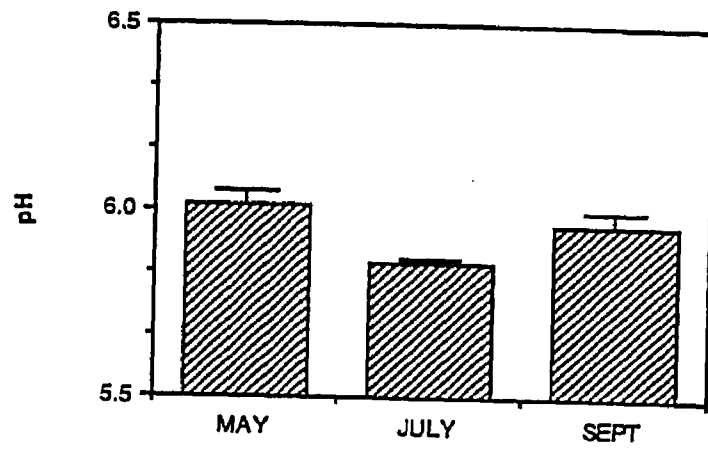


Figure 28. Interstitial water pH averaged over the managed and unmanaged marshes at Avoca Island, Louisiana, 1990.

Interstitial Nutrients

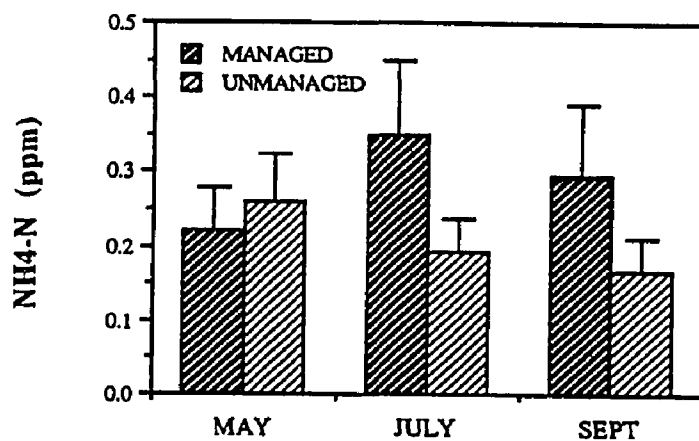
Ammonium and Phosphorus

Although there were no significant differences in ammonium or P concentrations between managed and unmanaged marshes (ammonium: $p=0.1833$; P: $p=0.1351$), with month (ammonium: $p=0.8543$; P: $p=0.9645$), or with the interaction of month x management (ammonium: $p=0.3517$ [Figure 29a]; P: $p=0.2564$ [Figure 29b]), the values tended to be higher in the managed marsh in July and September. Ammonium and P are the two nutrients considered most likely to be growth limiting in Louisiana's coastal wetlands. However, the high organic matter content of wetland soils and the occurrence of reducing conditions, both of which increase the availability of ammonium and P, may help to alleviate the growth limiting effect (Patrick and De Laune, 1977). According to Patrick and De Laune (1976) it is possible that P is not limiting in Louisiana marshes because of the relatively new, unweathered sediments. They support this hypothesis with data from a field experiment in which no growth response was observed when P was added to a streamside stand of *Spartina alterniflora*. Of course, the response in freshwater marshes may be very different. Even if ammonium and P are growth-limiting at Avoca Island, the levels are not significantly different between the managed and unmanaged marshes.

Macronutrients--Ca, Mg, Na, and K

Concentrations of the macronutrients, Mg and Ca, were greater in the unmanaged marsh than in the managed marsh (Mg: $p=0.0005$, [Figure 30a], Ca: $p=0.0013$, [Figure 30b]). There were no significant differences in the macronutrients, Na or K, associated with management (Na: $p=0.2683$; Mg: $p=0.4949$). Na, K, Ca, and Mg varied significantly with month (Na: $p=0.0001$; Ca: $p=0.0019$; Mg: $p=0.0015$; K: $p=0.0001$ [Figure 31 a-d]). Na increased significantly each month. Ca, Mg, and K concentrations did not significantly change from May to July. Both Ca and Mg increased significantly in

a)



b)

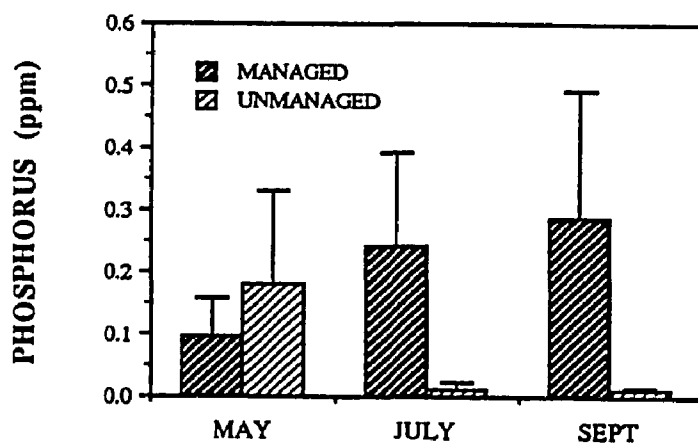


Figure 29. Interstitial water (a) $\text{NH}_4\text{-N}$ and (b) phosphorus in a managed and an unmanaged marsh measured over time at Avoca Island, Louisiana, 1990.

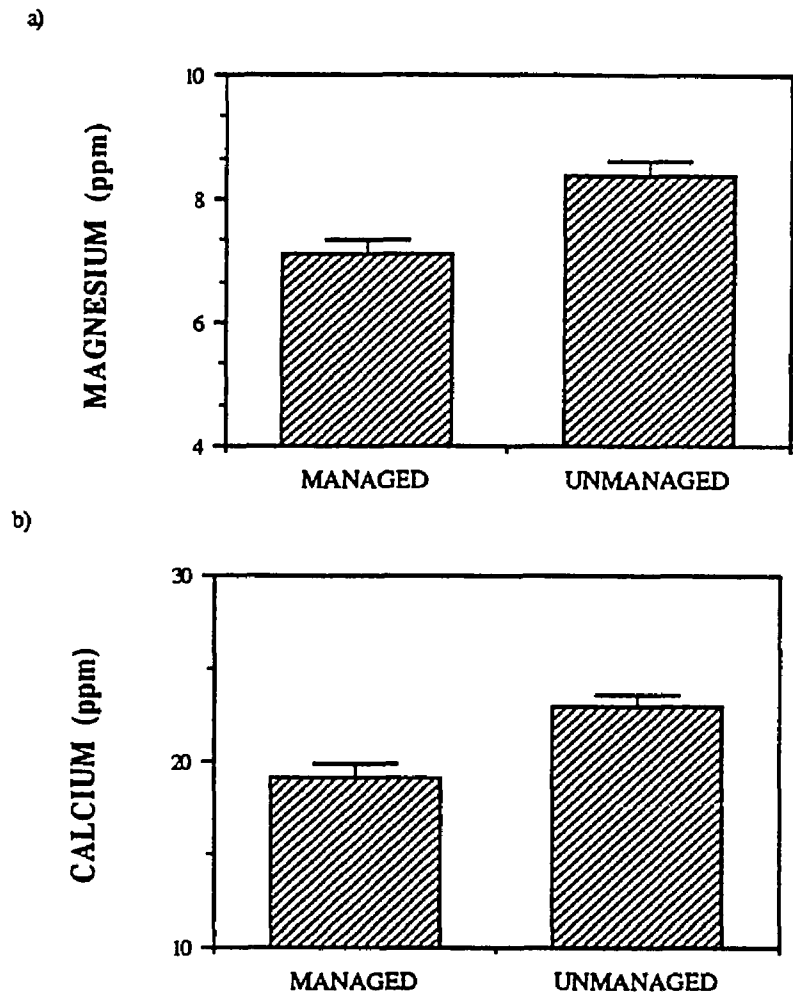


Figure 30. Interstitial water (a) magnesium and (b) calcium averaged over time, at Avoca Island, Louisiana, 1990.

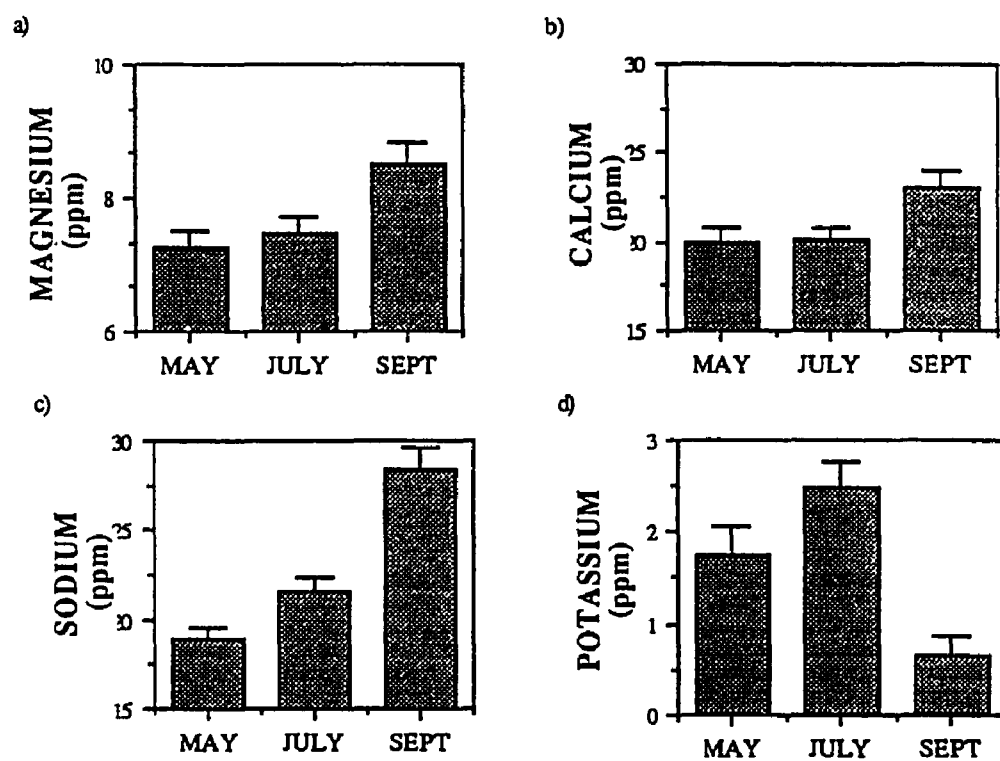


Figure 31. Interstitial water (a) magnesium, (b) calcium, (c) sodium, and (d) potassium, averaged over the managed and unmanaged marshes at Avoca Island, Louisiana, 1990.

September while concentrations of K were significantly lower in September than in either May or July. There were no significant differences due to the interaction of month x management for any of the macronutrients (Mg: $p=0.4850$; Ca: $p=0.2899$; Na: $p=0.8572$; K: $p=0.1814$).

Micronutrients--Mn, Zn, Fe, and Cu

Of the micronutrients Mn, Zn, Fe, and Cu, only Mn concentrations exhibited significant changes during the course of this study. Averaged over all three sample dates, concentrations of Mn were significantly higher in the managed marsh (0.0772 ± 0.01 ppm) than in the unmanaged marsh (0.0488 ± 0.01 ppm [$p=0.0168$]). The effect of marsh management on Mn varied with month ($p=0.0218$, [Figure 32]). Mn concentrations, which did not significantly differ between the managed and unmanaged marshes in May, were significantly greater in the managed marsh in both July and September.

Concentrations of Mn in the managed marsh increased from May to July and from July to September. In the unmanaged marsh Mn concentrations, which did not significantly change from May to July, increased significantly from July to September. The concentrations of Cu, Zn, and Fe did not significantly differ with management (Cu: $p=0.2362$; Zn: $p=0.6997$; Fe: $p=0.9812$), with month (Cu: $p=0.5130$; Zn: $p=0.4990$; Fe: $p=0.5984$), or with the interaction of month x management (Cu: $p=0.5178$; Zn: $p=0.8198$; Fe: $p=0.6415$).

Extractable Soil Nutrients, Soil pH, and Bulk Density

Concentrations of extractable soil P, Ca, and Mg were significantly higher in the unmanaged marsh than in the managed marsh (P: $p=0.0124$; Ca: $p=0.0011$; Mg: $p=0.0053$) in soil cores collected in July (Table 6). There were no differences between managed and unmanaged marshes in the concentrations of extractable soil Na, K, Cu, Fe,

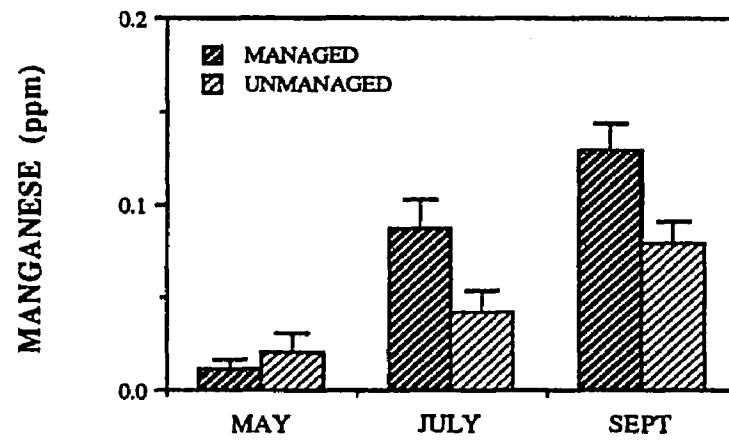


Figure 32. Interstitial manganese measured over time in a managed and an unmanaged marsh at Avoca Island, Louisiana, 1990.

Table 6. Extractable soil nutrients, pH and bulk density of soils collected at Avoca Island, Louisiana in July, 1990

	MANAGED	UNMANAGED
P	0.796 + 0.054 a	0.962 + 0.031 b
Na	0.877 + 0.116 a	0.951 + 0.077 a
K	0.721 + 0.167 a	0.589 + 0.048 a
Ca	3.849 + 0.251 a	5.000 + 0.197 b
Mg	0.868 + 0.064 a	1.095 + 0.039 b
Fe	0.179 + 0.020 a	0.183 + 0.015 a
Mn	0.135 + 0.032 a	0.087 + 0.007 a
Zn*	8.617 + 1.32 a	12.30 + 1.61 a
Cu*	1.599 + 0.08 a	1.44 + 0.02 a
pH	6.00 + 0.05 a	5.90 + 0.03a
Bulk density	0.0299 + 0.0046 a	0.0413 + 0.0048 a

* values in mg/kg; all others reported in g/kg

Bulk density in g cm⁻³

Mn, or Zn (Na: $p=0.6002$; K: $p=0.4526$; Cu: $p=0.0723$; Fe: $p=0.9120$; Mn: $p=0.1510$; and Zn: $p=0.0873$). Also, soil pH and soil bulk density did not significantly differ between the managed and unmanaged marshes (pH: $p=0.1281$; bulk density: $p=0.0982$). The bulk density values measured at Avoca Island are lower than those reported by Hatton (1981) for fresh marshes of the Louisiana delta ($0.09 \pm 0.01 \text{ g/cm}^3$ for inland marshes to $0.11 \pm 0.03 \text{ g/cm}^3$ for streamside marshes). However, Taylor et al. (1988), reported bulk densities of 0.04 to 0.10 g/cm^3 at Jean Lafitte National Park which they said were consistent with the bulk densities of fresh-intermediate marshes.

Vegetation Response

Species Composition

A total of 41 species were identified during this study. Thirty-seven of these species were collected in the managed marsh and 33 were collected in the unmanaged marsh. Species found only in the managed marsh were: *Myriophyllum spicatum*, *Eupatorium* sp., *Ptilimnium* sp., *Bacopa* sp., *Hypericum virginicum*, *Cynoctonum* sp., *Hymenocallis caroliniana*, an unidentified sedge, and a thistle. Species found in the unmanaged marsh but not the managed marsh were: *Panicum hemitomom*, *Scirpus* sp., *Setaria* sp., and *Carduus* sp. The species found only in the managed marsh or only in the unmanaged marsh were present as minor species and none of them were present on all three sample dates.

Species Dominance Values

The two species with the highest SDV's in the managed marsh in May and September were *Bidens laevis* and *Sacciolepis striata*; in July, *Bidens laevis* and *Phyla nodiflora* had the highest SDV's (Table 7 a-c). *Eleocharis* sp. (large) had the third highest SDV in the managed marsh in May, *Sacciolepis striata* had the highest SDV in July and *Phyla nodiflora* had the third highest SDV in September (Table 7 a-c). Averaged over all

Table 7. Species dominance values for the managed and unmanaged marshes in (a) May, 1990 (b) July, 1990, (c) September, 1990, and (d) averaged over all three sample dates for each marsh at Avoca Island, Louisiana.

