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THE CREATION AND RESTORATION OF WETLANDS BY NATURAL PROCESSES IN THE LOWER ATCHAFALAYA RIVER SYSTEM: POSSIBLE CONFLICTS WITH NAVIGATION AND FLOOD CONTROL OBJECTIVES

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ABSTRACT

The role of the Atchafalaya River is creating new wetlands in the Atchafalaya Bay delta, as well as its role in restoring deteriorating wetlands in peripheral marshes, is discussed and documented. Measurements from maps and aerial imagery indicate a local dramatic reversal in the rate of wetland loss. The loss rate for all of coastal Louisiana is 128 km^2/yr. Data from the examination of numerous Landsat scenes during flood periods from 1973 to 1980 demonstrate the direct relationship between suspended sediment influx and wetland loss reversal. Frequency of inundation was calculated by relating sediment plume distribution to river stage and wind conditions to give a synoptic picture of sediment influx for the period in question.

Plans for proposed navigation and flood control projects under consideration by the U. S. Army Corps of Engineers are reviewed and possible conflicts with the documented wetland restoration are discussed. Identified here are alternate flood control and navigation proposals that will accomplish the goals of navigation and flood control without removing the current potential for wetland restoration by the Atchafalaya River.

INTRODUCTION

The piracy of Mississippi River flow by the Atchafalaya River represents a major, though expected, change in the course of the largest river system on the North American continent. Throughout the last five to seven thousand years since sea level reached its present elevation, upstream diversions of Mississippi River flow have caused the locus of sedimentation to change approximately once every one to two thousand years (Kolb and Van Lopik, 1966) (Figure 2). This process ensured a net gain of wetlands in the region until the end of the 19th century. In modern times, however, the continued accretion of wetlands in the Mississippi deltaic plain has effectively ceased, primarily because of the intentional closure of many distributaries and the completion of the artificial levee system. Most of the sediment necessary for maintaining existing wetlands and creating new ones presently debouches near the edge of the continental shelf at the mouth of the Mississippi River. A loss of coastal wetlands of 102 km^2/yr (39.4 mi^2/yr) in the Mississippi deltaic plain has been documented by Gagliano (1981) and an additional wetland loss of 26 km^2/yr (10.0 mi^2/yr) in the chenier plain (marginal Mississippi deltaic plain) has been documented by Gosselink et al. (1979) along the southwest Louisiana coast (Figure 2).

The building Atchafalaya delta system south of Morgan City, Louisiana (Figure 2), represents the only area within the state where wetlands are substantially increasing. Before the formation of the Atchafalaya delta, concepts of Mississippi deltaic growth were largely derived from interpretations of preserved sedimentary sequences, as there have been no actively prograding subaerial shallow water deltas since the late 19th century. Thus, examination of the Atchafalaya delta provides

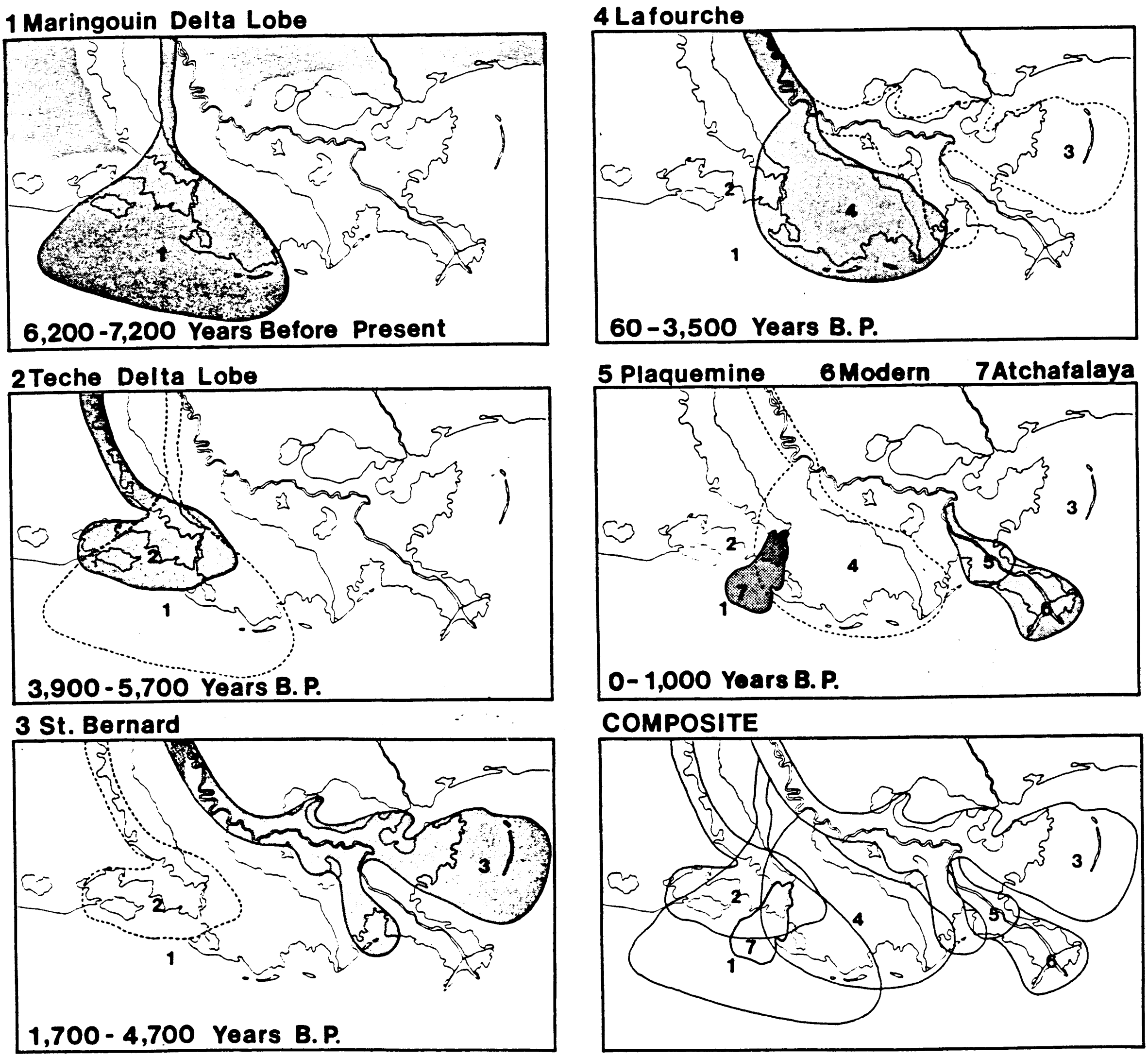

Figure 1. Mississippi River delta lobes and the evolution of the Louisiana coastlaine (after Adams and Baumann, 1980, as modified from Frazier, 1967).

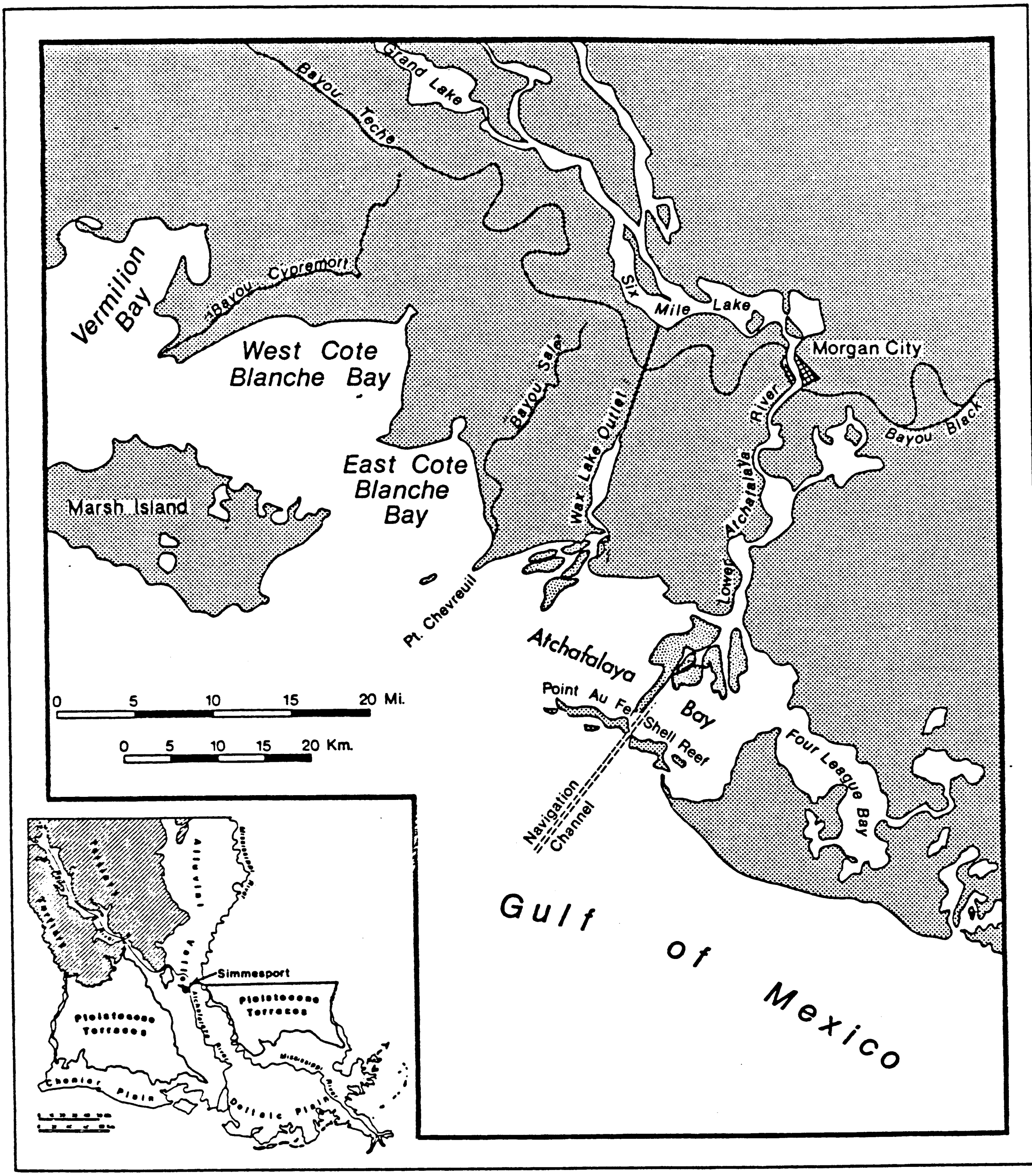

Figure 2. Regional setting, Atchafalaya Bay.

insights into the validity of ecosystem evolutionary concepts and coastal dynamic processes. In turn, a better understanding of processes results in an enchanced capability to assess the impacts of various management alternatives on the creation and restoration of wetlands.

The general objective of this study was to determine the potential of the evolving Atchafalaya delta to mitigate wetland loss. Specifically, the study was designed to (1) assess the net gain or loss of wetlands in the area influenced by the lower Atchafalaya River, (2) assess the conditions favorable for sediment transport to the adjacent marshes, and (3) broadly assess the probable impacts of a navigation and levee system proposal on wetland restoration.

AREA DESCRIPTION

General

The Atchafalaya River system begins at its disjunction with the Mississippi River near Simmesport, Louisiana (Figure 2). Before entering Atchafalaya Bay, the river flows for 226 km through a broad basin containing large expanses of swamp and bottomland hardwood forests. The basin developed between the natural levees of the modern Mississippi and Lafourche-Mississippi systems on the east and the uplifted Pleistocene terraces and Teche-Mississippi levees on the west (see Figure 1 and 2). An artificial levee system designed to act as a floodway has reduced the width of the basin.

Below Morgan City (Figure 2), the vegetation grades from swamp forest to emergent marshes. These wetlands, as well as Atchafalaya Bay, were formerly the sites of active deltaic sedimentation including the Maringouin, Teche, and Lafourche systems (Figure 1). Until the early 1950s, it had been 1500 years since substantial sedimentation occurred in this area (Shlemon, 1975).

Geologic and Hydrologic Setting

The geologic events and cultural modifications that altered geologic processes leading to the development of the Atchafalaya delta have been partially described by several investigators (Thompson, 1956; Fisk, 1952; Shlemon, 1972, 1975; Cratsley, 1975), and have been summarized by Roberts et al. (1980) and Adams and Baumann (1980). The mechanism that resulted in the capture of Mississippi flow by the Atchafalaya River is part of the Mississippi deltaic plain development process described by Kolb and Van Lopik (1966) and Frazier (1967), and the reader who desires a more comprehensive review than that which follows is referred to this literature.

The Atchafalaya River has existed as a distributary of the Mississippi River since at least the 1500s (Fisk, 1952). Until the early 1950s sedimentation was largely confined to the area above Morgan City, although some clays were transported out to the Gulf of Mexico as early as the middle of the 19th century (Thompson, 1951). Initially, diversion was accelerated by channel dredging and the removal of log jams, leading to an increase in the amount of flow down the Atchafalaya River, which has a decided gradient advantage over the existing Mississippi River course. Fisk (1952) predicted that the flow would abandon the Mississippi River course for the Atchafalaya River by 1975. To prevent total capture, a control structure was completed in 1963 that limited diversion to 30% of the combined 1950 Mississippi River and Red River flow regime.