(a)

<u>MAY</u> <u>SPECIES</u>	<u>MANAGED</u>	<u>SPECIES</u>	<u>UNMANAGED</u>
<i>Bidens laevis</i>	28.42	<i>Phyla nodiflora</i>	27.75
<i>Sacciolepis striata</i>	25.34	<i>Bidens laevis</i>	27.26
<i>Eleocharis</i> (large)	21.52	<i>Hydrocotyle ranunculoides</i>	24.04
<i>Hydrocotyle umbellata</i>	20.04	<i>Eleocharis</i> (small)	22.96
<i>Eleocharis</i> (small)	16.96	<i>Hydrocotyle umbellata</i>	21.75
Unidentified parts	16.53	Unidentified parts	15.17
<i>Hydrocotyle ranunculoides</i>	15.37	<i>Eleocharis</i> (large)	12.94
<i>Phyla nodiflora</i>	14.82	<i>Aeschynomene indica</i>	11.55
<i>Aeschynomene indica</i>	9.92	<i>Sacciolepis striata</i>	11.51
<i>Ludwigia</i> sp.	7.41	<i>Cyperus</i> sp.	4.07
<i>Solidago</i> sp.	3.47	Unidentified monocot	3.06
<i>Galium</i> sp.	3.44	<i>Galium</i> sp.	2.94
Unidentified monocot	3.04	<i>Solidago</i> sp.	2.68
<i>Sagittaria latifolia</i>	2.65	<i>Dichromena colorata</i>	2.47
<i>Myriophyllum spicatum</i>	2.39	<i>Ludwigia</i> sp.	2.33
<i>Dichromena colorata</i>	1.69	<i>Sagittaria latifolia</i>	2.23
<i>Cyperus</i> sp.	1.47	<i>Polygonum</i> sp.	1.69
<i>Ptilimnium</i> sp.	1.36	<i>Eichhornia crassipes</i>	1.63
<i>Bacopa</i> sp.	1.30	<i>Fiurena squarosa</i>	0.66
<i>Alternanthera philoxeroides</i>	1.30	<i>Scirpus</i> sp.	0.66
<i>Eupatorium</i> sp.	0.77	<i>Lymnobia</i> sp.	0.62
<i>Eichhornia crassipes</i>	0.73		

(b)

JULY			
SPECIES	MANAGED	SPECIES	UNMANAGED
<i>Bidens laevis</i>	32.28	<i>Eleocharis</i> (small)	29.42
<i>Sacciolepis striata</i>	25.87	<i>Bidens laevis</i>	27.62
<i>Phyla nodiflora</i>	27.75	<i>Phyla nodiflora</i>	24.77
<i>Eleocharis</i> (small)	19.91	<i>Aeschynomene</i>	17.63
<i>Eleocharis</i> (large)	18.44	<i>Hydrocotyle umbellata</i>	16.03
<i>Hydrocotyle umbellata</i>	15.39	<i>Sacciolepis striata</i>	13.24
<i>Ludwigia</i> sp.	12.31	<i>Eleocharis</i> (large)	10.37
Unidentified parts	10.42	Unidentified parts	9.04
<i>Aeschynomene indica</i>	9.90	<i>Ludwigia</i> sp.	8.60
<i>Dichromena colorata</i>	6.46	<i>Dichromena</i>	7.44
<i>Hydrocotyle ranunculoides</i>	6.25	Unidentified monocot	6.59
<i>Alternanthera philoxeroides</i>	4.30	<i>Hydrocotyle ranunculodes</i>	6.56
<i>Polygonum</i> sp.	3.18	<i>Sagittaria latifolia</i>	3.75
<i>Solidago</i> sp.	2.81	<i>Vigna luteola</i>	2.85
<i>Eupatorium coelestinum</i>	1.95	<i>Cyperus</i> sp.	2.67
<i>Sagittaria latifolia</i>	1.38	<i>Galium</i> sp.	1.74
<i>Cyperus</i> sp.	1.31	<i>Fuirena squarosa</i>	1.35
Unidentified monocot	1.31	<i>Cyperus oxylepis</i>	1.30
<i>Galium</i> sp.	1.28	<i>Polygonum</i> sp.	1.26
<i>Eichhornia crassipes</i>	0.80	<i>Alternanthera</i>	1.17
<i>Hymenocallis</i> sp.	0.77	<i>Setaria</i>	1.11
<i>Myriophyllum</i> sp.	0.73	<i>Solidago</i> sp.	0.86
<i>Fuirena squarosa</i>	0.71	<i>Cyperus niger</i>	0.83
<i>Vigna luteola</i>	0.67	<i>Eupatorium coelestinum</i>	0.72
		<i>Solidago sempervirens</i>	0.72
		<i>Panicum</i> sp.	0.63
		<i>Taxodium distichum</i>	0.58
		<i>Scirpus</i> sp.	0.58
		<i>Carduus</i>	0.57

(c)

<u>SEPTEMBER</u>			
<u>SPECIES</u>	<u>MANAGED</u>	<u>SPECIES</u>	<u>UNMANAGED</u>
<i>Bidens laevis</i>	46.37	<i>Bidens laevis</i>	25.72
<i>Sacciolepis striata</i>	39.02	<i>Eleocharis</i> (small)	24.33
<i>Phyla nodiflora</i>	15.04	<i>Phyla nodiflora</i>	21.55
<i>Ludwigia</i> sp.	14.84	<i>Aeschynomene</i>	21.50
<i>Eleocharis</i> (small)	14.20	<i>Ludwigia</i> sp.	20.90
Unidentified parts	11.07	Unidentified parts	15.88
<i>Hydrocotyle umbellata</i>	10.75	<i>Sacciolepis striata</i>	14.74
<i>Aeschynomene indica</i>	8.10	<i>Hydrocotyle umbellata</i>	12.19
<i>Eleocharis</i> (large)	6.36	<i>Eleocharis</i> (large)	7.53
<i>Fuirena squarosa</i>	4.11	<i>Sagittaria latifoli</i>	5.73
<i>Eichhornia crassipes</i>	2.73	<i>Carex</i> sp.	5.36
<i>Alternanthera philoxeroides</i>	2.17	<i>Vigna luteola</i>	4.12
<i>Hydrocotyle ranunculoides</i>	2.10	<i>Fuirena squarosa</i>	4.00
<i>Eupatorium coelestinum</i>	1.86	<i>Cyperus</i> sp.	3.84
<i>Solidago</i> sp.	1.82	<i>Solidago</i> sp.	2.67
<i>Cynoctonum</i>	1.79	<i>Dichromena colorata</i>	2.26
<i>Carex</i> sp.	1.68	<i>Hydrocotyle ranunculoides</i>	1.95
<i>Cyperus</i> sp.	1.45	Unidentified monocot	1.34
Unidentified monocot	1.42	<i>Eichhornia crassipes</i>	1.28
<i>Salvinia</i>	1.40	<i>Lymnobia</i>	1.13
<i>Hypericum</i>	1.19	<i>Salvinia</i>	0.68
<i>Lymnobia</i>	1.07	<i>Polygonum</i>	0.65
<i>Thistle</i>	1.02	<i>Galium</i> sp.	0.63
<i>Cyperus oxylepis</i>	0.94		
<i>Dichromena colorata</i>	0.87		
<i>Solidago sempervirens</i>	0.84		
<i>Bacopa</i>	0.79		
<i>Vigna luteola</i>	0.75		
<i>Sagittaria latifolia</i>	0.74		
<i>Cyperus niger</i>	0.71		
<i>Sedge</i>	0.71		
<i>Taxodium distichum</i>	0.70		
<i>Galium</i> sp.	0.70		
<i>Polygonum</i> sp.	0.70		

(d)

OVERALL SPECIES	MANAGED	SPECIES	UNMANAGED
<i>Bidens laevis</i>	39.19	<i>Bidens laevis</i>	26.69
<i>Sacciolepis striata</i>	33.03	<i>Eleocharis</i> (small)	25.66
<i>Phyla nodiflora</i>	16.73	<i>Phyla nodiflora</i>	23.42
<i>Eleocharis</i> (small)	16.03	<i>Aeschynomene</i>	19.23
<i>Hydrocotyle umbellata</i>	13.48	<i>Hydrocotyle umbellata</i>	14.78
<i>Eleocharis</i> (large)	12.70	Unidentified parts	14.22
Unidentified parts	12.65	<i>Sacciolepis striata</i>	13.53
<i>Ludwigia leptocarpa</i>	12.22	<i>Ludwigia leptocarpa</i>	13.50
<i>Aeschynomene indica</i>	9.64	<i>Eleocharis</i> (large)	8.92
<i>Hydrocotyle ranunculoides</i>	6.56	<i>Hydrocotyle ranunculoides</i>	8.25
<i>Dichromena colorata</i>	2.85	<i>Sagittaria latifolia</i>	4.07
<i>Solidago</i> sp.	2.60	<i>Dichromena colorata</i>	3.91
<i>Alternanthera philoxeroides</i>	2.59	Unidentified monocot	3.49
Unidentified monocot	1.84	<i>Cyperus</i> sp.	3.34
<i>Fuirena squarosa</i>	1.70	<i>Vigna luteola</i>	2.63
<i>Galium</i> sp.	1.66	<i>Carex</i> sp.	2.53
<i>Sagittaria latifolia</i>	1.59	<i>Solidago</i> sp.	2.19
<i>Eichhornia crassipes</i>	1.53	<i>Fuirena squarosa</i>	1.99
<i>Eupatorium coelestinum</i>	1.50	<i>Galium</i> sp.	1.69
<i>Cyperus</i> sp.	1.38	<i>Polygonum</i>	1.12
<i>Polygonum</i> sp.	1.28	<i>Eichhornia crassipes</i>	0.87
<i>Myriophyllum</i>	0.73	<i>Lymnobia</i>	0.68
<i>Bacopa</i>	0.72	<i>Cyperus oxylepis</i>	0.45
<i>Cynoctonum</i>	0.66	<i>Scirpus</i> sp.	0.41
<i>Carex</i> sp.	0.60	<i>Alternanthera</i>	0.41
<i>Hypericum</i>	0.50	<i>Setaria</i>	0.36
<i>Vigna luteola</i>	0.47	<i>Cyperus niger</i>	0.28
<i>Salvinia</i>	0.45	<i>Eupatorium coelestinum</i>	0.25
<i>Pullinimum</i>	0.45	<i>Solidago sempervirens</i>	0.25
<i>Lymnobia</i>	0.43	<i>Salvinia</i>	0.23
<i>Thistle</i>	0.41	<i>Panicum</i>	0.22
<i>Cyperus oxylepis</i>	0.36	<i>Taxodium distichum</i>	0.21
<i>Solidago sempervirens</i>	0.30	<i>Carduus</i>	0.20
<i>Hymenocallis</i>	0.26		
<i>Eupatorium</i> sp.	0.24		
<i>Cyperus niger</i>	0.23		
<i>Sedge</i>	0.23		
<i>Taxodium distichum</i>	0.23		

sample dates, the three species with the highest SDV's in the managed marsh were: *Bidens laevis*, *Sacciolepis striata*, and *Phyla nodiflora* (Table 7d). In the unmanaged marsh, the three species with the highest SDV's in May were *Phyla nodiflora*, *Bidens laevis*, and *Hydrocotyle ranunculoides*; in July, the three species were *Eleocharis* sp. (small), *Bidens laevis*, and *Phyla nodiflora*; and in September the three species were *Bidens laevis*, *Eleocharis* sp. (small), and *Phyla nodiflora* (Table 7a-c). In the unmanaged marsh the species with the three highest SDV's were *Bidens laevis*, *Eleocharis* sp. (small), and *Phyla nodiflora* (Table 7d).

Standing Crop and Primary Productivity

Live, dead, and total biomass all increased with time (live: $p=0.0001$; dead: $p=0.0001$; total: $p=0.0001$ [Figure 33 a-c]) but there were no significant differences attributable to management (live: $p=0.2323$; dead: $p=0.6457$; total: $p=0.3128$). Live and total biomass increased significantly in July and in September, while dead biomass was significantly greater only in September. The interaction of month with management was not significant (live: $p=0.8114$; dead: $p=0.2032$; total: $p=0.9598$). During visits to the managed and unmanaged marshes at Avoca Island in April we found little or no emergent vegetation growth. Emergent plant growth in these marshes exhibited an increasing growth curve throughout the summer in contrast to the oscillating growth curves observed with species such as *Spartina patens* or *Spartina alterniflora* (Gosselink, 1984).

Net primary productivity was calculated using the Smalley method (which accounts for plant mortality between sampling intervals), the Milner and Hughes method (which does not account for plant mortality between sampling intervals) and Peak live + dead biomass (Table 8). None of these methods accounts for herbivory or decomposition and are therefore underestimates of productivity. No significant differences in net

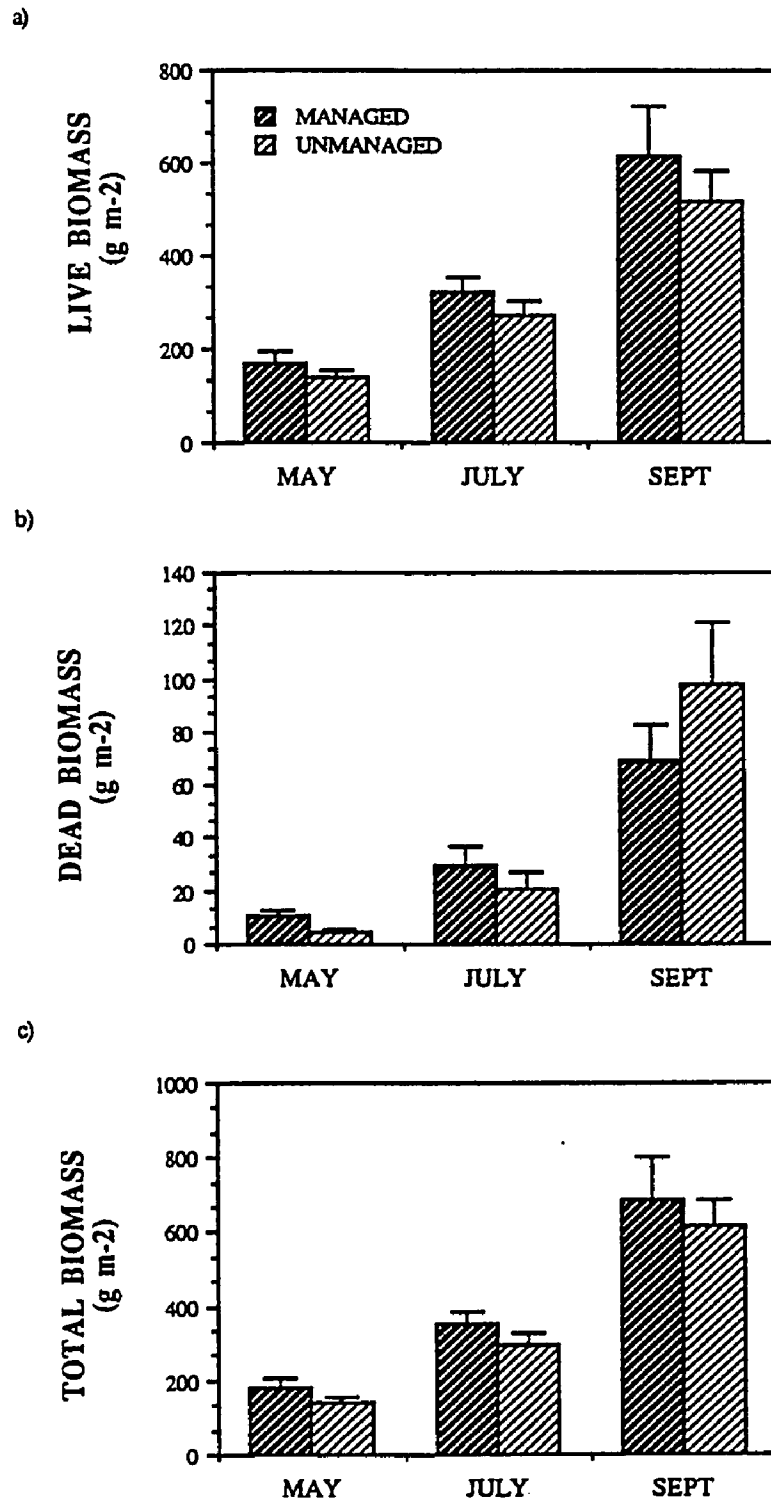


Figure 33. (a) Live, (b) dead, and (c) total emergent plant biomass measured over time at Avoca Island, Louisiana, 1990.

TABLE 8. Estimates of net primary productivity (g m^{-2}) for managed and unmanaged marshes at Avoca Island, Louisiana in 1990.

SITE	N	PEAK L + D	MILNER-HUGHES	SMALLEY
Managed	16	707.2 ± 112.3^a	635.9 ± 103.8^a	714.8 ± 113.0^a
Unmanaged	16	621.4 ± 73.0^a	533.5 ± 65.8^a	622.0 ± 73.1^a

primary productivity were detected between the managed and unmanaged marshes with any of these methods (Smalley: $p=0.5290$; Milner and Hughes: $p=0.5603$; Peak L + D: $p=0.5234$).

CO₂ Assimilation

The effect of management on CO₂ exchange rate varied with species (treatment x species interaction significant: $p=0.0374$ [Figure 34a]). When averaged over the three sample dates, *Sacciolepis* exhibited a greater CO₂ exchange rate in the managed marsh than in the unmanaged marsh (Figure 34a), although this difference was small (8%). There was no significant difference in the CO₂ exchange rate of *Hydrocotyle* between the managed and the unmanaged marshes (Figure 34a).

The two species differed in their CO₂ exchange rates with month (month x species interaction significant: $p=0.0001$ [Figure 34b]). In *Hydrocotyle*, the CO₂ exchange rate decreased from May to July and again in September (Figure 34b). In *Sacciolepis*, the CO₂ exchange rate increased significantly from May to July and then decreased to a level significantly lower than May or July in September (Figure 34b).

Although both *Sacciolepis striata* and *Hydrocotyle umbellata* exhibited a seasonal decline in CO₂ exchange rates, the seasonal patterns were different. CO₂ exchange rates (Figure 34b) and SDV's (Table 7 a-c) of *Hydrocotyle umbellata* were greatest in May and decreased on each sample date thereafter. CO₂ exchange rates of *Sacciolepis striata* increased from May to July (Figure 34b) but the SDV's did not appreciably change in either marsh until September (Table 7a-c). In September, the SDV's of *S. striata* were at the highest point for both the managed and unmanaged marshes even though the CO₂ exchange rates were lower than at any other time in the study. Seasonal declines in CO₂ exchange rates have also been reported for *Spartina patens* and *Distichlis spicata* (Turitzan

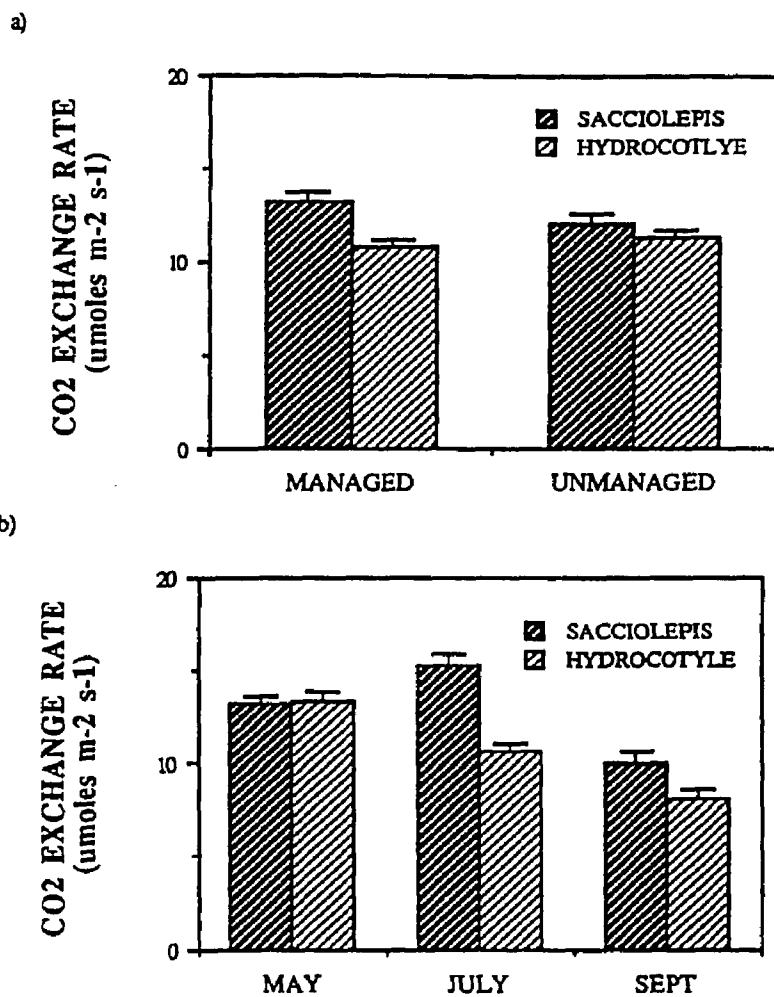


Figure 34. Leaf CO₂ exchange rates of *Sacciolepis striata* and *Hydrocotyle umbellata* (a) in a managed and unmanaged marsh averaged over all sample dates, and (b) in a managed and unmanaged marsh over time at Avoca Island, Louisiana, 1990.

and Drake, 1981) in a Chesapeake Bay saline marsh. De Jong et al. (1982) also reported a seasonal decline in the CO₂ exchange rates of *S. patens*, *Scirpus olneyi*, and *D. spicata*. They reported that this decline was strongest in *S. patens*.

Diel Characterization of CO₂ Exchange Rates

Both *S. striata* and *H. umbellata* exhibited a bi-modal pattern of CO₂ fixation (Figures 35a and 36a). However, when PPFD is compared to the CO₂ fixation rate of each species (Figures 35b and 36b), it is evident that the decrease in CO₂ fixation measured at approximately noon was due to the lower light levels which occurred at that time. A similar diel pattern for CO₂ exchange rate (without the decrease we observed at noon) was described by Pezeshki et al. (1987 a and 1987b) in *Spartina patens*, *Spartina alterniflora*, and *Panicum hemitomon* under controlled conditions in a growth chamber.

SUMMARY

Although the use of water control structures at Avoca Island did not result in significant differences in water depth, i. e., the distance from the floating vegetation mat to the underlying bottom, soil conditions at the 15 cm depth were less reduced in the managed marsh than in the unmanaged marsh. In addition, Eh at the soil surface, while not being statistically different, also tended to be higher in the managed marsh compared to the unmanaged marsh. Thus, these results suggest that the managed marsh soil may be less reduced than the unmanaged marsh. The reason for this apparent difference is as yet unclear since water depth, at least when measured, was not affected by management. While there was a statistically significant difference in conductivity, with the managed marsh having lower conductivity, this difference of 0.04 $\mu\text{hos/cm}$ is probably not

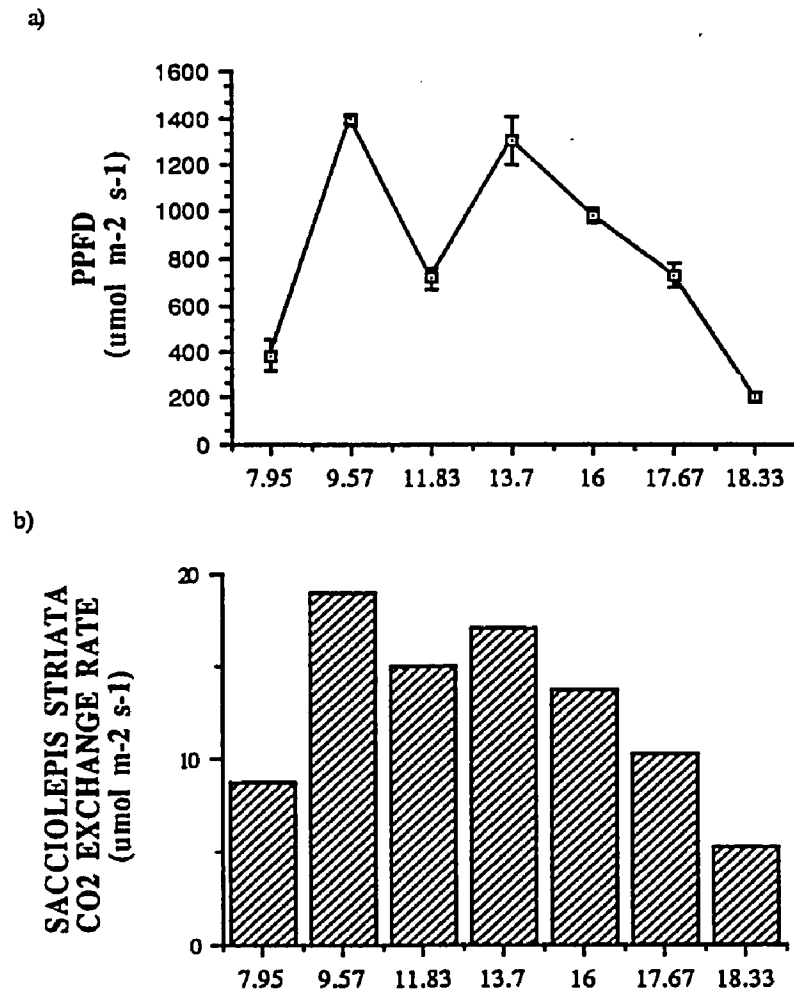


Figure 35. (a) Light levels and (b) diel leaf CO₂ exchange rates of *Sacciolepis striata* August 17, 1990 at Avoca Island, Louisiana.

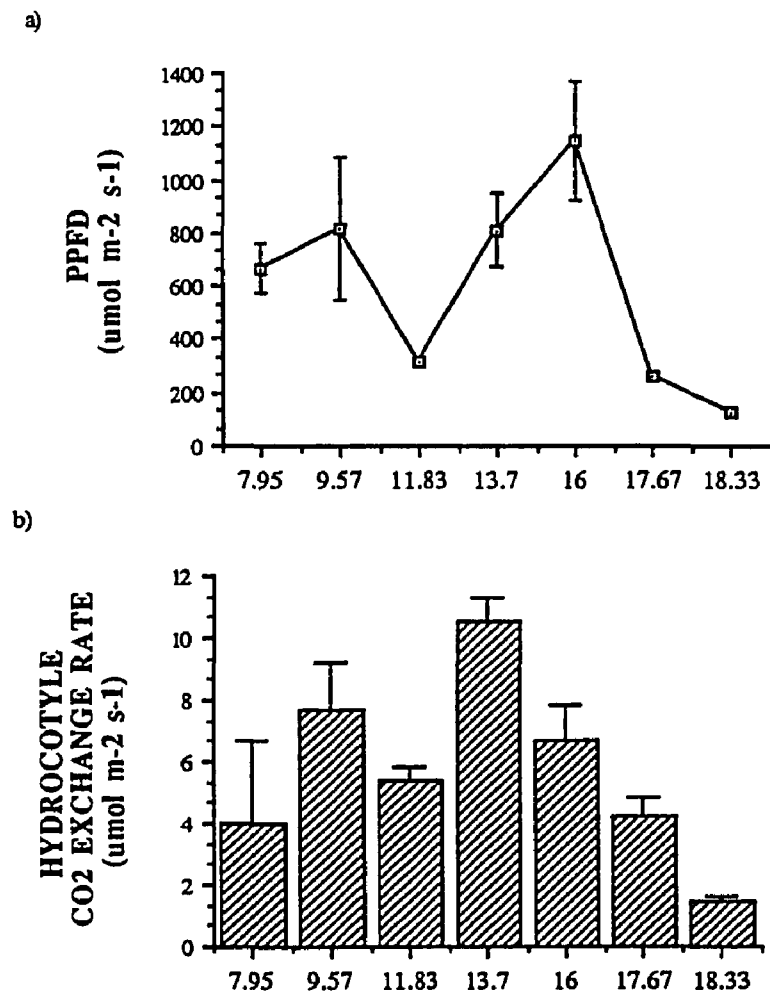


Figure 36. (a) Light levels and (b) diel leaf CO₂ exchange rates of *Hydrocotyle umbellata* on August 17, 1990 at Avoca Island, Louisiana.

biologically significant. There were no significant differences in pH, $\text{NH}_4\text{-N}$, or P between the managed or unmanaged marshes, although $\text{NH}_4\text{-N}$ and P tended to be higher in the managed marsh. The presence of lower concentrations of Ca and Mg in the managed marsh were the only significant differences in macronutrients associated with management. Concentrations of the micronutrient, Mn, were significantly greater in the managed marsh. Soil pH and bulk density did not exhibit any significant differences between the managed and unmanaged marshes. Of the extractable soil nutrients, concentrations of P, Ca, and Mg were all significantly higher in the unmanaged marsh. There were no significant differences in concentrations of Na, K, Fe, Mn, Zn, and Cu.

Live, dead, and total biomasses and primary productivity (regardless of the method of estimation used) did not significantly differ between the managed and unmanaged marshes. However, this marsh has only been managed since 1987 and differences may require a longer period to manifest themselves. Although the CO_2 exchange rate of *Hydrocotyle umbellata* did not significantly differ between the managed and unmanaged marshes, the CO_2 exchange rate of *Sacciolepis striata* was significantly greater in the managed marsh but the difference is small (8 %). Both *S. striata* and *H. umbellata* exhibited a seasonal decline in CO_2 exchange rates. Species dominance values in the managed and unmanaged marshes differed somewhat, but it is not clear that these differences were due to management.

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CHAPTER 5

THE ROLE OF SALINITY AND WATER LEVEL IN CONTROLLING FRESHWATER MARSH RECOVERY FROM SALTWATER INTRUSION

INTRODUCTION

Saltwater intrusion is often cited as one cause of wetland loss or deterioration in coastal Louisiana. In 1977, Morgan predicted that a gradual encroachment of saline waters would occur as the deltaic plain subsides and sea level rises. This theory was supported by the work of Chabreck and Linscombe (1982) who reported the northward movement of more saline marsh types during the period from 1968 to 1978. Wang (1987) noted that straight-line canals constructed in coastal wetland areas can accelerate this movement of saltwater into brackish and freshwater marshes.

Water level or structural marsh management (involving the use of levees, weirs, flap-gates and other structures) is being used in many areas of southern Louisiana to address the problems associated with sea level rise and saltwater intrusion. However, associated with this use of structural marsh management is the potential problem posed by the occurrence of frontal passages and hurricanes which can deposit large volumes of saline water inside management areas. The time required for storm-deposited waters to drain from managed marshes is greater than that required under natural conditions because of the barrier posed by the levees used to establish management areas (Meeder, 1987).

The response of freshwater marsh communities to saltwater intrusion may not be uniform and may, in fact, depend on a variety of factors including the level, duration, and abruptness of exposure, initial species composition of the marsh and the depth of flooding (Mendelssohn and McKee, 1988; McKee and Mendelssohn, 1989). The damage resulting from the flooding of impoundments by storm surges has been reported

previously and includes longer periods of marsh inundation (Meeder, 1987), decreased wildlife food in the following growing season (Ensminger and Nichols, 1957), and loss of all vegetation (Shiflet, 1963). However, the factors controlling the recovery of impounded marshes after a saltwater intrusion event have not been investigated. The present study was designed to determine whether the salinity and water levels that occur during the period following dieback of the original freshwater marsh vegetation are major determinants of the degree of recovery a freshwater marsh will experience after a saltwater intrusion event.

MATERIALS AND METHODS

Experimental Design

Forty sods were collected from a freshwater marsh located near Lake des Allemands in southern Louisiana (29° 52' 30") on April 20, 1990. These sods of intact vegetation and soil were placed in buckets having dimensions of 28.3 cm (diameter) by 35.6 cm (depth). The sods with their intact vegetation were transported to a greenhouse on the Louisiana State University campus where they were established as mesocosms.

The mesocosms were maintained in a moist, but well-drained condition in the greenhouse until May 14, 1990. At that time, 30 of the mesocosms were subjected to a simulated saltwater intrusion event during which water levels were raised to approximately 10 cm above the soil surface and interstitial water salinity was increased to 15 ppt with Instant Ocean (Aquarium Systems, Inc., Mentor, OH, USA). This saltwater intrusion event lasted for 6 weeks (from May 14-July 4, 1990) and resulted in the dieback of virtually all of the aboveground biomass.