The basin fill by lacustrine deltas was accelerated by man through the construction of levees along the basin margins. Anastomotic drainage gradually gave way to a more hydraulically efficient, single, well-defined river channel, which periodically "improved" by dredging for navigation and for efficiency in passing flood waters through the basin. By the early 1950s, fine-grained sediment was being deposited in Atchafalaya Bay (Shlemon, 1972, 1975) and the development of a single river channel through the delta fill in the basin was essentially complete (Roberts et al., 1980).

By the end of the 1960s, the southernmost lakes in the basin above Morgan City were approaching a sediment-filled state and by 1972 coarse-grained (sand) sediments were forming shallow bars in Atchafalaya Bay (Roberts et al., 1980). Thus, by 1972, the locus of sedimentation had shifted from the basin to the bay and the stage was set for subaerial development of a new delta and the concomitant restoration of deteriorating peripheral marshes.

During the spring of 1973, record flows near the head of the Atchafalaya River (Simmesport) exceeded 19,800 cms. In comparison, the average flow from 1938 through 1972 was 5,126 cms (COE, 1974), and average annual peak flow was 12,121 cms (Roberts et al., 1980). Peak outflows during the flood of 1973 probably exceeded 28,000 cms because of the additional flow created by the opening of the Morganza Spillway (Adams and Baumann, 1980). Floods during 1974-75 were lower than in 1973 but remained abnormally high. Accompanying the high discharges during the 1973-75 period were unusually high concentrations of sediment transported as suspended loads. Peak loads exceeded 5×10^5 metric tons per day during all three floods and attained a maximum of 6×10^5 mt/d during May 1975 (Roberts et al., 1980).

The effects of these floods on land building in Atchafalaya Bay have been well documented. Increases in the subaerial extent of the bay delta have been monitored with the use of Landsat data (Rouse et al., 1978), and with the aid of detailed topographic and bathymetric surveys (Adams and Baumann, 1980; Roberts et al., 1980). The formation of islands through bifurcation in Atchafalaya Bay is discussed by van Heerden and Roberts (1980a, 1980b), and the factors affecting the colonization of vegetation on the newly formed islands have been examined by Johnson et al., (1981). This paper emphasizes the restoration aspects of previously established marshes as a part of the evolution of the Atchafalaya delta system.

MATERIALS AND METHODS

Definition of Study Area

The first priority of this study was to define the spatial dimensions of the area affected by Atchafalaya River sediment. Examination of several Landsat images depicting the extent of the turbidity plume of the river during the record 1973 flood revealed that the study area could potentially encompass several thousand square kilometers. In order to achieve a more workable situation, we loosely defined the study area as the wetlands extending eastward from Bayou Sale (Figure 2) to that area where no change in the wetland loss rate occurred during the time intervals examined (91 W longitude, Figure 3). The northern boundary is defined by bayous Black and Teche, which form a continuous ridge strongly controlling the circulation pattern in this area (Figure 2).

Land Loss and Gain Rates

The change in area of wetlands and open water through time and the maximum eastward effects of the Atchafalaya River on this change were determined with the use of high-altitude color infrared photography.

Individual film positive frames of NASA, flight request #0754 (Landus, EPA) October 1978, were enlarged to 1:24,000 by the method described in Adams et al., (1978). The scaled film positive was overlaid on the most recent edition of USGS 7½-minute (1:24,000) topographic maps. The changes in land and water were transferred directly onto the map. Land was considered to be those areas supporting emergent vegetation or those clearly above the high-water mark. Newly deposited and unvegetated spoil was included as land if the spoil was subaerial at the time the image was taken.

The land and water areas were determined by overlaying the map with a mylar rectangular-grid cell network. Each 7½-minute topographic map contained 32,400 cells, 0.518 ha in size, and each map was divided into 81 data blocks of 400 cells, each on which the statistical analyses were based.

After the 1978 inventory of land and water areas was completed, the same procedure was employed for analyzing the 1972 land-water distribution, using NASA, mission 194, color infrared photography. The year 1972 was used because it was the last year before the series of major annual floods.

In order to determine the trend in land loss or gain prior to 1972, the data generated by Wicker (1980) and Wicker et al. (1980) were utilized. In Wicker's studies the habitat distribution in 1955-56 and 1978 were mapped and digitized. The habitats delineated by Wicker (over 100) were reduced to land and water, and the 1978 analysis was compared with our results. Statistical comparison revealed no significant difference in results, justifying the incorporation of the 1955-56 land-water ratios as determined by Wicker et al., (1980). This also substantiated the validity of the grid cell technique.

We assumed that any reversal in land loss was related to an increased input of sediment to the wetlands associated with the abnormal floods of 1973, 1974, and 1975. Since these events preceded this study, direct monitoring of sediment input was not possible; however, Landsat imagery provided adequate temporal coverage of this period.

Some 64 Landsat scenes covering 1973-1980 were borrowed from the U. S. Army Corps of Engineers, New Orleans District. The scenes are representative of various environmental conditions (weather, river stage, tides) that occur in the area. Adequate coverage during the spring flood season was crucial, and 23 of the scenes were taken during the flood stage of the Atchafalaya River.

The spatial extent of highly turbid water depicted on the scenes was qualitatively assessed[1] and transferred to a map base. The tidal stage, river discharge, and wind speed and direction were obtained for comparison (COE 1973-1980; DOC 1973-1980). After repeating this process for all scenes, a composite map of the extent of the turbidity plume during various flood conditions was generated.

The frequencies of wind direction, wind speed, and river discharge since 1972 were compiled from these sources in order to approximate the extent of the turbidity plume during all time intervals. By combining the results of this exercise with the composite map effort, the percentage of time a specific marsh area was inudated with turbid Atchafalaya

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[1]A companion study by Robert Hughes, Department of Geography, Louisiana State University, Baton Rouge, La. 70803, is currently obtaining data on sediment concentrations in the water column concurrent with Landsat overpasses and quantitatively analyzing turbidity.

River water was determined. This, in turn, was compared with results of the land loss (gain) analysis to determine relationships between the change in land-water ratio and the inundation of turbid water.

RESULTS AND DISCUSSION

Change in Wetland Area

Before the emergence of the Atchafalaya delta and the concurrent floods of the early 1979s, existing wetlands peripheral to the lower Atchafalaya River were deteriorating rapidly. All of the quadrangle mapping units examined had a net wetland loss during the period 1955-1972 (Figure 3). The eastern limit of net gain or reduced loss during the 1972-1978 period was approximately 91°W longitude. Total loss for the earlier 17-year period in quadrangles west of 91°W was 7,805 ha (488 ha/yr). In contrast, the 1972-1978 interval was characterized by a much reduced rate of land loss and, in fact, a net wetland gain occurred in several quadrangles. Landsat imagery provided visual evidence of an apparent limit to sediment influx; turbid Atchafalaya River water was observed in natural channels east of 91°W on only three frames. This occurred during the floods of 1973 and 1975 following cold-front passages (northwest winds).

Generally, the quadrangles closest to the source of sediment experienced greatest wetland gains. Those farther away experienced wetland losses, but the trend is highly variable. Loss in one quadrangle greatly accelerated (Figure 3). This area is composed of flotant, a highly organic floating fresh marsh (Russell, 1942). Chabreck (1981) attributes most of the loss to floods of abnormal height and duration that occurred in 1973-1975. According to Chabreck, extensive mats of water hyacinth (*Eichhornia crassipes*) floated into the area during the flood and were

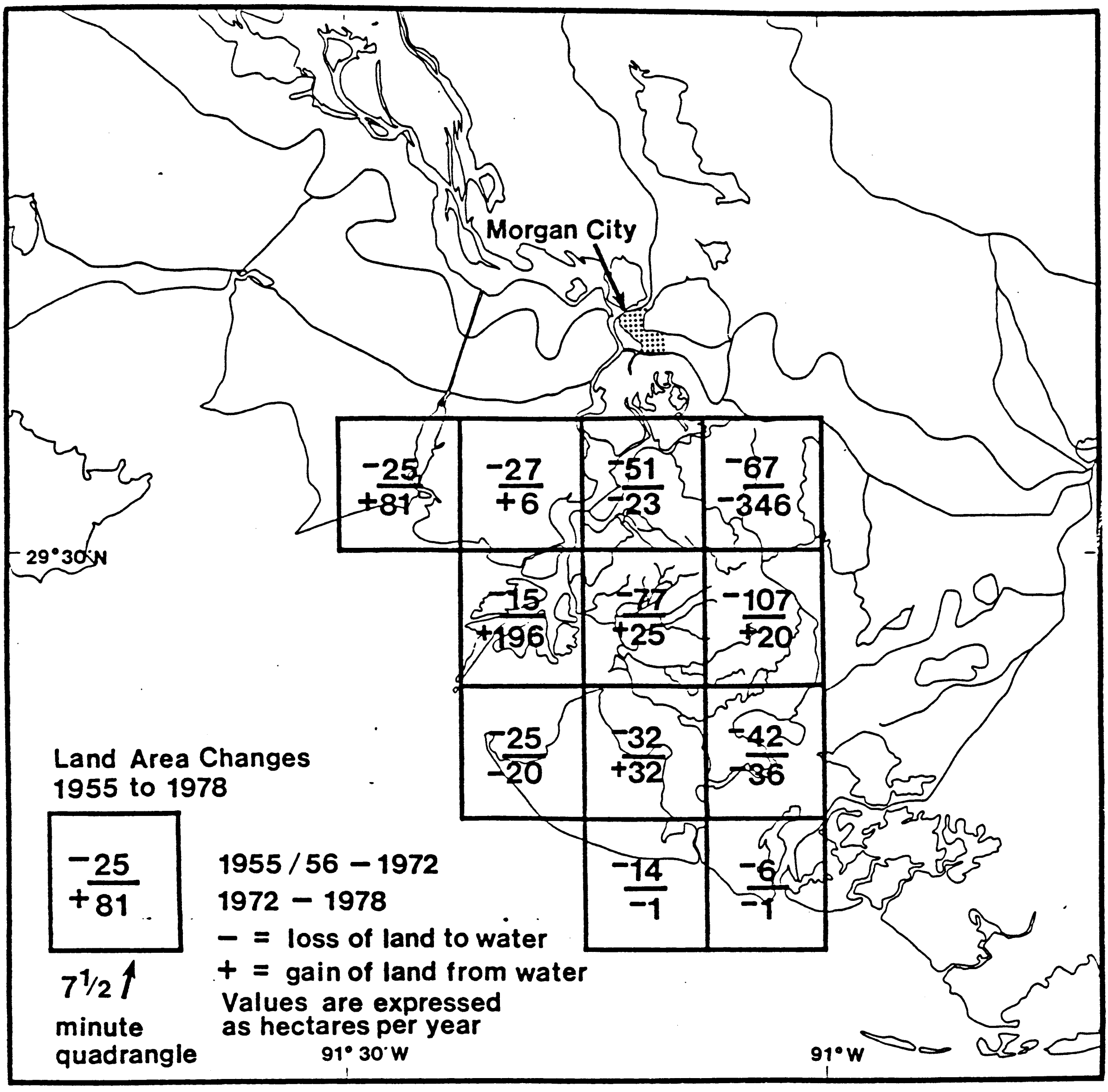

Figure 3. Land area changes for the study area, 1955 to 1978.

deposited on the marsh surface when floodwaters subsided. Their decay killed large areas of maidencane (Panicum hemitomon) and alligator weed (Alternathera philoxeroides), leaving open water areas that subsequently were invaded by subclimax species such as smooth beggertick (Bidens laevis) (Kinler et al., 1981). The increased loss in this much appears to be only a temporary phenomenon as the area is presently recovering to a climax stage (Kinler et al., 1981).

Most of the nonflotant marsh area experienced a slower rate of wetland loss. Overall, the nonflotant marshes west of 91° W experienced a net gain from 1972-1978 of 1,676 ha (279 ha/yr) with 1,277 ha (213 ha/yr) attributable to the formation of new wetlands on the lower Atchafalaya River and Wax Lake Outlets deltas (Figure 3). In comparison, the same area lost a total of 6,736 ha of wetlands from 1955 to 1972 (421 ha/yr). Thus, marshes peripheral to the new delta experienced a reversal from 421 ha/yr loss to 66 ha/yr gain. Most of this reversal occurred in the area to the east of the proposed Avoca Island levee extension (Figure 4).

Factors Regulating Wetland Restoration

The most important factors that were considered to regulate the spatial extent of wetland restoration are river discharge, location of the wetland parcel, wind direction, and tide stage. For the marshes between Wax Lake Outlet and the lower Atchafalaya River, discharge is the overriding factor. Landsat imagery shows that sediment-laden water floods over these wetlands during flows of 11,000 cms or greater, regardless of wind conditions. In comparison, the average annual peak flow is 12,121 cms (Roberts et al., 1980). Wind direction (easterly versus westerly components) has little effect because these marshes lie between the two Atchafalaya River outlets, lower Atchafalaya River and Wax Lake Outlet.