The 10 remaining mesocosms were maintained as controls. Five of these were randomly chosen as undisturbed controls and five as disturbed controls. Double-thick brown plastic bags were placed over vegetation in the disturbed controls and these bags

remained in place until the aboveground vegetation was dead. The bags were removed on July 4, 1990 and the aboveground vegetation was allowed to recover. The 5 undisturbed controls were not subjected to any killing treatment. Water levels in both the undisturbed and disturbed controls were maintained 5 cm below the soil surface throughout the experiment.

After dieback of the original vegetation, all 40 of the mesocosms were flushed with fresh water until salinity of the interstitial water returned to approximately 1-2 ppt. Flushing with freshwater occurred during the period of July 4- July 30, 1990. The mesocosms were then randomly assigned to a recovery-phase salinity/water level treatment combination (3X2 factorial arrangement). Three recovery-phase salinity levels (low [1-2 ppt], medium[6 ppt], and high [9-12 ppt]) in combination with two different water levels (5 cm below and 10 cm above the soil surface, hereafter referred to as drained and flooded, respectively) were established with five mesocosms assigned to each treatment combination. Salinity treatments were begun on July 30, 1990 and the target salinity levels were reached by September 14, 1990. The treatment and control mesocosms were maintained until June 1, 1991 so that long-term responses to recovery-phase conditions could be identified. Figure 37 schematically depicts the experimental protocol.

Vegetation Parameters

Emergent plant species were identified in order to determine the species richness in each treatment combination. All 40 of the mesocosms contained *Panicum hemitomon*, *Sagittaria lancifolia*, and *Leersia orizoides* prior to the simulated saltwater intrusion event. Species richness was determined in October, 1990, March, 1991 and when the experiment was harvested June 1, 1991.

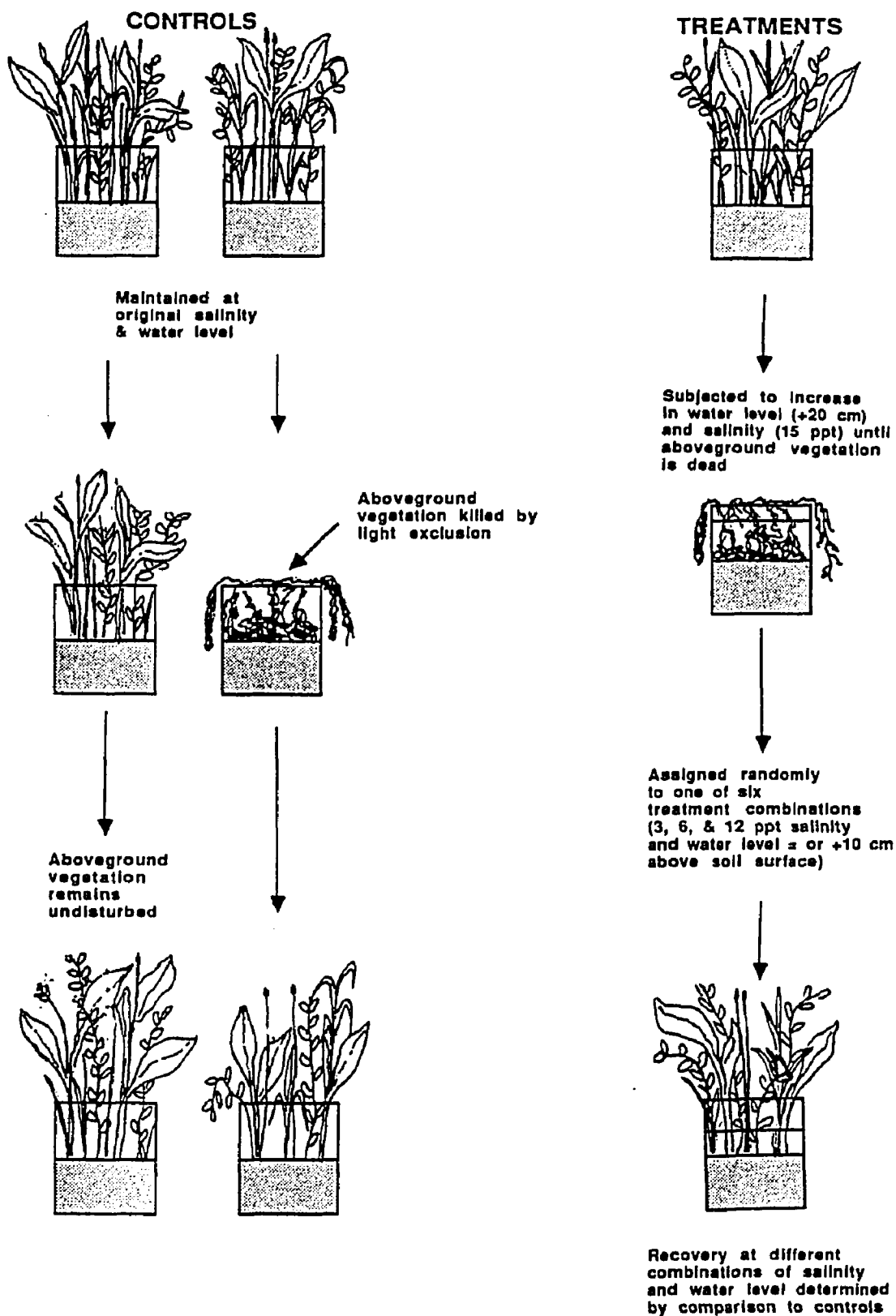


Figure 37. Schematic of the experimental protocol employed in the simulated saltwater intrusion event and subsequent recovery of the freshwater marsh mesocosms.

At the termination of the experiment, aboveground biomass was harvested, sorted according to species, dried at 65° C, and weighed to the nearest 0.1g. Soil was washed from the belowground biomass of each mesocosm, the root and rhizome material was then dried at 65° C and weighed to the nearest 0.1g. No attempt was made to separate root tissue into live or dead biomass. Belowground biomass was expressed on a per volume basis (g cm^{-3}) rather than a per pot basis since sod volume varied slightly among pots.

Decomposition rates were measured by comparing the loss of weight of balsam wood strips inserted into the soil of each mesocosm. Balsam strips were placed in each sod at depths of 5 and 13 cm. The preweighed balsam strips were placed in the soil on December 8, 1990 and removed at the end of the experiment. After removal, the labeled balsam strips were placed in deionized water and soaked at 4° C. The strips were then placed in fresh deionized water and sonicated for two hours using a Branson 3200 ultrasonic bath. The strips were rinsed and sonicated for another hour after which they were dried to a constant weight at 65° C and weighed to the nearest 0.0001g. Sonication was used to remove any soil particles adhering to the balsam strips.

Soil Parameters

Eh (redox potential) readings were made in the upper 1-2 cm of soil (hereafter referred to as surface Eh) and at a depth of 15 cm in each mesocosm on a monthly basis (with the exception of February). Measurements were made using a calomel reference electrode and a portable digital pH-mV meter. In order to base the readings on a standard hydrogen electrode, 244 mV was added to each reading (Faulkner et al., 1989). Soils were classified as aerated (>300 mV), moderately reduced (100 to 300 mV), reduced (-100 to 100 mV), and highly reduced (<-100 mV, [Patrick, 1980]). Eh readings were not corrected for pH since the average pH in the various treatments only ranged from a low of 6.28 to a high of 6.41.

Interstitial Water Analysis

Interstitial water samples were withdrawn from the soil in each mesocosm twice during the recovery phase (January and May, 1991) with the method of McKee et al., 1988. Salinity samples were collected at least monthly and pH and sulfide samples were collected monthly (with the exception of February).

Salinity of the water samples (corrected to 25° C) was measured using a Fisher Conductivity Meter Model 152 and pH was measured using an Altex Model 3560 Digital pH Meter with a Corning General Purpose Combination Electrode. The water was then filtered using a 0.45 micron millipore filter after which an aliquot was removed and frozen for NH₄-N analysis. Ammonium-N was analyzed at the University of Georgia Institute of Ecology using EPA method 350.1 (phenate) (U. S. EPA, 1979). The remainder of the filtered water was acidified with concentrated nitric acid and analyzed for essential nutrients (P, K, Ca, Mg, Na, Fe, Mn, Cu, and Zn) using a Fisher inductively coupled plasma argon emission spectrometer (ICP, Atom Comp Series 800). Immediately upon collection of the interstitial water, an aliquot was added to an antioxidant buffer solution (NaOH, ascorbic acid, sodium salicylate) for sulfide determination using a Lazar ISM-146 Micro Ion sensing electrode and a portable Cole Parmer Model 5985-80 Digi-Sense mV meter (Lazar, 1986).

Statistical Analysis.

SAS software was used to conduct statistical analysis (SAS, 1985). The effect of treatment on soil and plant variables were analyzed with analysis of variance. The repeated measures technique was used to analyze all variables except live aboveground and total belowground (live and dead) biomass which were measured only at the time the experiment was harvested. When the overall treatment effect between subjects was significant, single degree of freedom contrasts were used to compute comparisons between treatments. The effects of time and the time x treatment interaction were

determined on all variables except live and total belowground biomass which were measured only at the time the experiment was harvested. When the time x treatment interaction was significant, single degree of freedom contrasts were used to compute comparisons between treatments over time. All hypotheses were tested at the 0.05 probability level unless otherwise indicated. Unless noted, there were no significant differences between the disturbed and undisturbed controls and data from these two control groups were combined. The controls were included in all comparisons except when the flooded mesocosms were compared to the drained mesocosms.

RESULTS

Vegetation Variables

Species Richness

Averaged over time and the low, medium, and high salinity treatments, flooding resulted in significantly lower species richness ($3.7 \pm .5$) than in the drained ($5.5 \pm .4$) treatments ($p=0.0001$). As salinity increased, a significant linear decrease in species richness occurred ($p=0.0001$), the rate being significantly greater in the drained treatments compared to the flooded ($p=0.0263$, Figure 38a).

Although species richness decreased from October, 1990 to March, 1991 and then increased in May, 1991 in the control mesocosms, these changes were relatively minor (Figure 38b). This pattern reflects the occurrence of seasonal changes in species composition which normally occur in freshwater marshes. Species richness also changed with time in the low, medium, and high salinity treatments, but did not exhibit a seasonal trend (time x salinity effect: $p=0.0084$).

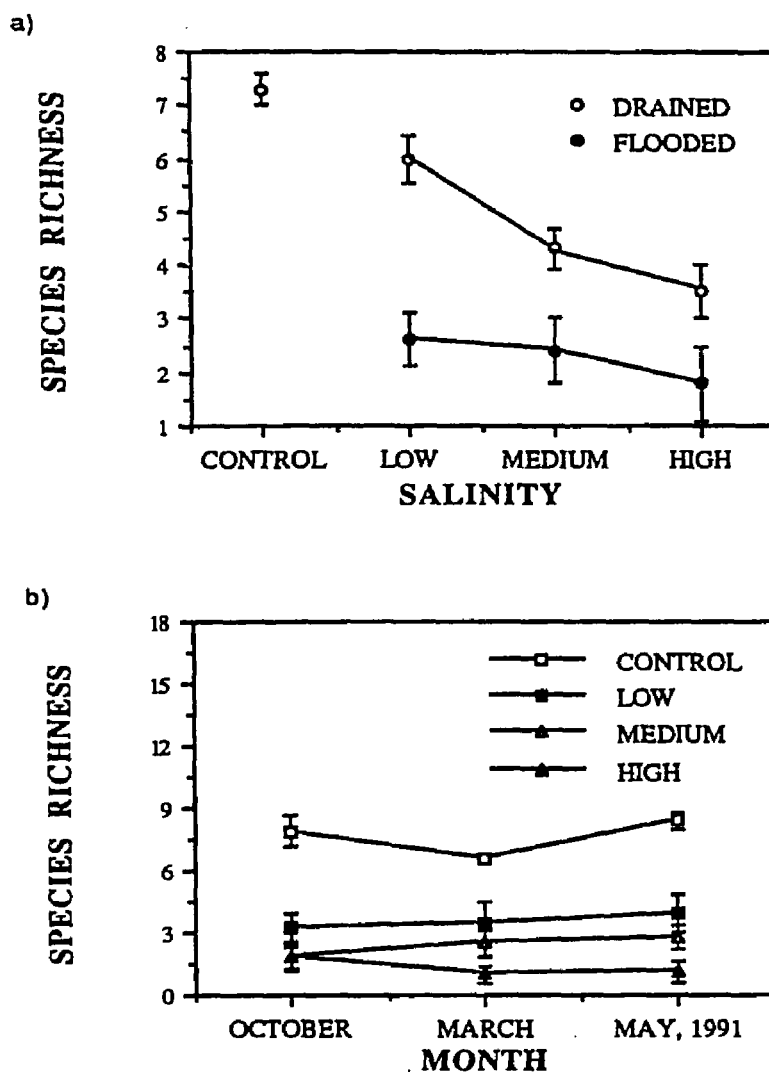


Figure 38. Species richness measured over salinity (a) by water level, and (b) by sample date.

Prior to the simulated saltwater intrusion event there were three species that occurred in all 40 of the mesocosms: *Sagittaria lancifolia*, *Leersia oryzoides*, and *Panicum hemitomon*. In addition, a variety of other freshwater marsh species were identified (Table 9).

The simulated salt-water intrusion event resulted in the complete elimination of *P. hemitomon* from the treatment mesocosms regardless of recovery-phase conditions, but *P. hemitomon* was present in 8 of the 10 controls at the time of harvest. Thus it appears that the seeds and rhizomes of *P. hemitomon* were unable to survive conditions present during the simulated saltwater intrusion event.

Leersia oryzoides was present in all of the undisturbed controls at harvest. However, the killing treatment imposed on the disturbed controls completely eliminated this species. In addition, *L. oryzoides* was present in only 5 of the 30 treatment mesocosms at harvest. This means that there was a 75% overall reduction in the frequency of *L. oryzoides* (50% decrease in frequency in the controls and 83.3% decrease in occurrence in the treatment mesocosms). Regrowth of *L. oryzoides* occurred in one of the drained low salinity mesocosms, two of the flooded low salinity mesocosms, and two of the flooded medium salinity mesocosms (Table 10).

S. lancifolia, which occurred with 100% frequency prior to the simulated saltwater intrusion event was found in only 11 of the 30 treatment mesocosms (36.7%) but was present in all 10 (100%) of the controls when the experiment was harvested. Thus the saltwater intrusion event and subsequent recovery phase conditions resulted in a 47.5% overall decrease in the frequency of *S. lancifolia* (Table 10).

Other researchers have shown that increases in salinity result in decreased growth and/or death of *P. hemitomon*, *L. oryzoides*, and *S. lancifolia*. McKee and Mendelssohn (1989) conducted a field study in which intact sods of freshwater marsh vegetation containing these three species were transplanted into a higher salinity marsh in which salinity subsequently increased to 15 ppt resulting in the death of these three species. A

Table 9. Species composition changes that occurred from the period prior to the simulated saltwater intrusion event through the killing treatment and the subsequent recovery phase.

<u>TREATMENT</u>	<u>SPECIES LOST</u>	<u>SPECIES GAINED</u>
<u>CONTROLS</u>		
<u>Undisturbed</u>	None	<i>Ludwigia leptocarpa</i> <i>Hypericum</i> sp. <i>Hydrocotyle</i> sp. <i>Saururus cernuus</i> Unidentified grass
<u>Disturbed</u>	<i>Leersia oryzoides</i>	<i>Cyperus</i> sp. <i>Solidago</i> sp. <i>Ludwigia leptocarpa</i> <i>Alternanthera</i>
<i>philoxeroides</i>		<i>Sacciolepis striata</i> <i>Aeschynomene</i> sp. <i>Thelypteris palustris</i> Thistle sp. 2 Unidentified
<u>LOW SALINITY</u>		
<u>Drained</u>	<i>Panicum hemitomon</i> <i>Taxodium distichum</i> <i>Asclepias lanceolata</i>	<i>Sacciolepis striata</i> <i>Ludwigia leptocarpa</i> <i>Saururus cernuus</i> <i>Vigna luteola</i> <i>Cyperus</i> sp.
<u>Flooded</u>	<i>Panicum hemitomon</i> <i>Ptilimnium capillaceum</i> <i>Polygonum</i> sp. <i>Taxodium distichum</i> <i>Galium</i> sp.	None

(Table 9 cont.)

<u>TREATMENT</u>	<u>SPECIES LOST</u>	<u>SPECIES GAINED</u>
<u>MEDIUM SALINITY</u>		
<u>Drained</u> <i>capillaceum</i>	<i>Panicum hemitomom</i> <i>Leersia oryzoides</i> <i>Aeschynomene</i> sp.	<i>Ptilimnium</i> <i>Sacciolepis striata</i> <i>Eclipta alba</i>
<u>Flooded</u> <i>capillaceum</i> <i>philoxeroides</i> (trace)	<i>Panicum hemitomom</i> <i>Taxodium distichum</i> <i>Polygonum</i> sp. <i>Aeschynomene</i> sp. <i>Cyperus</i> sp. <i>Vigna luteola</i> <i>Galium</i> sp. <i>Ipomoea sagittata</i>	<i>Ptilimnium</i> <i>Eclipta alba</i> <i>Alternanthera</i>
<u>HIGH SALINITY</u>		
<u>Drained</u>	<i>Panicum hemitomom</i> <i>Leersia oryzoides</i> <i>Polygonum</i> sp. <i>Aeshynomene</i> sp. <i>Taxodium distichum</i> <i>Galium</i> sp.	<i>Eclipta alba</i> <i>Sacciolepis striata</i>
<u>Flooded</u>	<i>Sagittaria lancifolia</i> <i>Panicum hemitomom</i> <i>Leersia oryzoides</i> <i>Taxodium distichum</i> <i>Polygonum</i> sp. <i>Hydrocotyle</i> sp. <i>Eleocharis</i> sp. <i>Aeschynomene</i> sp. <i>Cyperus</i> sp. <i>Ptilimnium capillaceum</i> <i>Galium</i> sp. 1 Unidentified	None

Table 10. Species identified prior to the killing treatment and at the time of harvest and the frequency of occurrence (%) for each species (100%=present in all mesocosms).

SPECIES LIST	INITIAL %	FINAL %
<i>Sagittaria lancifolia</i>	100%	47.5%
<i>Panicum hemitomon</i>	100%	20%
<i>Leersia oryzoides</i>	100%	25%
<i>Polygonum</i> sp.	82.5%	17.5%
<i>Eleocharis</i> sp.	75%	42.5%
<i>Taxodium distichum</i>	42.5%	10%
<i>Hydrocotyle</i> spp.	40%	37.5%
<i>Ptilimnium capillaceum</i>	20%	32.5%
<i>Galium</i> sp.	17.5%	17.5%
<i>Alternanthera philoxeroides</i>	15%	12.5%
<i>Aeschynomene</i> sp.	15%	10%
<i>Ipomoea sagittata</i>	5%	2.5%
<i>Cyperus</i> sp.	5%	15%
<i>Asclepias lanceolata</i>	2.5%	0
<i>Vigna luteola</i>	2.5%	2.5%
<i>Ludwigia leptocarpa</i>	0	22.5%
<i>Sacciolepis striata</i>	0	20%
<i>Eclipta alba</i>	0	10%
<i>Hypericum</i> sp.	0	10%
<i>Sarurus cernuus</i>	0	10%
Unidentified Thistle	0	5%
<i>Thelypteris palustris</i>	0	2.5%

**SPECIES PRESENT DURING THE EXPERIMENT,
BUT NOT AT THE TIME OF HARVEST:**

Aster sp.
Solidago sp.
 Unidentified sedge

subsequent decrease in salinity to 12 ppt did not result in regrowth of *P. hemitomon*, *S. lancifolia*, or *L. oryzoides*. In a subsequent greenhouse experiment, they found *S. lancifolia* to be the most sensitive of the three species to increases in salinity--showing evidence of tissue damage at salinities as low as 4.8 ppt. Both *P. hemitomon* and *L. oryzoides* had reduced biomass when salinities were raised, but were relatively tolerant of salinities up to 9.4 ppt for at least one month. Pezeshki et al. (1987) reported that *P. hemitomon* exhibited reduced growth when exposed to elevated salinities for even relatively short periods of time. They found that within 5 days of experiencing a salinity increase to 10-12 ppt *P. hemitomon* exhibited tissue damage and death. Even at lower salinities (5-7 ppt) the stomatal conductance and photosynthesis of *P. hemitomon* was reduced after only one week.

The current investigation is unique in that the emphasis is not on the effect of elevated salinities, but on the ability of the freshwater marsh species to recover from lethal levels of salt, at least to aboveground tissue, in conjunction with either drained or flooded conditions. I found that although the saltwater intrusion event resulted in a decrease in the frequency of *S. lancifolia* during the recovery phase, this species did have the ability to recover from the saltwater intrusion event. Regrowth of *S. lancifolia* was dependent upon development of favorable conditions subsequent to the killing treatment. It is important to note that recovery-phase conditions in the flooded high salinity treatment resulted in the loss of all species including *P. hemitomon*, *L. oryzoides*, and *S. lancifolia*. It is possible that the large tubers of *S. lancifolia* were more resistant to the flooding and saline waters present during the killing treatment than where the roots and rhizomes of *P. hemitomon* and *L. oryzoides*. This factor may have allowed *S. lancifolia* to regrow in all but the flooded high salinity treatment.

Several freshwater marsh species which were not present in any of the 40 mesocosms prior to the simulated saltwater intrusion event were found growing in at least one of the mesocosms during the recovery phase or at the time of harvest. These species

were: *Ludwigia leptocarpa*, *Saururus cernuus*, *Sacciolepis striata*, *Eclipta alba*, *Hypericum* sp., an unidentified thistle, and *Thelypteris palustris*. In addition, regrowth of species present prior to the simulated saltwater intrusion event occurred during the recovery phase (Table 10). The presence of these species indicates that the seeds, tubers, and/or rhizomes present in the soil were able to withstand the elevated salinity and submerged conditions present during the six week long saltwater intrusion event.

Live Aboveground Biomass

A significant linear decrease in live aboveground biomass occurred as salinity increased ($p=0.0001$, Figure 39). When compared to the controls, live aboveground biomass decreased by 30.6% in the low salinity treatment, 46.7% in the medium salinity treatment, and by 90.4% in the high salinity treatment. Even when low post-die-back salinity levels were achieved, long-term recovery of the freshwater marsh vegetation was noticeably decreased as compared to vegetation in the control mesocosms. Although differences in live aboveground biomass associated with flooding were not statistically significant at the 0.05 probability level, live aboveground biomass tended to be lower in the flooded treatments ($p=0.0874$).

Belowground Biomass

Recovery-phase treatments had a significant effect on belowground biomass ($p=0.0326$) such that, averaged over the salinity treatments, the flooded mesocosms ($.0185 \text{ g cm}^{-3} \pm .0007$) had greater belowground biomass than the drained ($.0153 \text{ g cm}^{-3} \pm .0007$) treatments (flooding: $p=0.0044$). In addition, it is interesting to note that belowground biomass was significantly lower in the disturbed ($.0162 \text{ g cm}^{-3} \pm .0006$) control than in the undisturbed ($.0206 \text{ g cm}^{-3} \pm .0014$) control ($p=0.0198$, Figure 40). This may have been due to the utilization of carbohydrate reserves for regrowth following

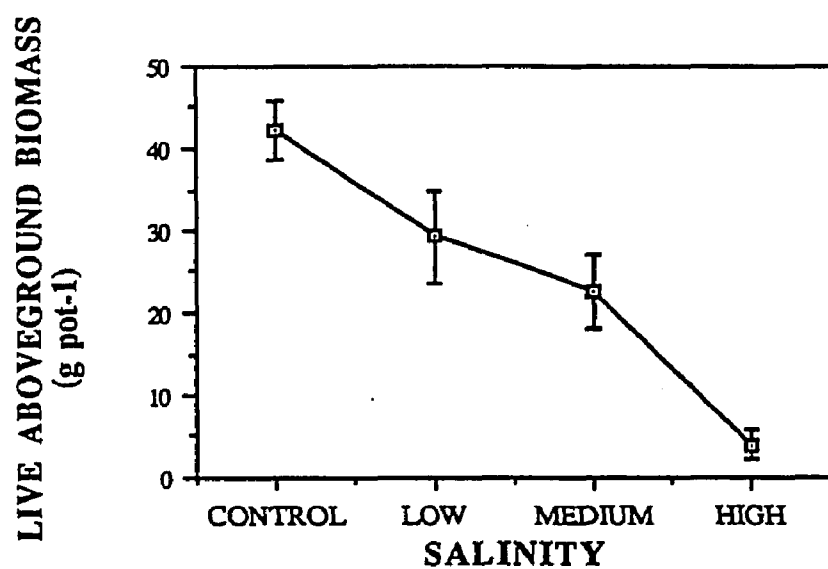


Figure 39. Live aboveground biomass, averaged over the drained and flooded treatments, as salinity increased.

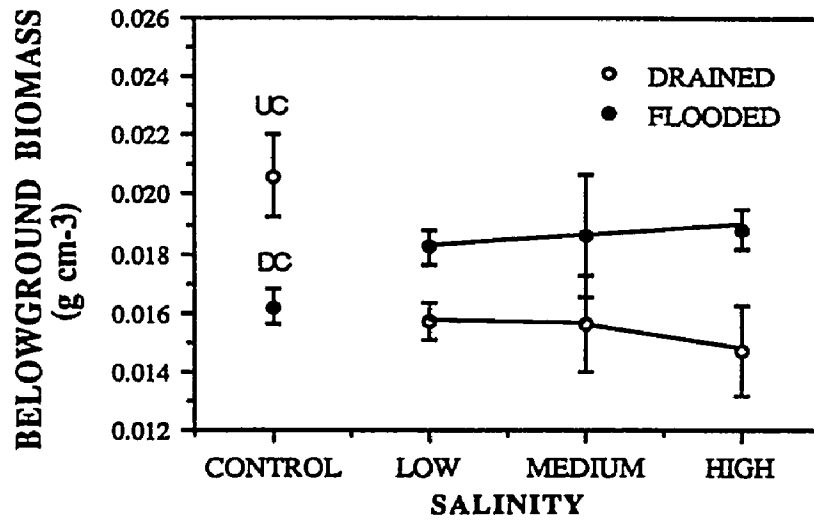


Figure 40. Belowground biomass in the controls (disturbed and undisturbed) and the drained and flooded mesocosms as salinity increased.

removal of the brown plastic bags from the disturbed controls once the aboveground biomass was killed. There were no significant differences in belowground biomass associated with increased salinity (Figure 40, [$p=0.2724$]).

Decomposition

Surface soil (upper 5 cm) decomposition rates varied significantly with treatment ($p=0.0089$, Figure 41) such that when averaged over salinity, significantly more rapid decomposition rates occurred in the drained mesocosms ($1.3 \text{ mg g}^{-1} \text{ day}^{-1} \pm .2$) than in the flooded ($.7 \text{ mg g}^{-1} \text{ day}^{-1} \pm .08$) mesocosms ($p=0.0034$). This finding explains the greater belowground biomass present in the flooded compared to the drained mesocosms.

Significantly different rates of decomposition occurred in the surface soils as salinity increased ($p=0.0302$), however, these changes varied significantly with flooding ($p=0.0068$) (Figure 41). Decomposition rates did not significantly differ between the undisturbed and disturbed controls ($p=0.4459$) suggesting that the lower amount of belowground biomass present in the disturbed control as compared to the undisturbed control (Figure 40) was due to the utilization of stored carbon for regrowth after the killing treatment. There was no significant treatment effect on decomposition rates at a depth of 13 cm ($p=0.2103$).

Soil Parameters

Soil Reduction

Recovery-phase conditions resulted in the development of significant differences in both surface Eh ($p=0.0001$) and Eh at the 15 cm depth ($p=0.0001$). Flooding reduces the rate of oxygen diffusion into the soil by a factor of 10,000 (Greenwood, 1961). The continued respiration of vegetation and soil microorganisms in flooded soils causes the development of biochemically reduced soil conditions (Turner and Patrick, 1968). Therefore, it is not surprising that the flooded mesocosms were significantly more

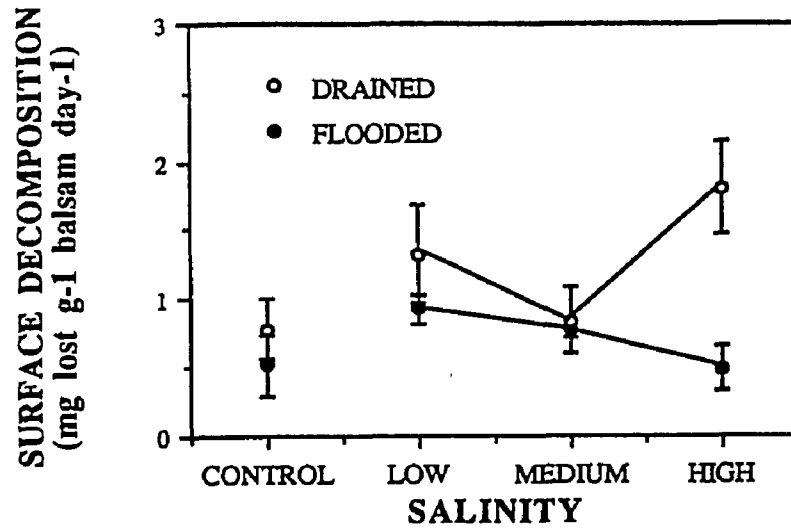


Figure 41. Surface (upper 5 cm) soil decomposition rates in the controls and in the drained and flooded treatments as salinity increased.

reduced than the drained mesocosms (averaged over salinity) at both the soil surface ($p=0.0001$) and at a depth of 15 cm ($p=0.0069$) (Surface: Drained Eh: $275 \text{ mV} \pm 18$, Flooded Eh: $-105 \text{ mV} \pm 11$; 15 cm deep Eh: Drained: $-77 \text{ mV} \pm 12$; Flooded: $-125 \text{ mV} \pm 8$).

Regardless of the salinity treatment, surface soils in the drained mesocosms were only moderately reduced while those in the flooded mesocosms were reduced (low salinity treatment) or highly reduced (medium and high salinity treatments, [Figure 42a]). Soils at the 15 cm depth were classified as reduced in all but the flooded high salinity mesocosms which, when averaged over all sample dates, were highly reduced (Figure 42b).

Soil reduction, averaged over the drained and flooded treatments, became increasingly severe as salinity increased in both the surface soils ($p=0.0012$) and at a depth of 15 cm ($p=0.0001$, [Figure 43]). However, it is evident from Figure 42a that this development of more reduced conditions at the soil surface was primarily due to changes that occurred in the flooded mesocosms.

Surface Eh varied significantly with time across the treatments (time x treatment: $p=0.0008$). Averaged over salinity, more reduced surface soil conditions developed over time in the flooded treatments as compared to the drained treatments ($p=0.0001$, [Figure 44]). In addition, soil conditions became increasingly more reduced as salinity increased although soil reduction was more severe in the flooded low, medium, and high salinity mesocosms than in the drained low, medium, and high salinity mesocosms ($p=0.0214$, [Figure 44]). There were significant differences in surface Eh between the disturbed and undisturbed controls over time (time x undisturbed control versus disturbed control: $p=0.0478$), however, these differences were not consistent. Soil reduction at the 15 cm depth did not vary with time ($p=0.4251$) or with the time x treatment interaction ($p=0.3012$).

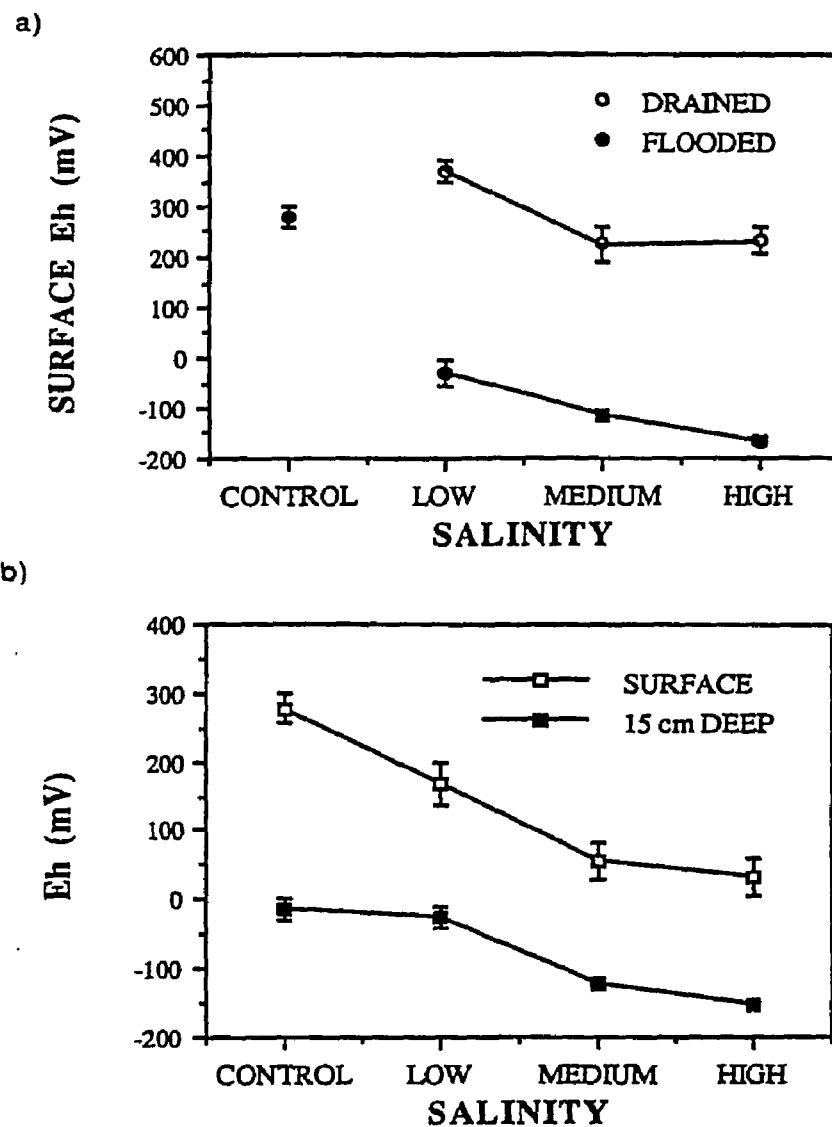


Figure 42. Soil reduction (a) at the surface in the drained and flooded mesocosms as salinity increased, and (b) at the surface and at a depth of 15 cm as salinity increased.