In the marshes east of the Atchafalaya River, wind direction has a pronounced effect on the eastward extent of the Atchafalaya River plume. Turbid Atchafalaya River water covers at least 90 percent of the marsh

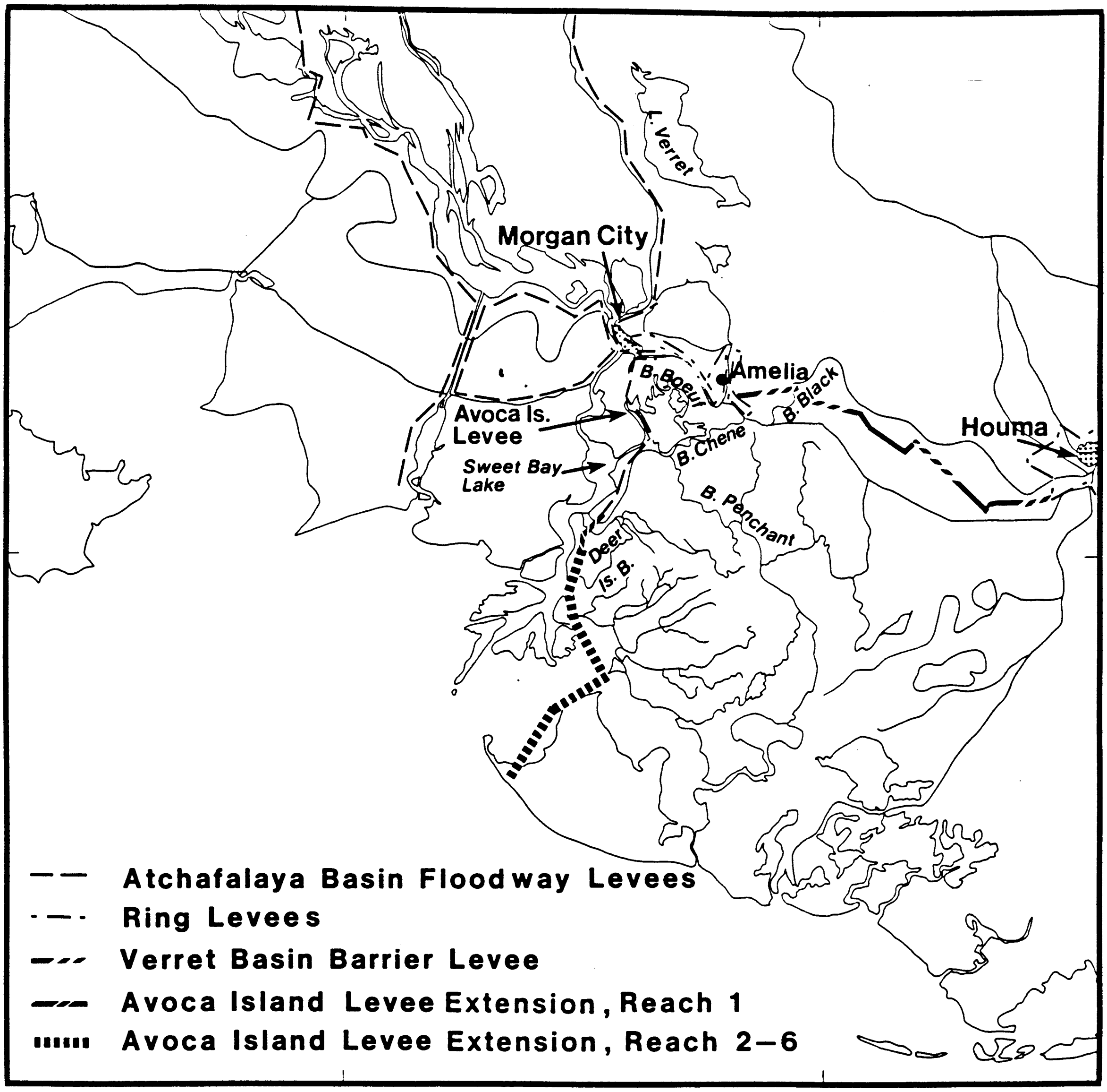

Figure 4. Present and proposed levee alignments in the lower Atchafalaya Basin, lower Atchafalaya River, and Lake Verret flood basin.

area west of 91°W when flows are 15,400 cms or greater and winds are from the southeast at seven knots. The same areal coverage occurs during flows of 12,350 cms when the wind is from the northwest at eight knots. On the basis of wind direction and river discharge frequencies from 1971-1980, it is estimated that 90 percent coverage occurs approximately six percent of the time. This value may overestimate the frequency of marsh flooding over a longer period because floods during the 1970s were exceptionally high and long.

Probable Impacts of the Avoca Island Levee Extension

The existing Avoca Island levee (Figure 4) is part of the lower Atchafalaya basin floodway system and is designed to curb backwater flooding in the Lake Verret swamp basin north and east of Morgan City. The flooding of the marshes that border this swamp basin on the south does not have a major direct cultural impact because urban development is sparse in the immediate area. Raising water levels over these marshes, however, reduces the hydraulic gradient of drainage from the developed Lake Verret watershed lying to the north (Figure 4). Because the Atchafalaya River is lengthening its course via delta development, the river's gradient has decreased, resulting in higher stages with the same discharge. This, in turn, aggravates the backwater flooding problems of the Lake Verret watershed. As a partial solution to these flooding problems, the Army Corps of Engineers, New Orleans District, has proposed a 5-km extension of the Avoca Island levee (Figure 4, reach 1). In the future, further extensions may be considered (Figure 4, reaches 2-6).

One result of the proposed extension would be to cut off the present junction of Bayou Chene and the Atchafalaya River and relocate it 5 km downstream (Figure 4). The Bayou Chene-Penshant system (Figure 4) is the major avenue for sediment and water transport from the Atchafalaya River to fresh marshes as well as an important marine transportation corridor. Moving the junction downstream will reduce the flows from the Atchafalaya River into these marshes because of the lower river stages at the new location.

Water level records taken by the Corps of Engineers during 1974-79 at gauges located at Morgan City, Sweet Bay Lake, and Deer Island Bayou (Figure 4) provide a data base for estimating how much lower the stages will be at the new location. Mean monthly water surface slopes from 1974 to 1979 ranged from 4.55 cm/km to -0.74 cm/km. The negative values normally occur during the fall months when discharge is low and a seasonal change in wind stress (Sturges and Blaha, 1976) causes a net landward transport of Gulf water. The highest positive slopes are associated with high discharge during the spring months. The mean flood season slope for 1974, 1975, and 1979 was 2.21 cm/km. Thus, we can infer that is the levee extension had existed during the 1970s it would have resulted in a mean stage reduction at the proposed junction of Bayou Chene and the Atchafalaya River of 11 cm (2.2 x 5 km) during the flood season and a maximum reduction of 22.8 cm (4.55 x 5 km).

Reduced flows into the Chene-Penchant system could cause a moderate reduction in sediment transport and rate of wetland restoration, but this impact is considered by the authors to be potentially much smaller than another effect of the levee extension. As a part of the proposed flood protection plan, an inflatable dam would be situated at the new junction of Bayou Chene and the Atchafalaya River when reach 2 (COE, 1981) is implemented. During high flows the dam would maintain water levels at Amelia (Figure 4) below 0.91 m (3.0 ft) National Geodetic Vertical Datum (NGVD) (COE, 1981), thus restricting the flow when it is most needed for wetland restoration. In addition, because regional subsidence is increasing the potential for flooding in the Verret watershed, closure time for the inflatable dam will also increase, thus shutting off the sediment supply to the peripheral marshes and impeding navigation activity in the Bayou Chene-Black transportation corridor. These conclusions are based on the results of the following analysis of water level trends at Amelia since 1955.

There is a positive correlation between water level at Amelia and Atchafalaya River discharge. This implies a relationship between Atchafalaya River discharge and occurrence of flooding at Amelia. Liner regression analysis, using water level at Amelia as the dependent variable, yields the following equation:

$$WL = 7.4 + .01\ Q$$

$r^2 = .73$ ($P < .01$) and the slope is significantly greater than zero ($P < .01$)

where: WL = annual mean water level at Amelia from 1955-1980, expressed in cm NGVD, based on daily 8:00 a.m. readings (COE, 1955-80);

Q = annual mean discharge at Simmesport from 1955-1980 expressed in cms based on daily calculations (COE, 1955-80).

The residuals (observed error) were then plotted against time in order to determine whether a linear trend existed (Figure 5). Projecting the trend to a hundred-year basis, mean annual water level at Amelia is rising independent of river discharge, at a rate of 0.85 m/century. This rate is within the range of apparent sea level rise (caused primarily by subsidence), indicated by water-level gauging stations to the east that are not influenced by major river flows (Baumann, 1980). Another indication that the apparent rise in water level at Amelia is related to subsidence is that a noticeable increase in the rate of rise

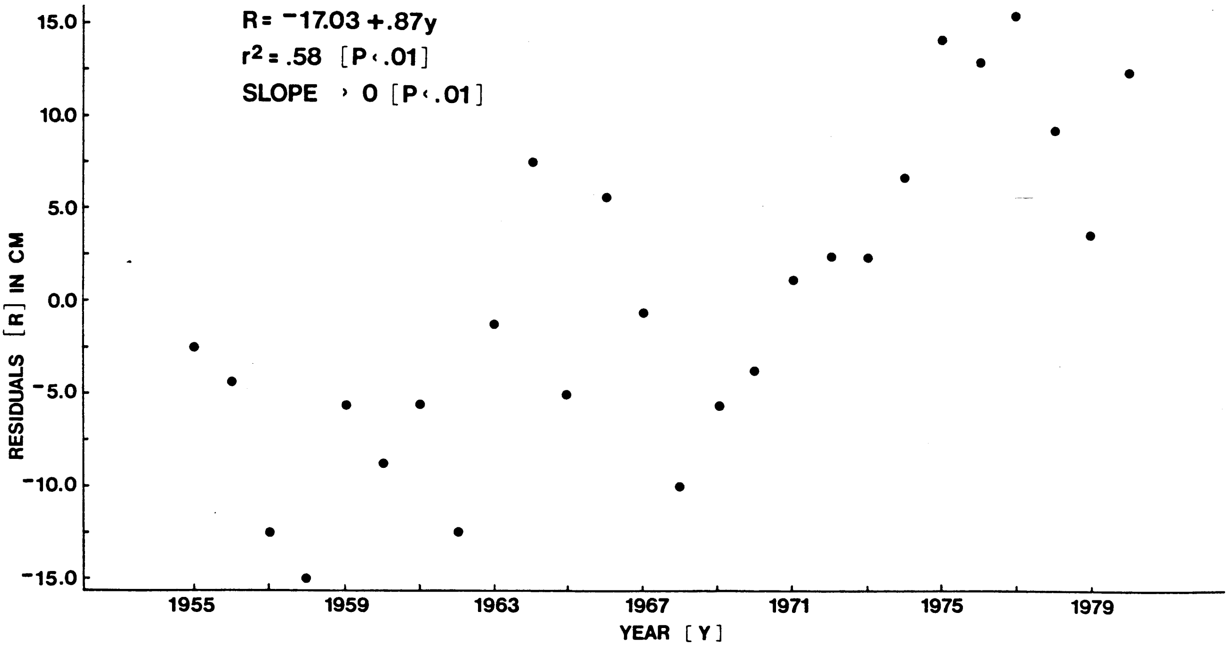

Figure 5. Residuals (observed error).

occurred during the late 1960s (Figure 5), a pattern also observed for the gauges to the east (Baumann, 1980). We cannot, however, totally rule out that the lengthening of the Atchafalaya River through time is causing some portion of the apparent 0.85 m/century rise in water level at Amelia.

Assuming that the 1955-80 linear trend in water level rise will continue, the average gauge reading at Amelia will be 0.91 m (3.0 ft) by the year 2018. Thus, the flooding of communities in the Lake Verret watershed, the closure of Bayou Chene to marine traffic, and the exclusion of Atchafalaya flow and sediment to the marshes will occur routinely under average water level conditions.

Mitigating measures should be taken before these conditions are reached. One option may include periodic extension of the Avoca Island levee, but this option will never totally resolve the flooding problem. Moreover, continued extension of Atchafalaya levees would eventually cause sediment needed for wetland creation and maintenance to be lost to deep offshore waters, a situation like that affecting the present Mississippi River delta.

Other options include ring levees around communities or a continuous levee from Morgan City to Houma (Figure 4), both of which would require the construction of pumping stations and perhaps navigation locks. The cost of these options would be high, but they would resolve current flooding hazards, (backwater, headwater, and tidal) and would permit the continuance of wetland creation and restoration.