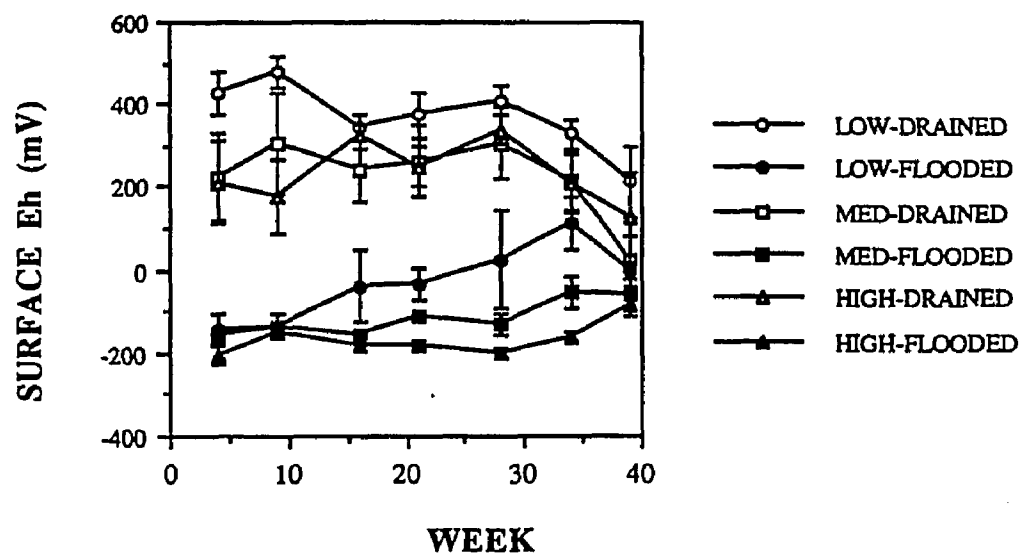


Figure 43. Soil reduction over time in the low, medium, and high salinity treatments in the drained and flooded mesocosms.

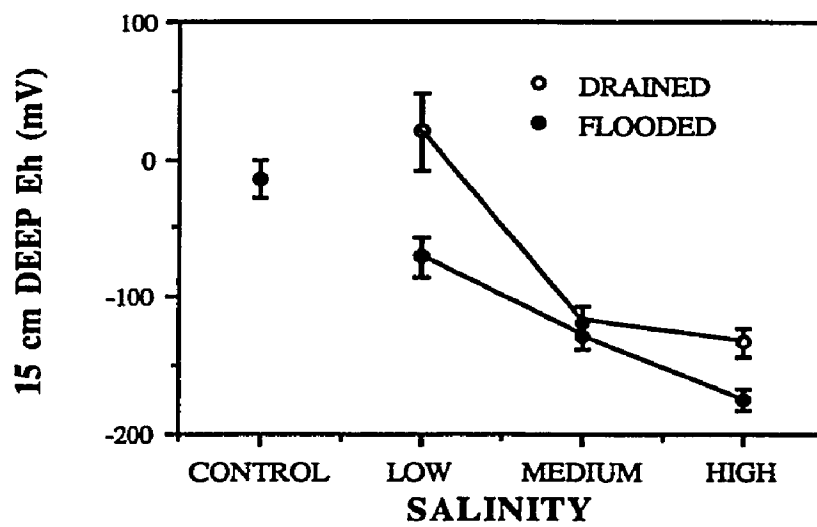


Figure 44. Soil reduction at a depth of 15 cm in the drained and flooded mesocosms as salinity increased.

Flooding of soils results in a dramatic decrease in the rate of oxygen diffusion (Gambrell and Patrick, 1978). This reduced rate of diffusion and continued demand for oxygen by plant roots and soil microorganisms soon results in the development of reduced soil conditions. Hook (1984) and Kozlowski (1984) have reported that plants experiencing an increase in the depth and duration of flooding may develop root oxygen deficiencies, experience decreased nutrient uptake, and/or be exposed to the buildup of toxic compounds (such as sulfide) in the soil. All of these conditions could, singly or in tandem, result in decreased growth and/or the elimination of particular species from the affected area.

Interstitial Water

Salinity

The recovery-phase conditions resulted in a significant linear increase in interstitial salinity levels as salinity treatments increased from low to high ($p=0.0001$). However, this increase was significantly different in the drained and flooded treatments (Figure 45a, $p=0.0001$). When averaged over salinity, the drained treatments had significantly lower salinity levels ($5.25 \text{ ppt} \pm .19$) than did the flooded ($6.55 \pm .23$) treatments ($p=0.0001$). This was true despite attempts to maintain equal salinities in the two water level treatments by periodic addition of an Instant Ocean solution throughout the recovery phase. McKee and Mendelssohn (1989) also experienced difficulties in maintaining equal salinity levels while conducting a greenhouse experiment investigating the effects of increased salinity and water level on freshwater marsh vegetation. However, they had a problem maintaining the salinity level in the flooded treatment rather than in the drained treatment.

During the recovery-phase interstitial salinity significantly increased in a linear fashion with time ($p=0.0001$) particularly in the drained high salinity treatment. In addition, salinity was significantly higher in the flooded treatments than in the drained treatments throughout this experiment (time x drained versus flooded: $p=0.0001$).

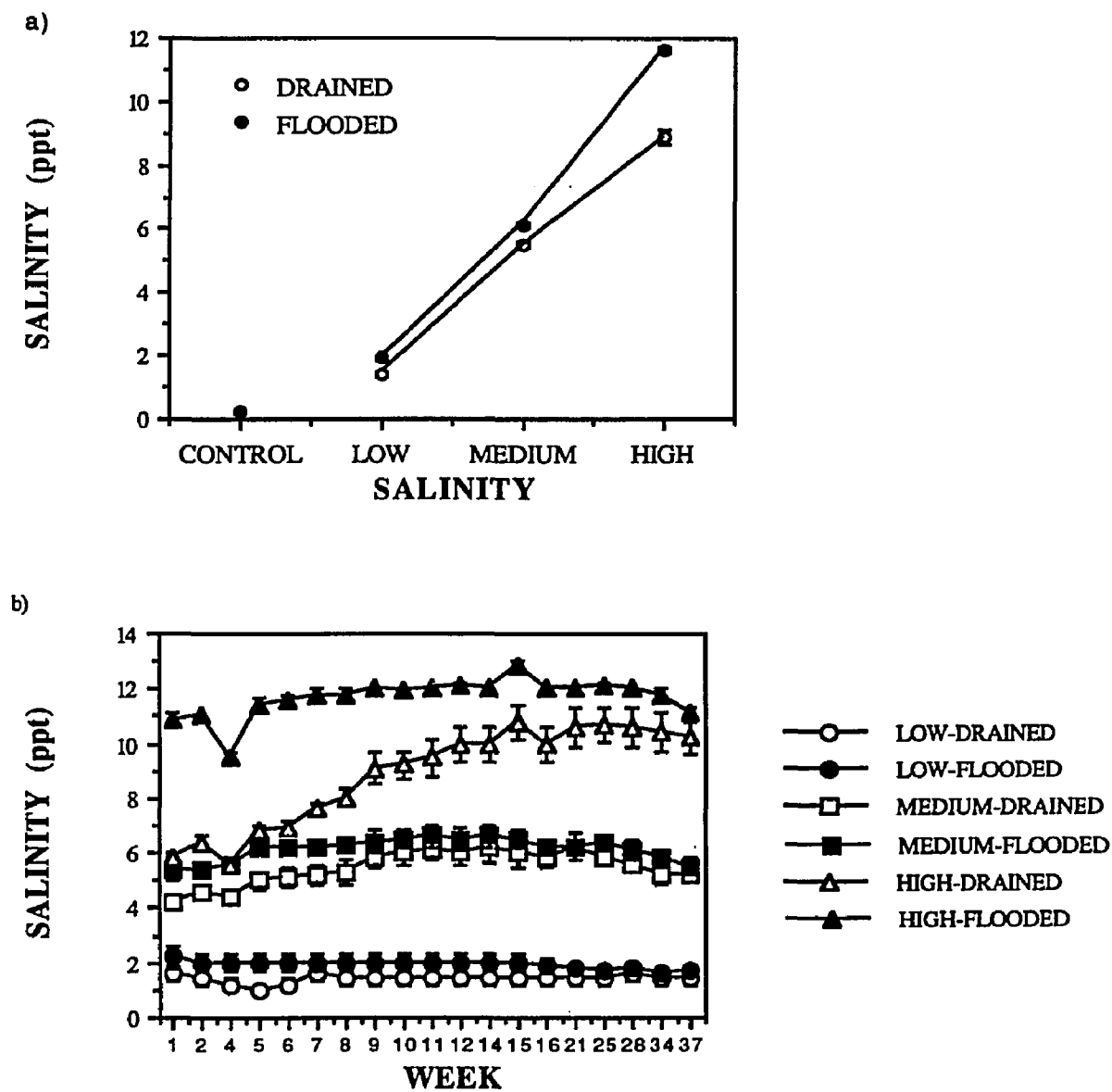


Figure 45. Interstitial salinity (a) in the controls and the low, medium, and high salinity drained and flooded mesocosms and (b) by salinity treatment in the drained and flooded mesocosms over time.

Periodic addition of higher salinity waters interspersed with the addition of fresh water ensured that evapotranspiration did not result in the concentration of salts and development of higher than target salinities. These periodic additions did result in significant oscillations in the salinity levels of each treatment (time x salinity x water: $p=0.0024$), especially the higher salinity treatments (Figure 45b) over the course of the recovery-phase. However, a similar effect would probably be evident in the field due to the occurrence of periodic rains.

Sulfide

Interstitial concentrations of sulfide increased significantly as salinity increased ($p=0.0001$, Figure 46a) however, this increase was significantly greater in the flooded mesocosms (37.4 ± 3.5) than the drained mesocosms (15.3 ± 1.3 ; $p=0.0001$, [Figure 46a]). When averaged over the salinity treatments, the flooded mesocosms had sulfide concentrations that were 2.4 times higher than those present in the drained treatments. This highly significant difference ($p=0.0001$) was probably due to the greater salinity levels that were, on average, present in the flooded mesocosms (Figure 45a), as well as the significantly more reduced soil conditions at both the soil surface (Figure 42a) and at a depth of 15 cm (Figure 42b).

Sulfide concentrations varied significantly over time with the recovery-phase treatments ($p=0.0001$). Throughout the experiment, higher salinity levels were associated with significantly increased sulfide concentrations (Figure 44b) but this increase varied with water level ($p=0.0001$, data not shown). Over time, there were significantly higher sulfide concentrations (averaged over salinity) in the flooded mesocosms than the drained mesocosms (time x drained vs flooded: $p=0.0002$).

Instant Ocean, which was used to create the salinity treatment levels has a chemical composition that is very similar to seawater and therefore served as a source of sulfate in much the same way Gulf waters would under natural conditions. Sulfate is

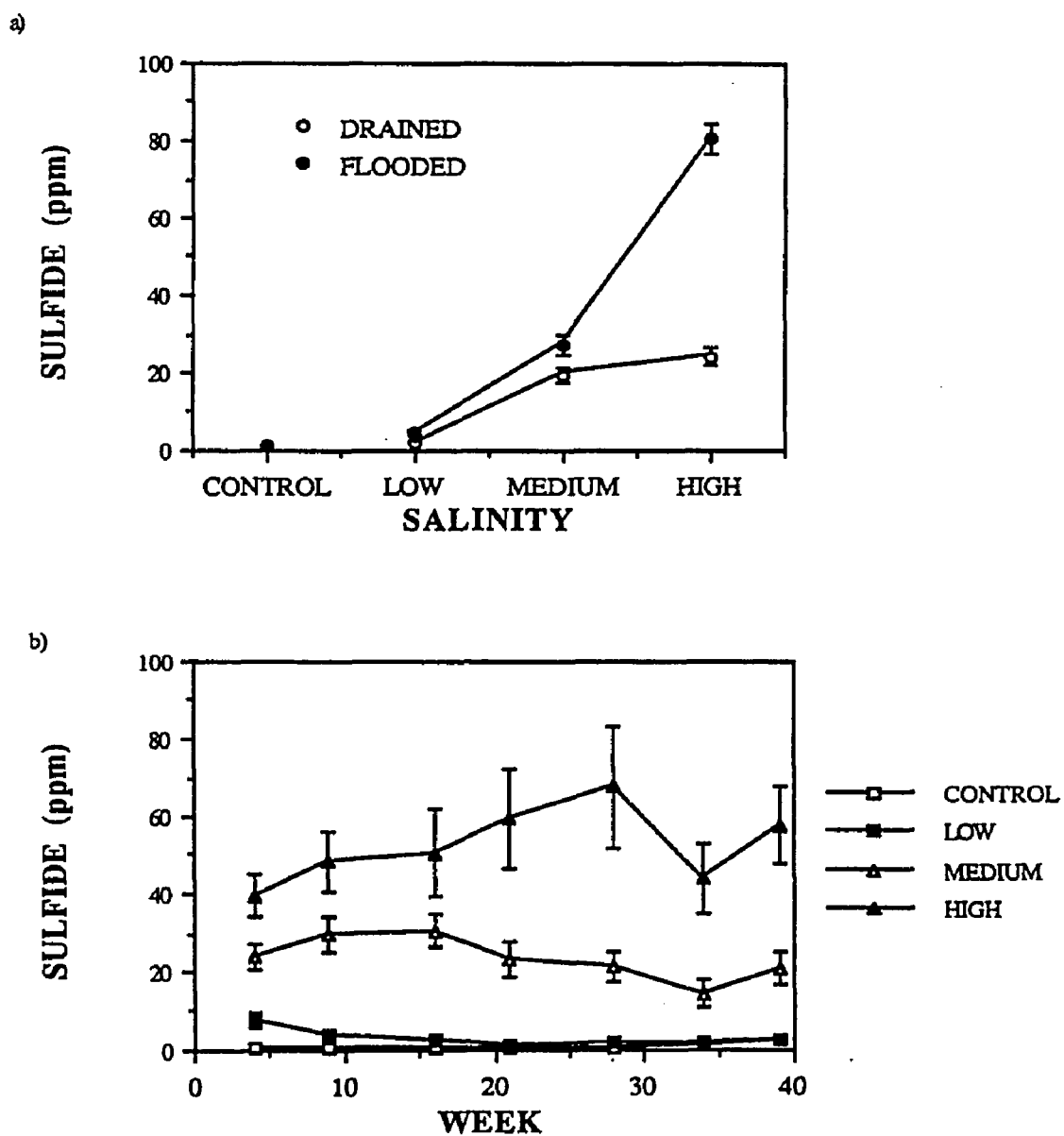


Figure 46. Sulfide concentrations (a) in the drained and flooded mesocosms as salinity increases and (b) over time in the control mesocosms and the low, medium, and high salinity treatments.

reduced to sulfide, a plant phytotoxin, under reduced soil conditions. The controls and the low salinity mesocosms received no salt solution or a very dilute salt solution, respectively, and had less reduced soil conditions than the medium and high salinity treatments (Figure 42a -b). The medium and high salinity mesocosms received greater amounts of the Instant Ocean solution and developed more reduced soil conditions (Figure 42a-b). Therefore, these mesocosms had proportionately greater amounts of sulfate available for reduction to sulfide (Figure 46) and hence significantly greater concentrations of sulfide, especially in the flooded treatments.

Sulfide, which is a phytotoxin at high concentrations (Ponnamperuma, 1972), accumulates in reduced soils and can exert a toxic effect in several ways: 1) directly inhibiting the activity of respiratory enzymes (Allam and Hollis, 1972; Pearson and Havill, 1988); 2) indirectly by complexing with metals essential for enzyme reactions (Havill et al., 1985); 3) indirectly by acting as an oxygen sink, and/or 4) by inhibiting nutrient uptake (Koch et al., 1990). Susceptibility of vegetation varies with species, but sulfide results in reduced plant growth (Koch and Mendelssohn, 1989; Linthurst, 1979; Mendelssohn and McKee, 1988).

Interstitial pH

The pH of the interstitial water was significantly affected by recovery-phase conditions ($p=0.0001$). Interstitial pH increased in a linear fashion as salinity increased ($p=0.0001$). However, this linear increase in pH varied significantly with water level--being more gradual in the drained mesocosms than in the flooded mesocosms ($p=0.0223$, Figure 47). Averaged over the salinity treatments, flooding did not result in any significant differences in pH ($p=0.8376$). There was a significant time x treatment effect on pH ($p=0.0001$) (Figure 47) but the average pH on each date remained between 6.2 and 7.0. This is within the range of pH found in a Louisiana fresh floating marsh as reported

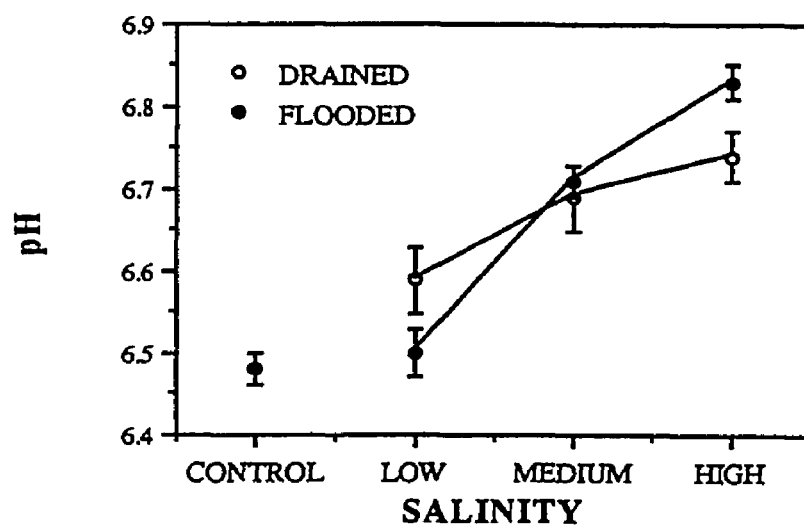


Figure 47. Interstitial pH in the drained and flooded mesocosms as salinity increased.

by Sasser and Gosselink (1984). It is probable that pH differences as small as the ones that occurred between the treatment combinations in this experiment were not biologically significant.

Macronutrients--Na, K, Ca, and Mg

Recovery-phase conditions resulted in significant differences in concentration of the macronutrients Na, K, Ca, and Mg ($p=0.0001$ for Na, K, Ca, and Mg). Increasing salinity was associated with significant linear increases in the concentrations of all four macronutrients ($p=0.0001$ for Na, K, Ca, and Mg, Figure 48a-d). These elements are the major cation components in seawater (Weyl, 1970) and since Instant Ocean has the chemical composition of seawater, it is not surprising that their concentration increased with increasing salinity.

As salinity increased, there were statistically significant differences in the concentrations of these macronutrients present in the drained mesocosms as compared to the flooded mesocosms. These differences were so small however (Figure 48a-d), that they were probably not biologically significant and were primarily the result of lower salinity levels in the drained mesocosms (Figure 45a). Averaged over salinity, there were no significant differences in the concentrations of Na ($p=0.4608$), Ca ($p=0.4845$), or Mg ($p=0.1531$) between the drained and flooded mesocosms. However, concentrations of K tended to be higher in the flooded mesocosms than in the drained mesocosms ($p=0.0519$, Figure 48b). This trend may have been due to the higher salinity levels reached in the flooded as compared to the drained mesocosms.

Concentrations of Ca and Mg varied significantly with treatment over time ($p=0.0001$). The higher salinity treatments had higher concentrations of Ca and Mg, but within each salinity treatment, both Ca and Mg decreased from January, 1991 to May, 1991 (data not shown)--possibly due to the decrease in salinity which occurred towards the end of the experiment (Figure 45b). Concentrations of Na decreased significantly

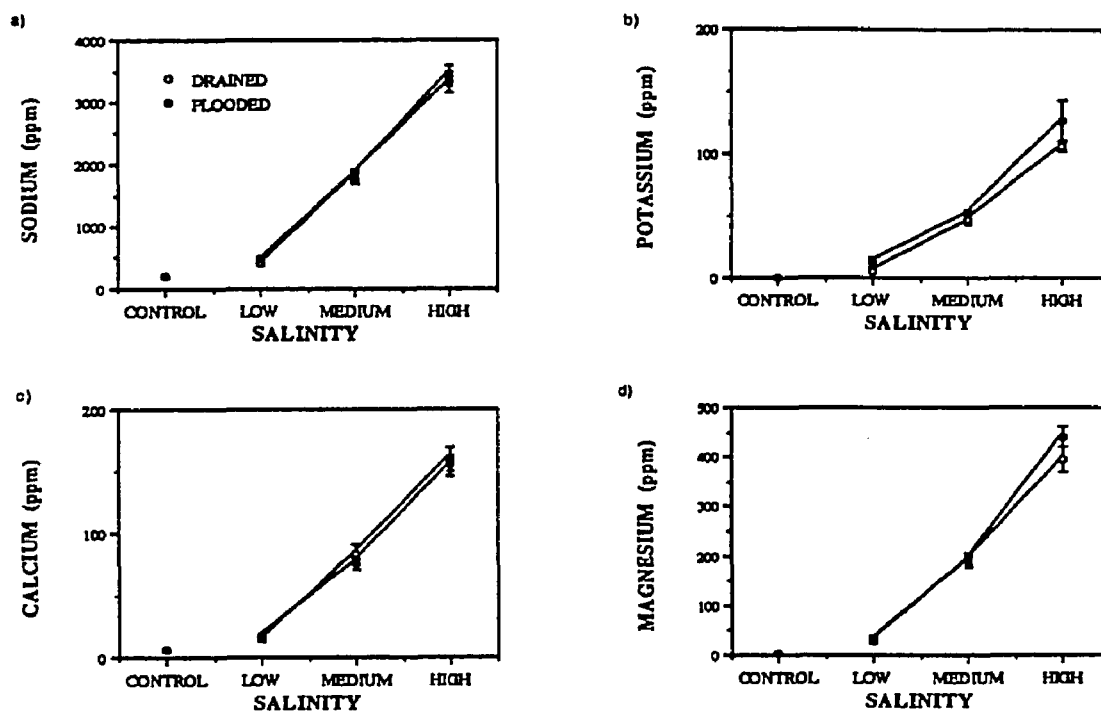


Figure 48. Concentrations of (a) sodium, (b) potassium, (c) calcium, and (d) magnesium in the drained and flooded mesocosms as salinity increased.

with time (time: $p=0.0040$, data not shown) in all treatment combinations. Again, this decrease was probably in response to the decrease in salinity that occurred near the end of the experiment. Concentrations of K did not vary with time ($p=0.0568$) or with the time x treatment interaction ($p=0.8291$).

Interstitial Iron and Zinc

Iron and Zn were present in significantly lower concentrations in the low, medium, and high salinity treatments than in the controls (Fe: $p=0.0001$; Zn: $p=0.0059$; [Figure 49a-b]). Averaged over salinity, there were no significant differences in Fe and Zn associated with flooding. However, as salinity increased, concentrations of Zn approached statistically significant higher concentrations in the drained mesocosms than in the flooded mesocosms ($p=0.0515$, Figure 49b). The overall effect of flooding upon Zn is a decrease in solubility (Ponnampuruma, 1972). In addition, the presence of sulfide could have reduced Zn concentrations through the formation of ZnS which would occur under reducing conditions (Engler and Patrick, 1975). The higher sulfide concentrations and more reduced conditions present in the flooded mesocosms (especially in the medium and high salinity treatments) would allow for the decrease in Zn in these mesocosms.

Iron is reduced from the ferric to the ferrous form as moderately reduced (≤ 120 mV) soil conditions develop. When redox potentials are low enough for sulfide to form, reducible Fe compounds in the soil will precipitate with sulfide to form insoluble ferrous sulfide (FeS) (Gambrell and Patrick, 1978; Patrick and Mikkelsen, 1971). The significantly lower Fe concentrations present in the low, medium, and high salinity treatments as compared to the controls may have been due to precipitation of Fe with sulfide. Higher concentrations of Fe were present over time in the controls than in the treatments, and while concentrations remained stable in the treatment mesocosms, they increased in the control mesocosms ($p=0.0025$). Concentrations of Zn did not change significantly with time ($p=0.9675$).

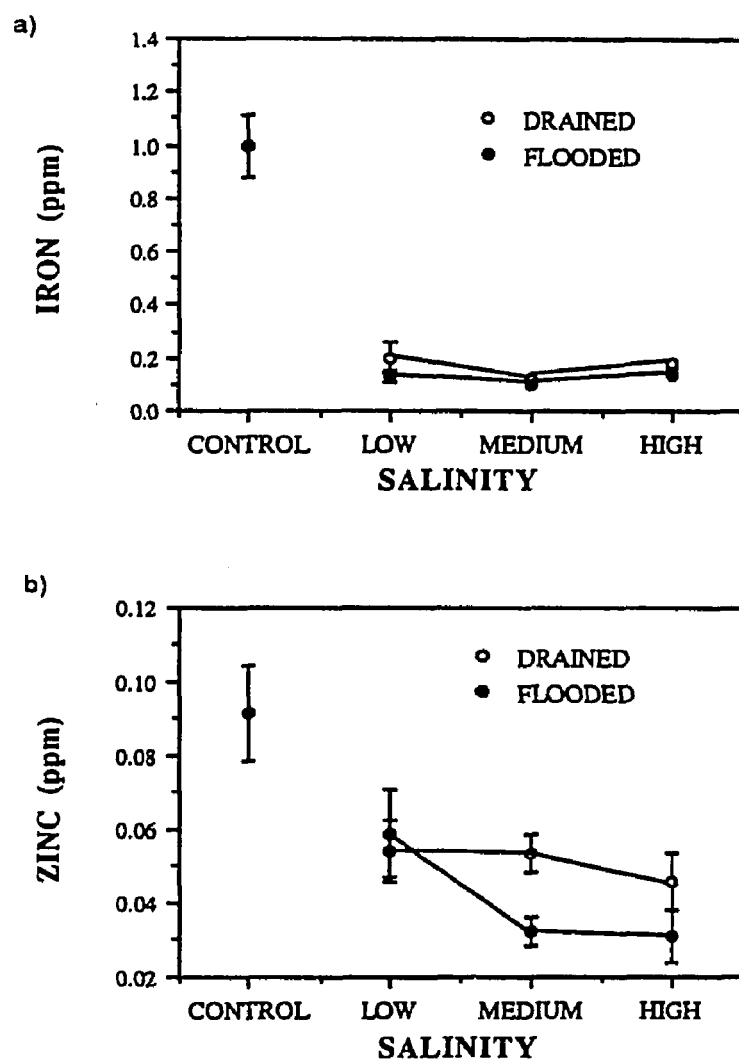


Figure 49. Concentrations of (a) iron and (b) zinc in the drained and flooded mesocosms as salinity increased.

Manganese

Concentrations of manganese increased significantly as salinity levels increased ($p=0.0001$), however, this increase was greater in the flooded mesocosms than in the drained mesocosms ($p=0.0090$, Figure 50). Concentrations of Mn decreased significantly with time--especially in the medium and high salinity treatments (time x salinity interaction: $p=0.0124$). The concentrations present in the control and low salinity mesocosms did not appear to change significantly with time (data not shown). These changes over time were similar to the changes observed in concentrations of Na, K, Mg, and Ca and were probably related to the decrease in salinity observed towards the end of the experiment. Manganese can, in the presence of sulfide under anaerobic conditions, precipitate as insoluble MnS (Engler and Patrick, 1975), however, since concentrations of Mn were higher in the treatment mesocosms than in the controls, this did not appear to occur.

Copper

As salinity increased, Cu increased significantly ($p=0.0001$), however, this increase was more pronounced between the low and medium salinity treatments than between the control and low salinity treatment or the medium and high salinity treatment (treatment x salinity effect: $p=0.0134$, Figure 51). There was a significant time x treatment interaction ($p=0.0030$) such that concentrations of Cu increased with increasing salinity. However, while concentrations increased in the medium and high salinity treatments between January, 1991 and the time of harvest, they decreased in the control and low salinity mesocosms.

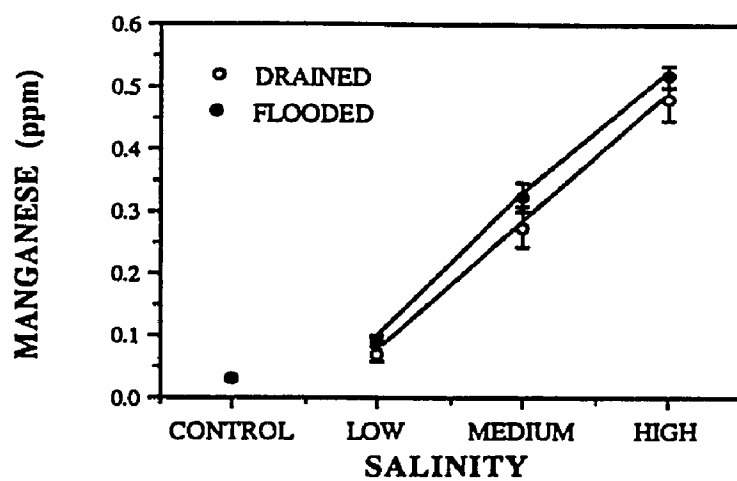


Figure 50. Concentrations of manganese in the drained and flooded mesocosms as salinity increased.

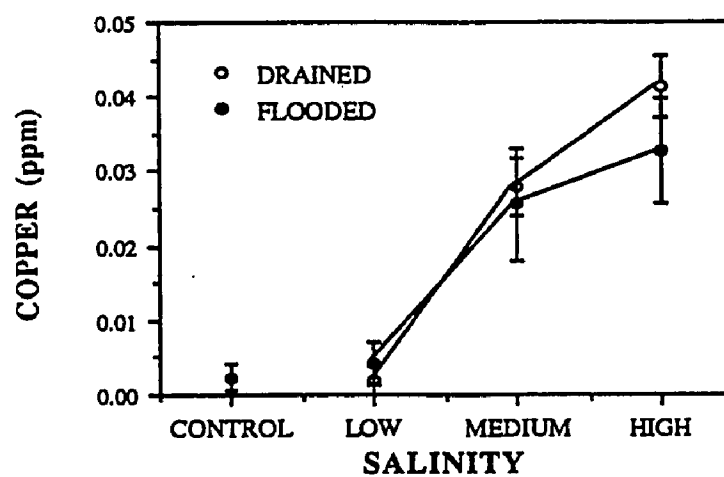


Figure 51. Concentrations of copper in the drained and flooded mesocosms as salinity increased.

Phosphorus

Phosphorus was the only element with significantly higher concentrations in the flooded mesocosms than in the drained mesocosms ($p=0.0001$, Figure 52). Phosphorus availability is indirectly tied to Eh because of transformations of other elements with which it may precipitate. Under aerobic conditions, such as existed in the drained mesocosms, P may exist as insoluble Fe-, Ca-, or Al-phosphate (Mahapatra and Patrick, 1969). When anaerobic conditions ($Eh \leq 200\text{mV}$) develop, Fe, Ca, and Al become more soluble, and P compounds hydrolyze (Redman and Patrick, 1965). This usually results in an increase in extractable P upon submergence of soils due to its increased solubility (Patrick, 1964).