The only feasible means of combatting subsidence in wetlands is through addition of sediment to the marsh surface. If Lousisana's coastal wetlands and their renewable resources are to be maintained, flood control and navigation projects must enhance, rather than impede, the deposition of inorganic sediment in these wetlands. If this objective is not actively pursued, we will continue to experience loss of wetlands. At the current statewide rate of 128 km^2/yr, this loss would equal an area the size of the state of Connecticut in less than a century.

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ASSESSING ECOLOGICAL VALUE OF CENTRAL FLORIDA WETLANDS:

A CASE STUDY

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ABSTRACT

A recently developed wetland valuation methodology was used to assess the relative ecological value of 819 freshwater wetlands in central peninsular Florida. Of these wetlands, 717 were located on Mississippi Chemical Corporation's 15,000-acre proposed phosphate mine-site in Hardee County. The remaining 102 wetlands were distributed throughout Polk, Hillsborough, Highlands, and Hardee Counties and included some of the largest and most significant wetlands in the region.

The purpose of the study was to numerically rank wetlands on the MCC property based on their ecological value and to compare these ranks with those of other significant wetlands in the region. Based on this information, MCC and the various regulatory agencies could then reach a consensus as to which wetlands could be mined, which could be mined only if they were later restored, and which must be preserved.

Wetland functions evaluated by the methodology include water quality enchancement, water detention, vegetative diversity, vegetative productivity, and wildlife habitat value. Application of the methodology showed that most of the wetlands on the MCC tract had a low to moderate ecological value when compared to regional wetlands. This was due to the small size of the MCC wetlands, their lack of contiguity with large perennial creeks or rivers, and their less than maximum structural vegetative diversity. After careful examination, the regulatory agencies accepted the study results, and a consensus was reached as to which wetlands were mineable.

INTRODUCTION

In February 1977 Mississippi Chemical Corporation (MCC) filed an Application for Development Approval for a 14,850-acre phosphate strip mine to be located in western Hardee County, Florida. Although a Development Order was issued by Hardee County eleven months later, granting MCC permission to proceed with its planned mine, the Florida Division of State Planning (DSP) subsequently filed a petition appealing the Development Order. This effectively precluded startup of the proposed mine until the issues listed in the DSP appeal could be satisfactorily resolved. Mining versus preservation and/or restoration of wetlands was one of the major issues.

Although DSP was concerned for all the on-site wetlands, greatest attention was given to the 72 bayheads on the tract and to the various wetlands associated with Hickory Creek (a small creek on the eastern side of the tract). In order to facilitate the determination of which wetlands were of greater ecological value, it was agreed that a valuation methodology should be developed and then applied to each of the wetlands on the MCC property. Providing that the design of the methodology was acceptable to all parties involved, the results could provide valuable aid in determining which wetlands should or should not be mined.

In order to compare the MCC wetlands with other regional wetlands of high ecologic value, 10 wetland systems (containing 102 individual wetlands) were also examined. These wetland systems were distributed throughout Polk, Hillsborough, Highlands, and Hardee Counties and included some of the largest and most significant wetlands in the region. Individual wetland size ranged from 0.3 to 1,607.9 acres, with 13 wetlands being larger than 100 acres and three wetlands being larger than

1,000 acres. Wetland types included marshes, shrub swamps, bayheads, cypress swamps, and mixed hardwood floodplain forests.

AREA DESCRIPTION

The MCC site contains a total of 717 wetlands, classified as either bayheads, shrub swamps, or fresh marshes. The bayheads are characterized by a diverse flora: the dominant overstory species being *Acer rubrun*, *Magnolia virginiana*, *Nyssa biflora*, *Quercus laurifolia*, *Ulmus americana*, and *Liquidambar styraciflua*. Shrub and ground cover strata are typically well developed, including species such as *Myrica cerifera*, *Itea virginica*, *Woodwardia areolata*, *Osmunda cinnamomea* and *O. regalis*, *Rhynchospora mileacea*, and *Saururus cernuus*. Shrub swamps usually contain dense stands of either *Cephalanthus occidentalis*, young *Acer rubrum*, or *Salix caroliniana*, and often adjoin either bayheads or fresh marshes. The species composition of fresh marshes is quite variable, commonly including *Panicum hemitomon* and *P. portoricense*, *Paspalum dissectum*, *Polygonum punctatum*, *Lindernia grandiflora*, *Cyperus polystachyos*, *Juncus effusus*, *Pontedaria cordata*, *Psilocarya nitens*, *Rhynchospora careyana*, *Centella asiatica*, *Sagittaria lancifolia*, *Spartina bakerii*, and various other species. A more detailed description of MCC's wetland flora is presented in MCC (1977) and Dames and Moore (1981). Additional data on these wetlands are summarized in Table 1.

TABLE 1
Wetland Data for Mississippi Chemical Corporation Property

	Wetland Type		
	Bayhead	Shrub Swamp	Fresh Marsh
No. of Wetlands on MCC Tract	72	36	609
Total Acreage	489.5	171.7	2,317.6
Average Acreage	6.80	4.77	3.81
Acreage Range	0.2-63.7	0.2-16.4	0.1-112.5
Average Acreage Class (V_1)	2.36	1.91	1.52
Average Contiguity (V_2)	3.26	3.52	1.95
Average Structural Diversity (V_3)	3.96	2.92	1.20
Average CEI (V_4)	3.52	3.40	2.61
Average IWV	13.00	11.53	6.71

MATERIALS AND METHODS

An in-depth description of the evluation methodology used on the MCC and regional wetlands is beyond the scope of the present paper. The original conceptual approach is discussed by Winchester and Harris (1979) and is followed by a detailed analysis of the methodology by Winchester (1981). However, the following simplified description is presented to acquaint the reader with the basics of the methodology.

Wetland functions or attributes evaluated by the methodology are water quality enhancement, water detention, vegetative productivity and diversity, and wildlife habitat value. These functions are assessed by measuring the following four wetland parameters: wetland size (V_1), wetland contiguity (V_2), structural diversity (V_3), and a comprehensive edge index or "CEI" (V_4). Each parameter is ranked between 1 and 10, with higher ranks being given to larger wetlands, to wetlands with greater contiguity to the regional hydrologic system, to more structurally diverse wetlands (either greater number of strata or greater horizontal zonation), and to wetlands with higher quality edges and greater edge to area ratios. Measurement of the first three parameters is described on Figure 1 and Tables 2 and 3. The edge to area ratio was calculated using the formula:

$$CEI = \frac{E_W}{2\sqrt{a}}$$

where CEI is the "comprehensive edge index," E_W is the sum of each wetland's weighted edge lengths, and a is the area of the wetland. Individual edges were weighted according to the gross primary productivity and the degree of structural relief or "edge drama" between the two adjoining plant communities.

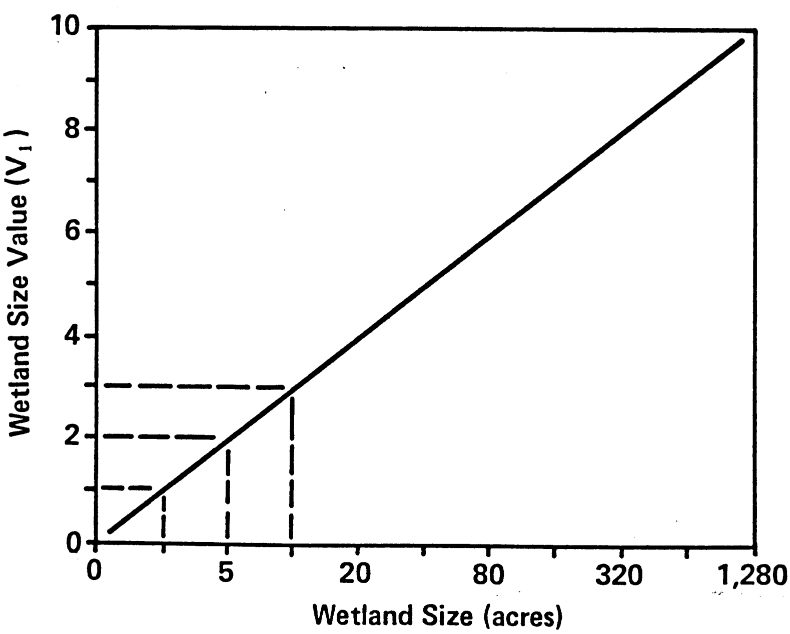

Figure 1. Wetland size and corresponding wetland size values (V_1).

TABLE 2
Determination of Contiguity Value (V_2) for Wetlands with Various Hydrologic Connections

	Contiguity Value (V_2)	
	Intermittent Systems	Perennial Systems
Isolated Wetland	1	1
Minor Ditched or Natural Connection	2	4
Major Ditched/Channelized Connection	3	5
Distinct Natural Connection or Adjoining Tributary with 5 cfs Flow	4	6
Adjoining Tributary with 5 cfs Flow	6	8
Adjoining Creek/River with 100 cfs Flow	-	10

TABLE 3
Determination of Structural Diversity Values (V_3) For Various Types of Wetlands

V_3	Number of Strata		Degree of Zonation
2	1		Little or None
4	25% of Area with 2 Strata	or	3 or More Zones (1 Strata)
6	2 Strata Throuhout		Absent
8	3 Strata Throughout		Absent
10	3 Strata Throughout	and	2 or More Zones

After the four parameter values have been determined, they are then added to obtain a numerical rank (the "individual wetland value" or "IWV") for each wetland. Since wetland size is given equal weight to the other three parameters combined, the equation for computing IWV is

$$IWV = 3V_1 + V_2 + V_3 + V_4.$$

In the present application, the IWV was adjusted to a maximum of 40

Table 4
Individual Wetland Parameters for 30 Highest-Ranked Wetlands on the Mississippi Chemical Corporation Property, Hardee County, Florida

Wetland Rank	Wetland Type	Acreage	Acreage Class	Continguity Class	Structural Diversity	Comprehensive Edge Index	IWV
1	Marsh	112.5	6.41	6	4	5.89	23.41
2	Bayhead	56.7	5.42	6	8	4.44	23.13
3	Bayhead	63.7	5.59	4	8	3.86	21.76
4	Bayhead	40.3	5.01	4	8	5.14	21.44
5	Marsh	49.6	5.24	6	4	5.91	21.09
6	Marsh	61.5	5.54	6	4	3.95	20.38
7	Marsh	28.2	4.41	6	4	7.09	20.21
8	Marsh	71.9	5.80	2	4	6.04	19.62
9	Marsh	35.4	4.77	4	4	7.02	19.56
10	Marsh	35.1	4.76	4	4	6.64	19.27
11	Marsh	42.0	5.05	4	4	4.75	18.60
12	Marsh	32.0	4.60	6	4	3.09	17.93
13	Bayhead	22.4	4.12	2	8	4.44	17.87
14	Marsh	21.4	4.07	4	4	6.42	17.75
15	Marsh	34.2	4.71	4	4	4.27	17.60
16	Marsh	24.4	4.22	4	2	7.52	17.45
17	Bayhead	22.1	4.11	2	8	3.66	17.31
18	Bayhead	13.0	3.30	4	8	3.84	17.16
19	Bayhead	12.1	3.21	4	8	4.05	17.12
20	Marsh	22.2	4.11	6	2	5.21	17.03
21	Marsh	21.9	4.10	6	4	3.21	17.00
22	Marsh	21.9	4.10	2	4	6.79	16.71
23	Bayhead	14.8	3.48	3	8	3.61	16.70
24	Bayhead	14.0	3.40	4	8	2.85	16.70
25	Bayhead	13.1	3.31	4	8	3.09	16.68
26	Bayhead	13.0	3.30	4	8	3.02	16.61
27	Bayhead	11.2	3.12	4	8	3.46	16.55
28	Shrub Swamp	16.4	3.64	4	6	3.65	16.38
29	Bayhead	8.6	2.72	4	8	4.40	16.37
30	Marsh	40.2	5.00	1	4	4.50	16.34

(rather than 60) to make it comparable with previous wetland analysis using the unmodified Winchester and Harris (1979) technique; consequently, the equation was:

$$IWV = 2/3\ (3V_1 + V_2 + V_3 + V_4).$$

RESULTS

Individual wetland values (IWV's) on the MCC tract ranged from 3.27 to 23.41. Of the 717 wetlands, seven (1.0 percent) had IWV's of 20 to 25. Thirty-five (4.9 percent) of the wetlands had IWV's ranging from 15 to 20, and 123 (17.2 percent) and IWV's ranging from 10 to 15. The remaining 552 wetlands all had IWV's less than 10, accounting for approximately 77 percent of the wetlands on the property. Average parameter values for each wetland type are given in Table 1. Individual wetland parameters for MCC's 30 highest ranking wetlands are given in Table 4.

The highest IVW encountered in the valuation of regional wetlands was 38.67, just a little below the possible maximum of 40. The wetland attaining this rank was a large (1,607.0-acre) swamp system containing cypress, bayhead, and palm-hardwood hydric hammock components. The various parameter measurements for the 13 largest (acreages 100) and highest ranking regional wetlands are presented in Table 5.

DISCUSSION

On a scale of 1 to 40, wetlands on the MCC property had individual wetland values (IVW's) of 3.27 to 23.41. On the same scale, selected wetlands from adjoining counties ranked as high as 38.67. Based on these rankings, the most ecologically valuable wetlands on the MCC property could be considered moderate in value when compared to high value regional wetlands.

Table 5
Individual Wetland Parameters for High-Ranking Wetlands in
Polk, Hillsborough, Hardee, and Highlands Counties

Wetland Rank	Wetland Type	Acreage	Acreage Class	Contінguity Class	Structural Diversity	Comprehensive Edge Index	IWV
1	Mixed Swamp	1,607.9	10.0	8	10	10.00	38.67
2	Mixed Swamp	1,059.8	9.66	10	10	8.50	38.31
3	River Swamp	617.6	8.93	10	8	5.23	33.35
4	Mixed Swamp	245.8	7.54	10	10	7.28	33.26
5	Mixed Swamp	538.7	8.68	8	10	5.46	33.01
6	Shrub Swamp	700.4	9.09	10	6	6.19	32.98
7	Marsh	1,471.9	10.0	4	4	8.69	31.13
8	Mixed Swamp	399.2	8.25	8	10	3.57	30.88
9	Mixed Swamp	509.0	8.59	4	10	5.22	30.00
10	Mixed Swamp	313.9	7.96	6	10	4.52	29.60
11	Marsh	982.0	9.53	5	4	6.47	29.38
12	River Swamp	121.5	6.52	10	8	5.14	28.46
13	Marsh	120.7	6.51	8	4	6.77	25.53

A number of factors contributed to the moderate IWV's encountered on the MCC property. First, MCC wetlands were generally small, the largest being only 112.5 acres and having a V_1 of 6.41. In contract, a number of regional wetlands had acreages above 500, and two were of sufficient size to get the maximum V_1 (10.0). While many of the regional wetlands were directly adjoining major, perennial water bodies, the MCC wetlands were associated only with intermittent stream systems and could only achieve a maximum V_2 of 6.0. Although the vertical structural diversity (number of strata) was similar for both MCC and regional forested wetlands, horizontal structural diversity (zonation) was higher for the regional swamps due to the presence of distinct vegetative components (e.g., cypress, bayhead, hydric hammock) within individual swamps. The highest comprehensive edge index values (V_4) for MCC and regional wetlands were 7.52 and 10.0 respectively.

In regard to the regional wetlands, it should be noted that the wetland systems evaluated did not include all the major systems in the region. A number of systems could not be included in the evaluation due to problems in obtaining access permission from landowners. Also, some floodplain forest systems were artifically truncated during the valuation process for the same reason. The highest IWV's theoretically possible for marsh, shrub swamp, and forested wetland systems are 34.0, 36.0 and 40.0. The maximum IWV's actually encountered in the regional valuation were 31.13, 32.98, and 38.67, respectively. Although it is possible that some of the regional wetlands which were not evaluated may have higher IWV's, the IWV's actually encountered are belived to be relatively close to the maximum for the region.

As mentioned earlier, the 72 bayheads on the MCC property were a chief concern of DSP, especially the bayheads associated with Hickory Creek. Based on their IWV's, the two highest ranking bayheads along Hickory Creek occurred in 19th and 27th place (see Table 4), indicating they were equivalent in value to a number of other wetlands on the MCC property. Consequently, the Hickory Creek wetlands were no longer given priority over the other similarly-valued wetlands on the site. Ultimately, the two bayheads were removed from DSP's "preservation" category and placed in the "mining with later replacement" category. The remaining 72 bayheads had IWV's varying from 23.13 (rank no. 2) to 6.71 (rank no. 337). However, the majority (65 percent) of the bayheads had IWV's falling between 10.0 and 16.0. In contrast, most (64 percent) of the MCC marshes had IWV's between 2.0 and 8.0.

After careful examination of the data, DSP and MCC reached the following agreement as to the wetlands on the tract:

1. The three highest ranking wetlands (a marsh and two bayheads with IWV's of 23.41, 23.13, and 21.76 respectively) could be mined only after wetland creation had been shown to be feasible on the MCC tract. Their combined area amounted to 232.9 acres.
2. Wetlands with IWV's of 7.0 or below could be mined without replacement or left undisturbed according to the dictates of the mine plan. This represented 15 percent of the total wetland acreage (488 acres) and included 55 percent (395) of the 717 individual wetlands on the tract.
3. The intermediate 319 wetlands, comprising a total of almost 2,300 acres, could be mined and later replaced or left undisturbed. In this case, prior demonstration of the feasibility

of wetland restoration was not required.

4. Together, these three items guaranteed that 85 percent of the original wetland acreage would be present at the end of the mine life.

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A WATER QUALITY MONITORING PROGRAM
FOR MANAGEMENT AND RESTORATION
OF TWO LOUISIANA ESTUARIES

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ABSTRACT

Surface water in the Breton Sound and Barataria Bay extuaries are being monitored weekly for several physical, chemical, and biological parameters. Data over a 10-year period, showing seasonal trends throughout each estuary, as well as the effects of the Mississippi River which flows between them, confirms some and conflicts with other management policies. The influence of existing fresh water diversions on the estuaries is considered in the discussion of new structures proposed in a comprehensive restoration program.

INTRODUCTION

The coastal waters of southeastern Louisiana are highly productive and dynamically active, but increasing salinity over the past several years is adversely affecting water quality in this area (Gagliano, 1981; Gagliano et al. 1973; Fruge, 1980; Craig et al. 1979). Increased salinity along the Louisiana coast is due primarily to the highly effective flood protection levee system along the Mississippi River (Gagliano et al. 1973). Two estuaries particularly impacted by the construction and maintenance of the river levee system are the Breton Sound estuary and Barataria Bay. Breton Sound estuary lies east of the Mississippi River and south of Bayou Terre Aux Boeuf, while Barataria Bay lies to the west of the river (Figure1).

The Breton Sound estuary consists of two hundred and fifty thousand acres, of which area approximately one quarter is open water and the rest is tidal marsh.

Barataria Bay covers approximately four million acres, of which only the eastern portion between the Mississippi River and the boundary between Plaquemines and Jefferson Parishes will be considered here. This portion consists of one-third open water and two-third marsh.

Marsh types in both estuaries within the boundaries of Paquemines Parish are predominately brackish, and range from saline to intermediate. Intermediate marsh accounts for five percent or less of the area in Breton Sound estuary and only slightly more in Barataria Bay (Fruge, 1980).

Both estuaries are in the modern deltaic region of the Mississippi River. Most of this region, except that near the mouth of the river, is in a pronounced state of degradation (Gagliano, 1981). The most prominant aspect of the degradation phenomena in these estuaries is

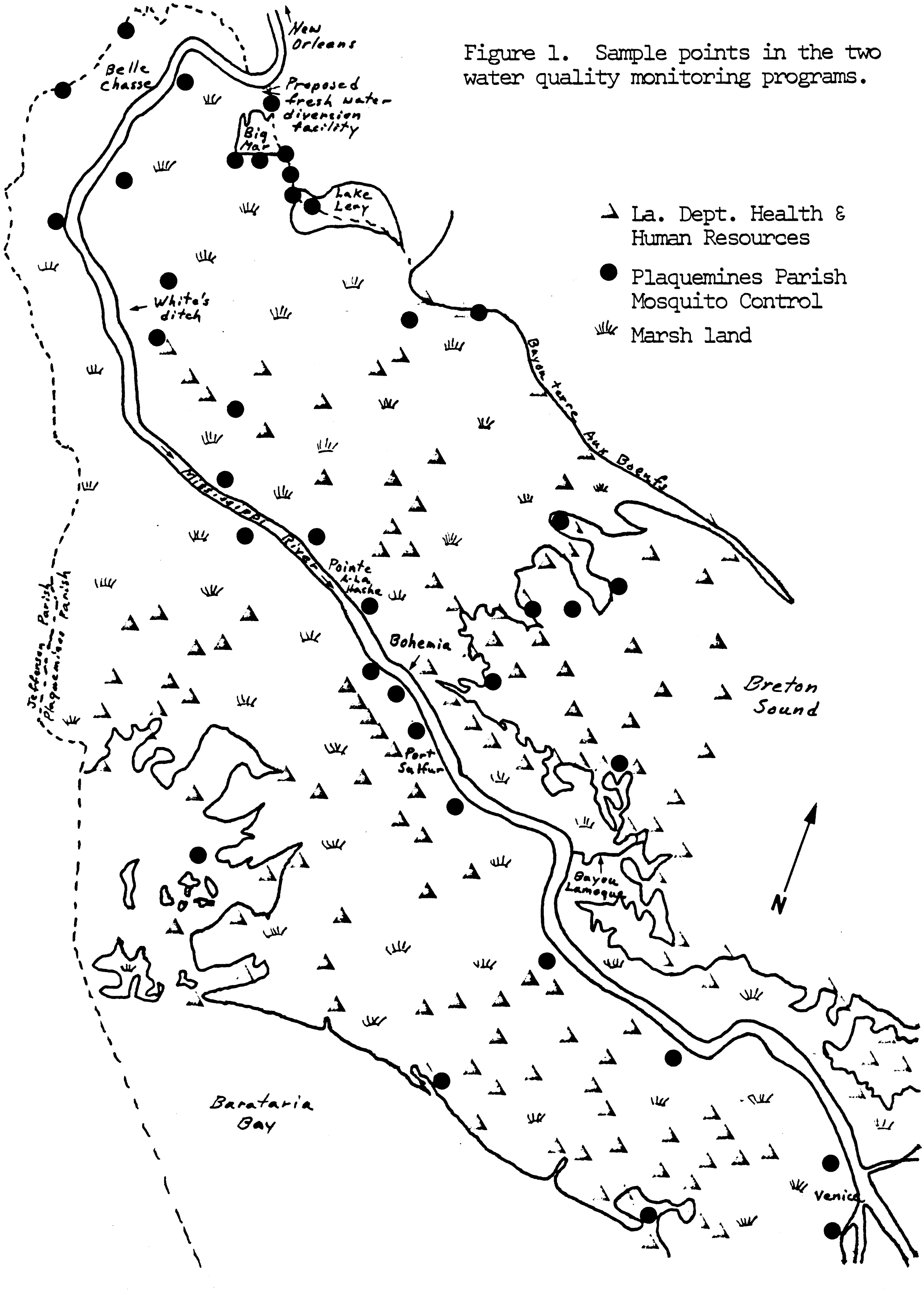

Figure 1. Sample points in the two water quality monitoring programs.

land subsidence (Gagliano, 1981; Craig et al. 1979) which is contributing significantly to an average loss of at least 13 square miles of land per year in these two estuaries (After Ader, personal communication).

A secondary aspect of the degradation phenomena is the encroachment of ecologically less productive brackish marsh into more productive intermediate and fresh marshes (Fruge, 1980). Now, in Plaquemines Parish very little area remains in intermediate marsh, and fresh marshes have all but dissappeared in both estuaries. Also, increased salinities in the open waters of the estuaries have been accompanied by invasion of commercial oyster grounds by the oyster drill (Thais haemostoma), other marine predators, and disease organisms (Van Sickle et al.; Pollard, 1973; Briethaupt and Dugas, 1979).

Combined effects of land loss and habitat deterioration are gradually reducing the size and quality of the nursery grounds in both estuaries. These debilitating effects on estuarine productivity are discussed by Frug (1980). Craig et al. (1979 estimates 8-17 million dollars of fishery products and services are lost annually in Louisiana because of deterioration of the estuaries. Prospects for the future productivity of these estuaries appear dim if present trends continue.

MATERIALS AND METHODS

In an attempt to counteract the invasion of oyster predators, three structures were constructed in Breton Sound between 1960 and 1975. The structures divert fresh water from the Mississippi River to "sweeten" the adjacent estuarine areas. These structures include a syphon at white's ditch and two gravity flow, gated structures at

Bohemis and Bayou Lamoque (Figure 1). The syphon at White's ditch consists of twin 50-inch-diameter pipes laid over the top of the levee with a design capacity of 250 cubic feet per second (cfs). The Bohemia structure is in the same size class as the syphon at White's ditch. The Bayou Lamoque structure consists of two sets of gates. The older set of gates delivers up to 4,000 cfs. Flow through the more recently constructed set of gates amounts to 6,000cfs. Combine maximum flow for the two sets of gates is 10,000 cfs (Cunningham, personal communication).

In 1960, anticipating problems with the quality of diverted fresh water, the Louisiana Department of Health & Human Resources (DHHR) began a comprehensive, and still on-going, program of routine inspections of water through these and other estuaries along the Louisiana coast.[1] The purpose of this program is to identify waters in which oysters might be contaminated with organisms that are pathogenic to humans.

Data collected by DHHR includes depth at which each water sample of 300 ml is taken in a sterile glass bottle, a record of the tide stage when the sample is taken, and air and water temperature on site. The water sample is transported in ice for laboratory determinations of pH, turbidity, salinity, and total and fecal coliform by the multiple tube procedure. Eighty sample points are located in the Breton Sound estuary, and fifty-seven are located in Barataria Bay between the Mississippi River and the boundary between Plaquemines and Jefferson Parishes (Figure 1). DHHR's data is housed in STORET.