Interstitial Ammonium

Concentrations of ammonium were significantly affected by the recovery-phase conditions ($p=0.0001$), especially flooding which resulted in significantly higher concentrations of ammonium ($p=0.0003$, Figure 53). In submerged soils, nitrate-nitrogen is rapidly denitrified (Patrick and Wyatt, 1964). Because aerobic conditions are required for microbial oxidation of ammonium to nitrate, the breakdown of organic matter in anaerobic soils results in accumulation of ammonium-N (Redman and Patrick, 1965). Even though organic matter breakdown is slow and incomplete in anaerobic soils, anaerobic microorganisms have a low requirement for N (Patrick and Mikkelsen, 1971). As a result, soils having a high percentage of organic matter will produce more ammonium (Redman and Patrick, 1965). The higher interstitial ammonium concentrations in the flooded mesocosms (Figure 53), especially in the high salinity treatment, may have been due to the high organic matter content of the freshwater marsh soils and the lower Eh present in the flooded mesocosms as well as the decreasing amount of live biomass which would utilize ammonium as salinity levels increased (Koch and Mendelssohn, 1989; Mendelssohn and McKee, 1988).

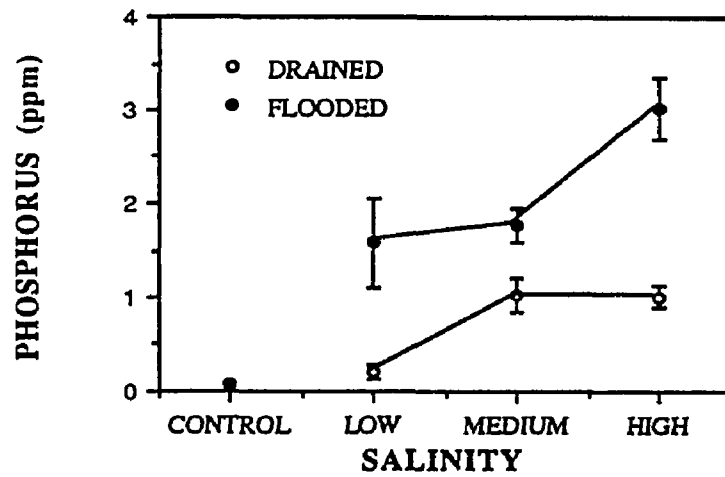


Figure 52. Concentrations of phosphorus in the drained and flooded mesocosms as salinity increased.

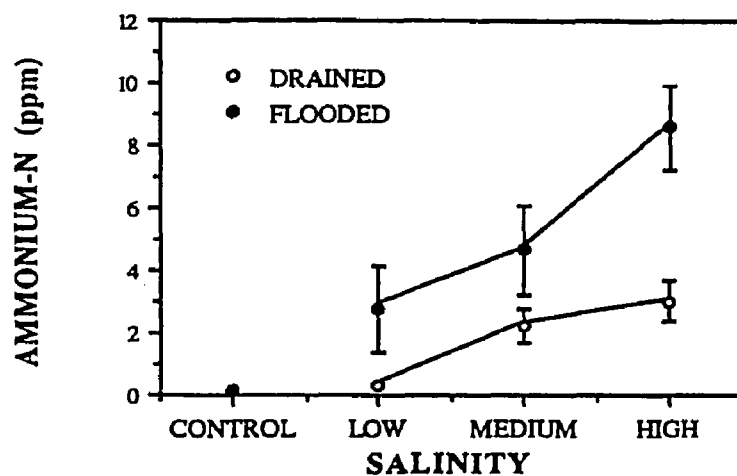


Figure 53. Concentrations of ammonium (a) in the drained and flooded mesocosms as salinity increases and (b) over time in the controls and the low, medium, and high salinity treatments.

SUMMARY

It has long been implicitly understood that increasing salinity in a fresh marsh would result in rapid and dramatic effects upon the vegetation. Because of the problems facing Louisiana's coastal wetlands, interest in this issue has resulted in several recent studies, such as this one, which provide insights into the mechanisms for this effect. Gosselink et al. (1977) and Drake and Gallagher (1984) have implicated increased salinity as one cause of decreased photosynthesis and growth of *Spartina alterniflora*, the dominant saltmarsh species in southern Louisiana. Mendelssohn and McKee (1988) found that *Spartina patens*, a dominant brackish marsh species, exhibited decreased growth (showing reduction in both height and biomass) and density when transplanted from a marsh having a salinity level of 15 ppt to one having 26 ppt. Increased water level also contributed to the reduction of *S. patens* density and biomass in this study.

This study goes one step further and considers the long-term ability of a freshwater marsh to recover from a die-back event caused by saltwater intrusion. The ability of a freshwater marsh to recover appears to be dependent on post die-back salinity and water level conditions. The species which revegetated the treatment mesocosms subsequent to the killing event had seeds or propagules which were able to withstand a period of 6 weeks in which the interstitial water salinity levels were at approximately 15 ppt. In addition, the freshwater marsh mesocosms had a sulfate source which, under the reduced conditions present once flooding was initiated, was reduced to sulfide. Despite the fact that sulfide, which is a known phytotoxin, was present at very high levels (data not shown) in the mesocosms during the saltwater intrusion event, revegetation did occur to some extent in all but the 12 ppt flooded treatment. This occurred despite the fact that freshmarsh species have been shown to be more susceptible to the effects of sulfide toxicity than saltmarsh species.

The level of inundation subsequent to dieback affected species diversity--fewer species were able to regenerate in the flooded treatment as compared to the drained treatment. While not statistically significant, live aboveground biomass tended to be lower in the flooded mesocosms when averaged over the three post-dieback salinity treatments than in the drained treatments. However, amounts of belowground biomass were significantly greater in the flooded mesocosms which had significantly lower rates of decomposition in the upper 5 cm of the soil.

Several conclusions can be reached based on the data collected in this experiment. First, the long-term effect of a saltwater intrusion event will be strongly influenced by post-intrusion salinity and water levels. Even after a relatively long saltwater intrusion event, such as the one simulated in this experiment, there is the potential for significant recovery of a freshwater marsh--as long as the post-intrusion salinity levels can be reduced--although complete freshwater conditions are not necessary for significant recovery. The extent of recovery decreased as post-intrusion salinity increased. In addition, the effect of flooding became more pronounced with higher salinity levels. The presence of significantly higher concentrations of sulfide in the flooded mesocosms as compared to the drained mesocosms may be the reason for this flooding effect.

This experiment has several implications with regards to the use of structural marsh management for addressing the problems facing Louisiana's wetlands. First, it is clear that the possibility of a saltwater intrusion event should not be overlooked when designing the structures to be used to accomplish the specific management goals. Structural marsh management designs should consider ways of rapidly removing impounded water from the management area. Also, since sulfide may exacerbate the salt effect, lowering water levels to aerate sod is even more important. It is especially important that this problem be considered for fresh marshes located closer to the coast or those created in more saline regions through the use of structural marsh management.

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OVERALL CONCLUSIONS

Hydrology is the primary forcing function of a wetland. Therefore, any activity or event that alters wetland hydrology may have profound effects on wetland form and function. Currently, the use of structural marsh management to address the problems facing Louisiana's coastal wetlands is becoming increasingly common. This method of management involves the use of structures which alter wetland hydrology with the goal of allowing the marsh managers to selectively raise or lower water levels by manipulating flap-gates, weirs, or other structures. One potential problem associated with this type of alteration is the decreased rate with which storm-deposited waters can drain off of managed marshes.

I investigated the effects of structural marsh management on the soil and vegetation of two brackish marshes (Rockefeller Wildlife Refuge and Fina LaTerre) and a fresh floating marsh (Avoca Island). In all three cases, the managed marshes were compared to nearby unmanaged marshes. In addition, I examined the effect of various post-die-back salinity and water levels on the ability of freshwater marsh vegetation to recover from a saltwater intrusion event. The information contained in this dissertation is intended to be used in developing effective monitoring schemes, predicting the effects of structural marsh management under various conditions, and determining the potential for recovery should a storm surge inundate a managed marsh.

At Rockefeller Wildlife Refuge, management resulted in a three-fold increase in live aboveground biomass as compared to the unmanaged marsh. However, at Fina LaTerre, live aboveground biomass was significantly greater in the unmanaged marsh than in the managed marsh. There were no significant differences in live aboveground biomass between the managed and unmanaged marshes at Avoca Island. Similar results were obtained when the primary productivity of each marsh was calculated. At Rockefeller primary productivity in the managed marsh was approximately 4 times greater

than in the unmanaged marsh. At Fina LaTerre, primary productivity was approximately twice as high in the unmanaged marsh while at Avoca Island there were no significant differences between the two marshes.

In order to determine what environmental factors control the differences observed in live aboveground biomass and primary productivity between management sites, water depth and soils data were collected. At Avoca Island the distance from the floating mat surface to the underlying clay pan was measured in order to determine if the use of levees and control structures would affect the oscillations or the levels of the floating mat. The water management schedule at Avoca Island did not result in any significant differences between the managed and unmanaged marshes. At Fina LaTerre and Rockefeller, the depth of standing water, if present, or the distance to the water table if standing water was not present was measured. At both Fina LaTerre and Rockefeller, there was, on average, a significantly higher water level in the unmanaged marsh as compared to the managed marsh. In this respect, both plans met the intended goal of lowering water level.

Surface soil conditions, like water depth, were similar in both marshes at Avoca Island. At Rockefeller, surface soils were significantly less reduced in the managed marsh. However, at Fina LaTerre, despite lower water levels in the managed marsh, surface soils were significantly more reduced in the managed marsh. Based on measurements made by Cahoon (1990), in the same marsh, these more reduced soil conditions may have been due to: 1) the lower mineral content of surface soil in the managed marsh which could have resulted in a lower ability to poise Eh at higher redox potentials, or 2) the presence of more labile organic matter resulting from organic matter decomposition in the managed marsh which had higher levels of total organic matter. Measurement of soil conditions at a depth of 15 cm showed significantly less reduced conditions in the managed marsh at both Avoca Island and Rockefeller. In contrast, there were no significant differences in the 15 cm Eh of Fina LaTerre soils.

However, in view of the potential problem associated with storm surges, it is interesting to note the effect precipitation can have on water level in a managed marsh. At Rockefeller, in 1989, the significant overall differences in water depth and Eh (both at the surface and at a depth of 15 cm) were primarily due to the extremely low water table in May. Between the sample dates of May and August, precipitation resulted in a dramatic water level increase in the managed marsh and development of reduced soil conditions. The management plan called for a drawdown of water levels in the managed marsh during 1989, but the effects of heavy precipitation offset the ability to lower water level. This is indicative of the problems that would result from a storm surge except that this water was fresh rather than saline.

Interstitial sulfide concentrations at Avoca Island, which is a fresh marsh, were below detection limits (i. e. 0.1 ppm). Both the Fina LaTerre site and Rockefeller are brackish marshes that are influenced to some degree by saline water. At the Fina LaTerre site, sulfide concentrations averaging 8-9 ppm were measured in both the managed and unmanaged marshes. In contrast, at Rockefeller, sulfide concentrations were approximately 25 times greater in the unmanaged marsh than in the managed marsh. Since sulfide is a phytotoxin, the higher concentration in the unmanaged marsh at Rockefeller could have been a factor in the lower live aboveground biomass and primary productivity measured there.

Salinity was below 0.5 ppt (i. e. fresh water) in both the managed and unmanaged marshes at Avoca Island. At Fina LaTerre, although interstitial salinity concentrations were statistically greater in the managed marsh ($3.3 \text{ ppt} \pm 0.1$) than in the unmanaged marsh ($2.3 \text{ ppt} \pm 0.1$) this difference was not likely to be significant biologically since the dominant plant species was *Spartina patens*, a common brackish marsh grass. In contrast to both Avoca Island and Fina LaTerre, interstitial salinities at Rockefeller were more than twice as high in the unmanaged marsh. This was due to the exclusion of saline waters

from the managed marsh and the occurrence of heavy precipitation which would serve to flush the soils.

Of the three marshes studied, the management plan at Rockefeller was the most successful at meeting the goals of structural marsh management. This is evidenced by the higher live aboveground biomass and primary productivity. The management plan resulted in lower water levels, less reduced soil conditions at both the surface and at a depth of 15 cm, and lower sulfide and salinity concentrations. It must be noted that the management plan at Rockefeller had been in effect for approximately 32 years when this study was conducted.

In contrast, management at Fina LaTerre, which had only occurred for 4 years prior to this study, resulted in lower live aboveground biomass and primary productivity than the unmanaged marsh. Although the more reduced conditions in the managed marsh contributed to lower primary productivity, it is likely that there were other factors not measured that were controlling primary productivity.

The structural marsh management plan at Avoca Island appears to have had little effect on the variables reported here. However, it is important to note that the plan had only been in effect for 2 years prior to this study and it is possible that any beneficial or harmful effects of management would not be detected this early. On the other hand, it is also possible that water-level management of floating marshes will have no effect due to the oscillating nature of these areas.

Results of the greenhouse experiment indicate that post-die-back recovery-phase conditions are very important in determining the extent to which a freshwater marsh can recover from a lethal saltwater intrusion event. These data prove that regrowth of freshwater marsh vegetation is possible even after a 6 week saltwater intrusion event if post die-back conditions are favorable. As the salinity level occurring during the recovery phase increased, species richness and live aboveground biomass decreased. The decrease in species richness with increasing salinity was exacerbated by flooding such that the

combination of flooded conditions and low, medium, and high salinity levels resulted in the regrowth of only $3.7 \pm .5$ species.

Flooded mesocosms had more reduced soil conditions and greater concentrations of sulfide than did the drained mesocosms. These conditions resulted in a more stressful situation under which regrowth of vegetation had to occur. It is obvious that the potential of a managed freshwater marsh to recover from a saltwater intrusion event will depend on the ability to drain the marsh as well as the ability to freshen the interstitial water thereby avoiding high salinity, reduced soil conditions, and elevated concentrations of sulfide.

Comparison of the three managed marshes indicates that structural marsh management is neither all good nor all bad. These results do indicate that use of structural marsh management is not the universal answer to the problems faced by Louisiana's wetlands. It is likely that the benefits of marsh management will vary with location (east-west, near or far from the coast), with design, with marsh type, and with amount of money expended. In addition, methods of dealing with storm surges or unusual amounts of precipitation should be available so as to avoid the loss of marsh vegetation.

The use of standardized monitoring requirements that do not place an unrealistic burden on the landowner or marsh manager is extremely important if this type of information is to become available. I recommend that, in areas where structural marsh management is being used to prevent salt water intrusion, interstitial salinity be measured in the interior of the managed marsh as well as in a nearby unmanaged marsh. In addition, several permanent sites (the number to be dependent on the size of the managed area) should be established for measuring water depth in both the managed and a nearby unmanaged marsh. This is the very minimum in terms of monitoring requirements. Emergent plant species composition, percent cover, and land to water ratios are also potentially valuable tools in evaluating management.

VITA

Kathryn Marie Flynn was born on July 24, 1957, in Birmingham, Alabama. Upon graduation from John Carroll High School in 1975 she entered Auburn University. She received a B. S. degree in Botany from Auburn in March, 1980 after which she worked for several years in a human Virology laboratory at the University of Alabama at Birmingham. This experience, while valuable in many ways, convinced her to return to the world of plants. She entered the Marine Sciences department at Louisiana State University in August, 1983. Her thesis involved the growth and metabolic responses of *Taxodium distichum* (baldcypress) to different water levels. She received her M. S. degree in December, 1986 and, after a one year hiatus, she returned to Louisiana State University's Department of Oceanography and Coastal Sciences to pursue her Ph. D. She investigated the effects of hydrologic and salinity changes on Louisiana's marshes. Upon completion of her Ph. D. degree she will return to Auburn University where she will work as an Extension Forester/Assistant Professor in the School of Forestry. She will be involved with wetland issues in the state of Alabama.

DOCTORAL EXAMINATION AND DISSERTATION REPORT

Candidate: Kathryn Marie Flynn

Major Field: Oceanography and Coastal Sciences

Title of Dissertation: Effects of Hydrologic Alterations on the
Vegetation and Soils of Marshes in Southern
Louisiana

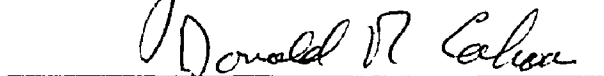
Approved:


Major Professor and Chairman

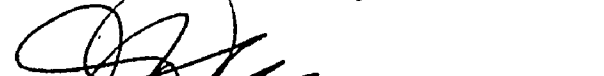

Dean of the Graduate School

EXAMINING COMMITTEE:













Date of Examination:

March 20, 1992