[1] DHHR and MC have a cooperative agreement for the exchange of data.

Plaquemines Parish Mosquito Control (MC) established a water quality monitoring program in January, 1980, with thirty-eight stations, of which some are identical with DHHR's for purposes of comparison (Figure 1). At each location, we record water temperature, dissolved oxygen, specific conductivity, and salinity _in situ_ with a Mark VIII automatic recording device[2] and collect a 200 ml water sample in a sterile bottle for laboratory determinations of pH, turbidity, and fecal coliform by the membrane filter technique. The purpose of the MC program is to establish a data base from which decisions can be made for managing and protecting the estuaries. Storage and management of MC data, and DHHR data as it is received, is planned for the Parish computer. Analyses of the data from both programs provide base line information of hydrological and biological importance for both estuaries.

RESULTS AND DISCUSSION

Temporal and spatial trends in the data will be reported elsewhere. This report will focus on another interesting aspect.

Within a few seasons after the first set of gates were constructed at Bayou Lamoque, striking effects were apparent in the adjacent estuarine area. For example, salinities of 6 to 9 parts per thousand (ppt) off the mouth of the bayou in January and February, 1979, fell to 1 ppt or less in April and May as fresh water from the Mississippi River at high stage flowed into the bay (Table 1). By mid-June the river had fallen and salinites in the bay returned to more than 13 ppt and stayed between 8 and 15 ppt for the remainder of the year. The

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[2] Martek Instruments, Inc., Irvine, California

Table 1. Average monthly salinities in 1979 and fecal coliform colonies for 1969-1980 in California Bay off the mouth of Bayou Lamoque.

Item	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.	Nov.	Dec.
Salinity[1] (°/00)	6.72	8.82	4.53	1.3	0.77	13.45	8.44	14.98	(3)	11.94	(3)	10.62
Fecal Coliform[2]	114	40	37	106	148	20	47	41	20	13	19	28

[1] Station 11, Spanish Pass.

[2] Determined by multiple tube technique. Data for station 11, Spanish Pass; 13, California Point; and 26, California Bay near pump house. Expressed in numbers per 100 ml.

[3] Samples not collected.

inflow of fresh water from the river occurs almost every year. Oyster predators and disease organisms have responded to the inflow of fresh water by retreating further offshore, resulting in the revitalization of several acres of oyster reefs which had been abandoned to the predators by fishermen.

Concomitant with the "sweetening" effect of the fresh water were problems of pollution. Fecal coliform counts rose as dramatically as salinities dropped when fresh water from the Mississippi River was introduced into the estuary (Table 1). Bays off the mouth of the bayou have been closed repeatedly to oystering as a result of excessive fecal coliform concentrations (Colson, personal communication).

Coliform desities downstream from the small capacity syphon at White's ditch (Table 2) are high enough for concern at distances greater then 10 miles. Coliform densities may remain unattenuated for extreme distances downstream from the syphon, because flow is essentially channelized with limited opportunities for filtration through the adjacent marsh. Nevertheless, these results, as well as those from Bayou Lamoque, indicate that a source of polluted fresh water should not be in close proximity to a sensitive estuarine area.

Consideration of the data and discussion with local consultants, officials of state and federal government agencies concerned with fish and wildlife in the coastal areas, local land owners, fishermen, and a review of the literature led us to develop management plans for the Breton Sound and Barataria Bay estuaries. Briefly, the plan for Breton Sound proposes (1) that the protection from further deterioration, and enlargement, if possible, of nursery grounds and harvesting areas is of paramount importance, and (2) that these goal can be attained through

Table 2. Effect of distance from fresh Water diversion on annual average salinity and fecal coliform concentration[1] of natural surface water

(White's Ditch, 1979)

Parameter	Distance 3	5	5.5	8	11	14.5
	Miles					
Salinity ($^{o}/_{oo}$)	3.5	3.7	3.9	4.8	4.5	5.3
Fecal Coliform (No./100ml)	220	131	166	95	62	23

[1] Colony counts from membrane filter technique.

careful regulation of the salinity gradient within the estuary.

Extending the gradient seaward in spring and summer (April through September) will largely eliminate the problem of salt water intrusion and will maximize production of vegetation and commercial and sport organisms. During this period of the year the fifteen ppt isohaline should be in approximately the same position as the Ford line (U.S. Army Engineer District, New Orleans, 1970) (Figure 2). Increased production of vegetation during the spring and summer months may retard land loss lightly by directly contributing to land building and by reducing the rate of erosion.

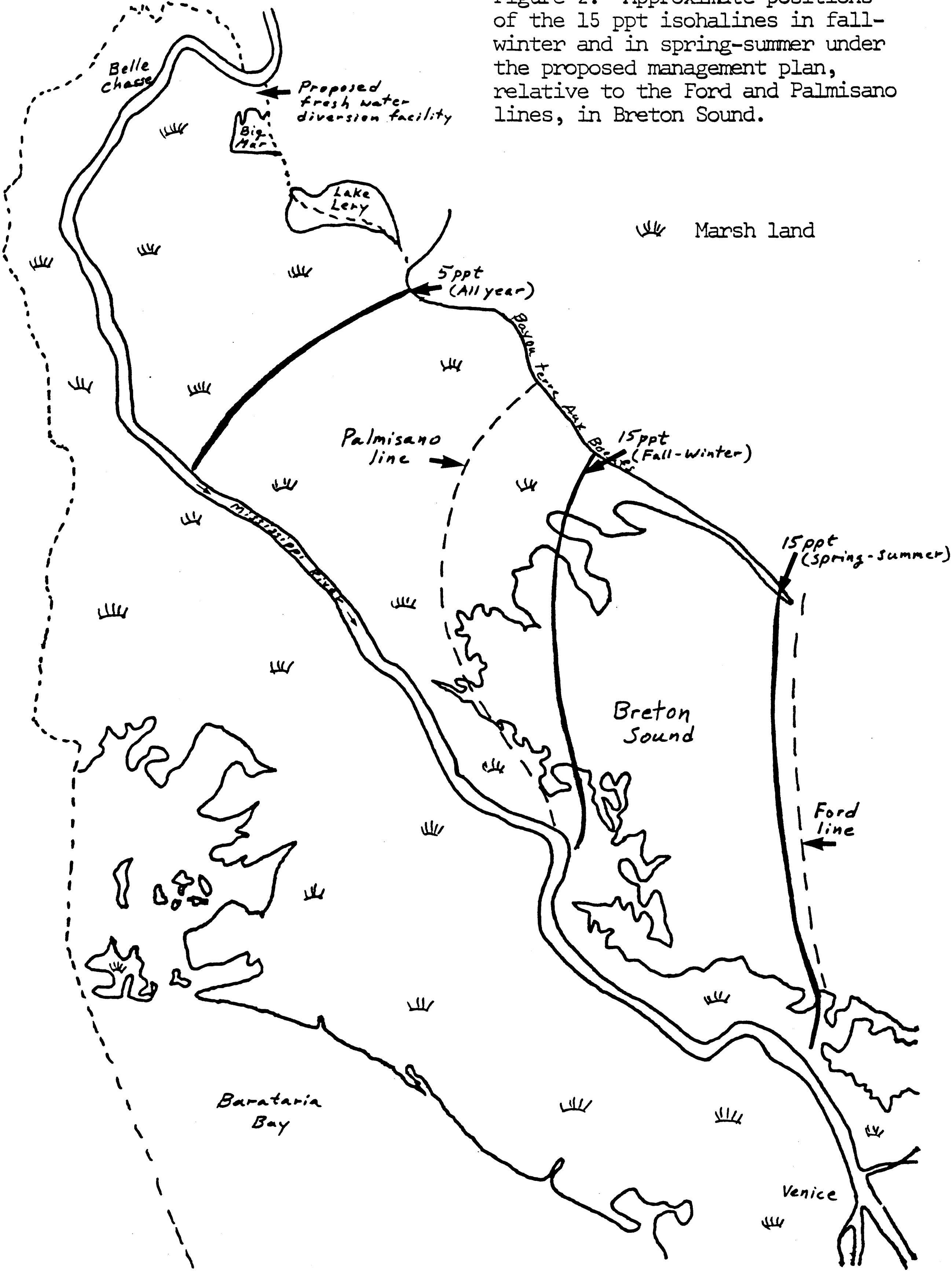
A steeper salinity gradient during fall and winter months (October through March) will maintain desirable ecological conditions within the marsh for the portion of the year. During the fall and winter period the fifteen ppt isohaline should be allowed to come shorewards to approximately the Palmisano line (Figure 2) suggested by a study group (U.S. Army Engineer District, New Orleans, 1970).

Analyses show that some fresh, nutrient-laden water, such as that available in the Mississippi River, is needed every month in Breton Sound estuary. The quantity needed ranges from 300 cfs in November to 2,400 cfs in the period from March through August for Breton Sound (Gagliano et al. 1973).

Regulation of the salinity gradient could be accomplished by diverting fresh water from the Mississippi River into an existing lake called Big Mar (Figures 1 and 2) at the upper end of the estuary. The diversion facility should be designed to deliver at least 6,000 cfs to assure adequate flow rates even at low river stage.

The 2,000 acre Big Mar would serve as a settling or stilling basin.

Figure 2. Approximate positions of the 15 ppt isohalines in fall-winter and in spring-summer under the proposed management plan, relative to the Ford and Palmisano lines, in Breton Sound.



A stilling basin at the upper end of the estuary would offer four advantages in addition to regulating the salinity gradient.

First, locating the stilling basin at the upper end of the estuary will assure that nutrient-laden river water is distributed throughout the marsh rather than being channelized. This maximizes the area of vegetation that will benefit from increased nutrients.

Second, the shallowness of the stilling basin (18 inches) will allow temperature of the cool river water to rise and be more like that in the marsh before leaving the stilling basin. Low water temperature in the marsh in the spring is detrimental to the development of brown shrimp and probably to other organisms. The careful addition of cooler water to the marsh during hot summer months will reduce the incidence of fish kills due to warm-water suffocation.

Third, water retention times of 6 hours or more (Table 3) should result in 90% or more of suspended sediments settling out in the stilling basin. Accumulated sediments will be dredged and can be used elsewhere. Water exiting the stilling basin will have less tubidity and may improve water transparency to sunlight in the marsh.

Fourth, pollution from the river should also be controlled in the stilling basin, as most of it is absorbed to suspended sediments and will settle out in the stilling basin. Also, the marsh between the stilling basin and the most critical nursery area will provide an important filter action or any pollution that might pass through the stilling basin.

Nevertheless, a rapid response capability should be developed to protect the estuary from pollution events originating in the river. Water quality in the river upstream from the diversion facility and in

Table 3. Estimated velocities, annual siltation rates, sediment accumulations, and water retention times for several diversion rates into Big Mar.

Diversion rates	Velocity[1]	Siltation rates per yr.[2]	1.5' Sediment accumulated in Big Mar	Water Retention time[3]
(cfs)	(knots)	(cubic Yards)	(years)	(hrs)
1,000	0.1	378,432	13.0	36
2,000	1.2	753,710	6.4	18
4,000	2.5	1,510,574	3.2	9
6,000	3.7	2,267,438	2.1	6
8,000	5.0	3,021,149	1.6	4.5
10,000	6.2	3,778,013	1.3	3.6

[1] Assuming an outfall channel 110' wide, 10' deep, with banks sloped 1.5 and a hydraulic gradient of 0.0005.

[2] Based on the annual silt load for the Mississippi River. Hence, these over-estimate the siltation rate, because diverted water will have less-than-average silt load.

[3] Based on an average depth of 1.5 feet.

the stilling basin should be determined on a daily basis. Water quality should be determined on a bi-weekly basis at sample locations arranged in two concentric arcs. The first arc two miles and the other five miles, below the stilling basin. The remainder of the estuary should be sampled on a weekly basis as is being done now in the existing programs. Discovery of an unusual pollution event would result in immediate closure of the gates of the diversion structure and raising of the crest gates on the spillways around the priphery of the stilling basin.

In addition to monitoring water quality, key benthic and nektonic organisms should be monitored on a regular basis in a cooperative program with other local, state, and federal agencies responsible for the protection of fish and wild life.

For Barataria Bay, a fresh water diversion facility should be located on the Mississippi River upstream from the city of New Orleans. However, the likelihood of such a facility being constructed in the near future is quite remote, in which case we offer an interim measure, as follows. Fresh water for Barataria Bay could be made available through the Algiers navigation locks to the intercoastal waterway (ICWW) and down Bayou Barataria to the Bay. The locks could be left wide open when the river is not more than 9 inches higher than the ICWW. This maximum head differential would permit up to 18,000 cfs through the locks (Table 4). At maximum head differential, water will flow through the concrete locks at a velocity of about 17.5 feet per second. Upon leaving the confines of the locks and entering the expanse of the ICWW, the water will immediately slow to about 2.5 feet per second. Allowing more water to pass through the locks would cause erosion of the ICWW banks.

Table 4. Flow characteristics through the Algiers locks[1] for various hydraulic heads.

Head (feet	Energy Gradeint	Volume (cubic ft./sec.)	Velocity in--- Locks	ICWW
			-----(ft/sec)----	
0.1	0.00013	6,624	6.4	0.9
.2	.00026	9,315	9.0	1.3
.3	.00039	11,385	11.0	1.6
.4	.00053	13,351	12.9	1.9
.5	.00066	14,904	14.4	2.1
.6	.00079	16,353	15.8	2.3
.7	.00092	17,595	17.0	2.5
.8	.00105	18,837	18.2	2.6
.9	.00118	19,976	19.3	2.8
1.0	.00132	21,114	20.4	3.0

[1] Assuming gates open wide. Hydraulic radius is 10 feet.

A current of 17.5 feet per second (11.5 mph, 10 knots) through the locks may prevent water-born traffic from entering the river from the ICWW. However, marine traffic need not be disrupted if the locks are operated in the usual manner whenever a vessel attempts to pass from the ICWW to the river. Lock operators have stated that it is possible to close the hydraulically operated gates against a head differential considerably greater than that proposed for fresh water diversion. Furthermore, the anticipated number of times the gates are operated for fresh water diversion purposes on a yearly basis should not cause any increased wear on the mechanism.

Diversion through the locks may occur on as many as 45 days during a dry year. During wet years, the river may be too high to permit safe diversion of fresh water through the Algiers locks at any time.

The maximum volume of water diverted from the river by the Algiers locks would represent approximately 10% of the 18,000 ckf needed in the bay on an annual basis according to Gagliano, et al. (1973). The amount of water diverted into the bay in most years will be less than 10% of the annual requirement, because fewer than 45 days will occur during which the river is less than nine inches above the water level in the canal. Nevertheless, any fresh water provided to Barataria Bay through the Algiers locks will help the bay, and the cost will be virtually zero; all that is required is to change the method of operation of the locks.

The quality of the water being diverted into Barataria Bay is of as much concern as the quality of the water being diverted into Breton Sound. Recent analyses of ICWW water and sediments below the Algeirs locks and downstream in Bayou Barataria[3] reveal no serious pollution problems. Identified pesticides and heavy metals were below toxic levels in sediments, while dissolved nutrients were present in moderate amounts in the water. Monitoring programs of water quality and key indicator organisms in the Mississippi River, the ICWW, Bayou Barataria, and in the Bay would be designed similarly to those suggested for Breton Sound.

Diverting fresh water through the Algiers locks will provide the needed time and distance for pollutants to be acted upon biologically or to physically settle out before the diverted water enters the environmentally sensitive estuarine area below the community of Lafitte.

Additional fresh water diversion through the Harvey Canal locks was considered and abandoned, because the 50-year old mechanism cannot close against a current sufficiently great to have any impact on fresh water needs in Barataria Bay.

[3] Data available upon request.

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COLONIZING PATTERNS OF TIDAL MARSH PLANTS AND VEGETATIONAL SECCESSION ON DREDGE SPOIL IN MISSISSIPPI

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ABSTRACT

Position and zone of plant colonization and establishment on barren shores of dredge spoil in relation to elevation and slope of the intertidal plane are described from observations and studies carried out over a 12 year period. Shifts or displacements of certain plant colonies and zones occur continuously for many years. *Spartina alterniflora* was consistently observed as the earliest colonizer of dredge spoil in an estuary of the northern Gulf of Mexico. Colonization and establishment of *S. alterniflora* occurred from seed at a consistently well-defined position above mean low water. Plant colonies and small immature stands usually spread centrifugally initially, then predominantly downward on the tidal plane from the point of establishment. The shift downward on the tidal plane is consistent over time. Vegetation established near the lower edge of the intertidal plane is generally composed of one to three species and remains relatively homogeneous for 5 to 6 years. About 30 other plant species which subsequently colonize certain spoil areas also arise from seed and many are characteristic of tidal marshes, however, many weedy species also occur. Displacement of mature stands was also downward on the tidal plane. The rate of succession and the variety of species increases with elevation in stands near the upper edge of the tidal plane. Shrubs and trees first appear on the most elevated parts of spoil deposits and man-made beaches. Shrub and tree establishment on spoil areas with elevations of 1.5 m (5 feet) or more above mean low water is usually much more rapid than establishment on places with lower elevations. Succession on graded slopes consistently occurs downward, often including a continuum of shrubs and trees near the tidal plane

INTRODUCTION

Plant colonization of new terrain formed in or along tidal marsh by natural processes is inconspicious because vegetative growth and spread of plants from adjacent areas generally keeps pace with the rate of accretion. Therefore, opportunities for the study of plant colonization, resulting from natural processes, are relatively rare and restricted to denudation of marsh terrain by hurricanes, flood erosion, fire, and marsh development behind rapidly forming sandy spits. The activities of man over the past two decades have resulted in much disturbance throughout the Atlantic and Gulf coasts of the United States. Among these were terrestrial, intertidal, and submarine topographic modifications required for navigation. "Spoil" islands result from deposits of substratum dredged for channel construction or maintenance or similar work. Spoil islands and shorelines constructed from deposits of dredged materials have provided unique opportunities to study the entire process of plant colonization, including the patterns of development, displacement and vegetational succession. Some of the observed phenomenon may be characteristic of pristine tidal marshes. No information is available on plant colonization of spoil islands in the estuaries of Mississippi.

The present paper reports on the key features of plant colonization, vegetation development, and succession on islands and shorelines constructed from dredge spoil along the Mississippi coast. How this information relates to natural tidal marshes, transplanting, and restoration projects is analyzed. Information on diked spoil areas is not included.

MATERIALS AND METHODS

Observations on plant colonization of spoil areas began in 1968 and documentation of species, elevation, tide relationships, seasonality, growth rates, and vegetational succession began in 1970. General observations were made on approximately 20 spoil islands with detailed study carried out on five spoil islands in Davis Bay, and a mile of sandy beach along northern shore of Davis Bay at Ocean Springs, Mississippi. Changes in the nitrogen, phosphorous, and potassium concentrations in the substratum of spoil deposited in two different locations were determined at selected intervals over several years using standard methods of analysis. Hydrogen ion concentration (pH) and oxidation-reduction potential (redox, Eh) was also determined periodically (Eleuterius, In review). General changes in the physical characteristics of the substratum were noted. We also compared the success of plant colonization on dredge material located 12 miles off the mainland shore with that on spoil islands near the mainland. Documentation of plant colonization, survival, and subsequent vegetation development in relation to configuration and elevation was carried out with transit, stadia rod, and many wooden stakes. Stakes were used to mark plants on the spoil islands. Elevation was referenced to elevational benchmark OSV2 Corps of Engineers, U. S. Army, (based on C=GSZ189). Dredging history of selected areas, prior to 1968, was obtained from the Corps of Engineers, U. S. Army, Mobile District.

RESULTS

From preliminary surveys and biological inventories conducted by the senior author in 1968-1972, several observable facts regarding plant colonization of spoil islands became apparent. Plant colonization of spoil islands in deep water offshore is ephemeral. Large waves and strong currents from the Gulf of Mexico and Mississippi Sound affect these spoil islands. The island substratum is constantly reworked by this high energy and numerous changes in topography and configuration result. Furthermore, spoil islands located far at sea generally result in sandy often dune-like deposits, because most of the organic matter is winnowed away by wind and waves. The sediment pattern of erosion, deposition, and erosion again is accompanied by a net loss in the volume of heaped sandy mass. Suspended material is dispersed over a large area of sea bottom. Hurricanes and local storms cause major changes in the configuration, topography, and elevation of offshore spoil islands. Consequently, all water and wind forces have a combined effect which eventually erode spoil islands to elevations near the surface of the water. Most spoil in deep water of offshore locations ultimately disappear under the surface of the sea. Thus, plant colonization of spoil islands which leads to established stands of vegetation occurs only on those spoil islands which are located near the mainland.

Spoil islands located in Biloxi, Davis, Back, Pt. Aux Chenes, and St. Louis bays are well protected from the great waves, currents, and winds characteristic of the open Gulf and Mississippi Sound. Many coastal bayous and river estuaries are also well protected. Spoil islands with sloping shores of grades less than 14 degrees are readily colonized by plants. Grades steeper than 14 degrees are not favorable to plant colonization apparently because the substratum is generally unstable and easily eroded. The slope of the spoil depends on the sediment composition. Over most of the coastal area of Mississippi the seabottom near the mainland is characterized by silt overlaying sand, with clay predominating at lower depths of 6.1 m (20 feet) or more. Most of the silty materials are generally washed away during the dredging operation and sands are difficult to deposit in heaps, but spoil islands composed of sandy clays generally have steep grades initially. All spoil islands undergo a "weathering" or "aging" process before plants begin to colonize them. Segregation of the mixed dredged sediments occurs with some re-organization into a layered substratum. The availability of nitrogen, phosphorous, and potassium concentration also changes with this process probably caused by adjustments in pH and Eh. Apparently chemical changes and adjustments constantly occur in dredge spoil and if one compares certain chemical aspects soon after the dredging occurred with those determined 10 years later, certain differences can be found (Eleuterius, In review). However, based on numerous observations made over many years, plant colonization does not seem to be strongly controlled or related to chemical aspects of undiked dredge spoil islands.

Except for the large change in Eh shifting from an extreme reducing environment toward a more oxidizing one which occurs soon after deep sediments are dredged up, no significant chemical change occurs in the substratum located low on the tidal plane which, based on the present knowledge available from plant ecology and plant physiology research were revealed which could inhibit plant germination or subsequent growth. The chemical characteristics of dredge spoil islands studied here appear to fall within the tolerances of the major tidal marsh plants and compare favorably within variations and ranges of undisturbed (natural) habitats (Eleuterius, In review). We feel that elevation, slope of the shore and stability of the spoil island substratum are far more important to plant colonization than the chemical aspects in the present study. However, spoil placed in earthen dikes are an entirely different problem, some remaining barren (devoid of vascular plants) for over 12 years. Spoil enclosed by earthen dikes have both chemical and physical characteristics which inhibit and often prevent plant colonization and subsequent plant growth (Eleuterius, In press).

Locally, tidal amplitude is about 54.8 cm (1.8 feet). Spoil extending to an elevation of 60.9 cm (2 feet) above MLW generally remain marsh islands, while those extending 0.9 m - 1.21 m (3-4 feet) above MLW have shrubs eventually occupying the area above 76.2 ± 15.2 cm (2.5 feet) MLW elevation. The vegetation of areas 1.3 m (4.5 feet) or more above MLW generally become dominated by trees.

On spoil islands extending 60.9 cm or 91.4 cm (2 or 3 feet) above MLW the primary colonizer is Spartina alterniflora. Numerous observations indicate that all plant species colonizing spoil islands arise from seed. Successful establishment of seedlings occurs high on the tidal plane. There appears to be no seasonality to colonization by Spartina alterniflora on the Mississippi coast. Although seeds are deposited lower on the tidal plane and seedlings may appear they are swept away. On low profile spoil islands extending 60.9 cm (2 feet) or less above MLW, seedlings become established on the highest areas. Because of frequent submergence, currents and wave action, successful seedling establishment takes longer on low profile islands 60.9 cm (2' MLW) than on more elevated spoil areas 0.9 m to 1.21 m (3-4' MLW). Spartina alterniflora becomes established on spoil from 0.5 m to 1.1 m (1.5 to 3.5 feet) and sometimes 1.21 m (4') above MLW. Successful establishment from seedlings is more readily and more frequently achieved in the upper end of the vertical range. Seeds are deposited on exceptionally high tides during spring and fall or frequently during local storms. Germination appears to be most successful when the seeds are placed high on the tidal plane and left undistrubed by subsequent tidal fluctuation. Often a single established seedling of Spartina alterniflora leads to a dense, monotypic stand. Sometimes seedlings become established six months after the spoil is deposited, other times 3 years. If a seedling becomes established at the 60.9 cm (2') elevation on a sloping shore of a spoil island extending 1.1 m (3.5') above MLW, subsequent vegetative growth results in a centrifugal spread from the point of origin. However, the preponderance of growth is downward on the tidal plane. Although the entire island may be covered by Spartina alterniflora at the end of 2 or 3 years, depending on the size of the emergent area and remaining dominant for several years. One half acre can be easily and completely covered by Spartina alterniflora in 2 years from a single transplant in Davis and Biloxi bays. The plants in the upper area eventually become shorter and less vigorous than those nearest the water. Distichlis spicata generally replaces the plants of Spartina alterniflora. This replacement of Spartina alterniflora begins on the highest area of the spoil and extends downward. The lower edge of the Spartina alterniflora colony or stand also continues to grow downward on the tidal plane. Four or more years may be required before Spartina alterniflora reaches the lowest limit of vertical distribution at 45.7 cm (1.5') below MLW. The entire zone of vegetaion shifts downward on the tidal plane, often completely leaving the place of origin.

The above described phenomenon has been consistently observed for over 12 years on numerous spoil islands along the Mississippi coast. The events in the pattern of establishment and subsquent vegetative spread downward on the sloping spoil island shores was almost exactly the same in all observed examples. Furthermore, all other plant species subsequently established on these spoil islands move downward on the tidal plane.

The general pattern of succession on the upper or higher part of spoil islands up to 1.1 m (3.5') above MLW is Spartina alterniflora, Distichlis spicata then the shrubs Iva frutescens and Baccharis hamilifolia. It may take 10 years before Juncus roemerianus begins to show up on these spoil islands, often in the Distichlis spicata zone. Frequently Juncus roemerianus becomes established in the upper edge of the Spartina alterniflora zone, eventually crowding out the later species. Juncus roemerianus grows very slowly in comparison to Spartina alterniflora or Distichlis spicata, but once it dominates an area no other species seems to be able to displace it. Vegetation development on these relative low profile spoil areas is toward the natural zonation patterns found at similar elevations in nearby undisturbed (pristine) tidal marshes. However, a considerable amount of time is required to replace the Spartina alterniflora - Distichlis spicata zonation with a Spartina alterniflora - Juncus roemerianus zonation characteristic of most adjacent tidal marshes. Most small spoil islands occur in bays, bayous, river estuaries. From 6 to 15 years may be required before Juncus roemerianus begins to colonize barren or vegetated spoil areas and from 10-20 years or longer before closed stands are formed.

The pattern of colonization and vegetation development described for spoil islands is not unique to them and may represent a universal process since it has also been observed on man-made beaches. East Beach located on Davis Bay (Ocean Springs, Mississippi) is a man-made beach, about 1.6 km (1 mile) in length, composed of sandy dredge spoil. The beach was pumped up about 35 years ago and replenished in 1962. The beach is maintained by frequent harrowing to eliminate most of the vegetative cover, except for a small tract. This small area about 45.7 x 60.9 m (150' x 200') has not been disturbed since 1969, consequently plant colonization and succession of vegetation could be observed periodically and documented. Today a perfect profile diagram can be obtained by direct observation of this site, which shows the distribution of shrubs and trees downward on the gently sloping spoil surface.

Spoil areas that remain greater than 1.1 m (3.5') above mean low water after 2 or 3 years generally occur in tidal marshes or very shallow estuarine waters protected by large land masses. Spartina patens generally colonizes these very tall, steep-sided (often greater than 45 degrees) spoil areas. Distichlis spicata has been observed to colonize low profile in the upper part of Back Bay of Biloxi (Popps Ferry). Several spoil areas located in Jackson County are more than 7.3 m (24 feet) in elevation (MLW) and Spartina patens covered these areas very rapidly (2 years) after the spoil was deposited. In Hancock County spoil deposits over 15.2 m (50 feet) in elevation (MLW) were also rapidly covered by Spartina patens which dominated the areas for several years. Subsequently, shrubs, trees, and variety of weedy species began to invade the areas. Most spoil areas above 1.5 m (5 feet) MLW are colonized by an array of plant species, resulting in heterogeneous distributions. However, the oldest shrubs and trees generally always occur on the highest portions of the spoil and the younger ones near the bottom of the spoil. Trees, shrubs and weedy plant species which inhabit spoil areas greater than 1.5 m (5 feet) above MLW are listed in Table 1.

Table 1. Plant species colonizing spoil areas 1.5 m (5 feet) above MLW or more in elevation.

Albizia julibrissin (Mimosa)
Amorpha fruticosa
Andropogon elliottii
Baccharis halimifolia
Boltonia asteroides
Cephalanthus occidentalis
Cinnamomum camphora (*Camphora camphora* of some authors)
Cladium jamaicense
Eleocharis cellulosa
Hibiscus moscheutos
Hydrocotyle bonariensis
Ipomoea stolonifera
Juncus brachycarpus
Juncus polycephalus
Kosteletzkya virginica
Lippia nodiflora
Lonicera japonica
Luwigia leptocarpa
Melia azedarach (Chinaberry)
Mikania scandens
Muhlenbergia capillaris
Panicum amarum
Panicum repens
Panicum virgatum
Paspalum distichum
Paspalum floridanum
Phytolacca americana
Pinus elliottii
Rubus sp.
Rumex crispus
Salicornia bigelovii
Salix nigra
Sambucus canadensis
Sapium sebiferum (Tallow Tree)
Scirpus robustus
Sesbania exaltata
Sesbania punicea
Sesbania vesicarum
Setaria geniculata
Setaria magna
Solidago sempervirens
Spartina patens
Spartina spartinae
Typha angustifolia
Uniola paniculata
Verbena canadensis

DISCUSSION

Colonization of spoil islands by seedlings of _Spartina alterniflora_ appears to be exceedlingly difficult at locations low on the tidal planes. However, mature plants have the capacity to invade these lower intertidal regions, which indirectly points out the vulnerability of the seedling stage in the life cycle of tidal marsh plants. _Distichlis spicata_ and _Spartina patens_ are also precocious colonizers of spoil islands, but the former occurs on low profile spoil islands in low salinity areas and the later on tall spoil heaps in well protected estuarine locations. Shrubs become established several years or more before _Juncus roemerianus_, but the direction of the vegetation succession is toward the vegetative structure characterized by nearby pristine tidal marsh.

Plant colonization and establishment of spoil islands at higher elevations than found in undisturbed marshes and the subsequent movement and displacement downward on the tidal planes appear to be part of a universal process. Trees and shrubs appear to crowd the upland area near the upper edge of tidal marshes. The combined effects of tides, salinity inundation and water logged soils keep the shrubs, trees and other plants held back.

Shrub and tree colonization also proceeded downward on man-made beaches. The exact cause for this phenomenon is unknown, but it indicates the higher regions of low profile spoil are more favorable to seedling establishment and subsequent growth than lower areas. However, the lower intertidal regions are destined to remain essentially the same because the constant effect of the tides. Higher regions are destined to change. On the man-made beach the higher portions of the spoil perhaps change in soil water, salinity and nutrients caused by leaching created a more favorable environment. Unfortunately, previous work does not reveal any strong evidence for this assumption, and it remains unsolved.

The work was restricted to convex (lens shaped) spoil islands and diked areas were not studied, primarily because diked areas do not produce good tidal marsh or habitats for other plants. Consequently, the configuration of an undiked spoil island was important. A slope not greater than 14 was essential for tidal marsh establishment. Protection of the spoil islands from strong winds and waves by their location in partially enclosed estuaries was important for plant colonization and sibsequent development of established stands. Substratum is also important, but generally those spoil islands near the mainland do not winnow out to sandy, dune-like habitats.

The results of this study correspond in certain features to undisturbed tidal marsh located near the spoil islands studied. Spartina alterniflora has been surveyed at depths of 45.7 cm (1.5 feet) below MLW (Eleuterius and Eleuterius, 1979). However, it has recently been determined that locally Spartina alterniflora in undisturbed marsh tends to grow downward on the intertidal plane (Eleuterius, In preparation). This corresponds exactly with our observations and determinations of the maximum depth ultimately reached by Spartina alterniflora on spoil islands.

The findings of this study also have direct bearing on transplanting work utilizing Spartina alterniflora and other tidal marsh species in restoration and marsh creation projects. While conducting a large transplanting study on the Mississippi coast 10 year ago, we utilized elevational determinations delineating vegetational zones in undisturbed marsh as vertical parameters and guidelines for carrying out transplanting work (Eleuterius, 1974). Several restoration and enhancement projects on the Mississippi coast also used these vertical parameters with the same result. Transplants made on the lower portion of the intertidal plane were consistently lost because of erosion and unknown factors. We surmise from these observations and experiences that it is better from an economic and ecological point of view to make transplants of each species higher on the intertidal plane than found in adjacent pristine marshes and let the plants adjust themselves downward until they reach locations and positions for optimum growth. This may take 4-6 years for Spartina alterniflora, which is, unfortunately, often longer than most funded projects allow for observing and reporting results. Furthermore, species such as Juncus roemerianus which have slower growth rates than Spartina alterniflora should not be neglected. Additional time is needed for this species to develop closed stands. This appears to be consistent with what occurs in pristine marsh, but the vegetational development occurs so slowly that one can not readily observe it. Restoration and marsh creation projects are becoming increasingly important and will continue to do so in the future. Man can speed up the rehabilitation effort by assisting natural processes.

Much can be gleaned from the colonization of spoil islands and other disturbed areas. Although some work has been done on spoil islands of the Florida coast by Beaman (1973) and Lewis and Dunstan (1975) much needs to be done. Characteristics of spoil islands which become colonized by plants and which develop a cover of vegetation should be recorded at frequent locations along the Gulf and Atlantic coasts and the results used in preparing future habitats from dredged material. If the proper habit is created, one conducive to plant colonization, the proper plants will get there. Man, however, can often speed up the natural process by transplanting.

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LONG-TERM OBSERVATIONS ON SEAGRASS BEDS AND SALT MARSH ESTABLISHED FROM TRANSPLANTS

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ABSTRACT

Observations on the growth of surviving transplants of seagrass have been made periodically since transplanting trials were carried out in the Mississippi Sound in 1971. *Halodule beaudettei*, *Thalassia testudinum* and *Cymodocea manatorum* were the three seagrass species transplanted. *H. beaudettei* had the highest rate of survival and a very rapid growth rate, forming relatively large beds in a short period of time. Seagrass beds constantly change in configuration and shift their location over relatively short periods of times. Proliferation of new roots and rhizomes from every erect shoot of seagrass does not occur and appears to be a major problem in transplanting programs. More than two or three years, after transplanting, is needed to adequately evaluate trials on seagrass tidal marsh plants.

The growth and stand-producing ability of *Panicum repens*, *Spartina alterniflora*, *Juncus roemerianus*, *Spartina cynosuroides*, *Spartina patens*, *Distichlis spicata*, and *Phragmites communis* varied over a ten year period. Most mature shoots of tidal marsh plants can produce new stand because of prolific rhizome and root growth. Stands formed near the lower edge of the intertidal plane remain vigorous and relatively homogeneous in vegetational structure, but those near the upper edge of the intertidal plane do not. New information on a successful containment method using hay bales for establishing marshes in estuaries is given.

INTRODUCTION

Degradation of tidal marshes and seagrass communities by dredge and fill operations has caused considerable concern for these estuarine and marine ecosystems over the past 20 years. Consequently, much research has been initiated to seek ways to mitigate disturbance from coastal engineering activities and to rehabilitate or restore previously destroyed areas. Vegeative propagation of local estuarine and marine flowering plants in transplanting projects has been one of the most promising techniques developed in restoration efforts. However, transplanting work has been limited to a small number of tital marsh and seagrass species. The long-term results of experimental transplanting studies and large scale pilot projects involving transplants or seeds have not been reported. The primary reason for this inadequacy is because of the short period of observation necessitated by most funding contracts.

Long-term observations on the survival, growth, general stand-producing ability, and successional patterns resulting from transplanting studies carried out in 1971-1973 are reported here and a list of species needing trials to determine their suitability as transplanting stock is given. Engineering problems, suggestions, and requirements from ecological studies relative to tidal marsh establishment are also evaluated and discussed. The ultimate fate of surviving seagrass transplants in relation to the peculiar marine environment of the Mississippi Coast is also explained. Reproductive problems of seagrasses and suggested approaches to obtain solutions are given in view of the low incidence of transplant survival.

MATERIALS AND METHODS

It is not our intention to reiterate the array of details surrounding the initiation and subsequent early results of the large-scale study upon which this paper is primarily based, but rather to comment on the observations made over a 10 year period and summarize the lessons learned. Under this section we shall, however, list the principle published sources of detailed information for the convenience of the reader. Some background is necessary in order to understand the points made in the paper, so general information will be given here.

Transplant trials were carried out seasonally in random plots to compare the performance of tidal marsh plant species, as to survival and subsequent growth. These plots were staked out on a five acre spoil area located in Simmons Bayou, Jackson County, Mississippi. For a selected number of species, the practice of using a single shoot was assessed against that of utilizing transplants rooted in peat pellets. The details of transplanting procedures are reported in Eleuterius (1974). Important information on transplanting in relation to elevation was obtained from two restoration projects carried out on the coast of Mississippi. General comparisons of growth rates and successional aspects of other vegetation are made on these projects. Description of an open-water dike constructed from hay bales is also provided (Eleuterius and Gill, In review).

Seagrasses were transplanted using several methods, however, the comments made here regarding them will be restricted to those made using construction rod and construction mesh anchors during the later stages of the work. Four galvanized pipes 7.6 cm (3 inches) in diameter and 6.7 m (22 feet) long were used to mark the rectangular transplanting sites offshore. Details of these transplanting studies are also found in Eleuterius (1974). New information resulting from the marking of the edge of numerous seagrass beds with iron rods, bent into staple-shaped pins (Eleuterius, In review) is also important. It is our purpose here to combine and synthesize pertinent information from these sources as well as previously unreported information into new perspectives regarding transplanting and restoration work.

RESULTS AND DISCUSSION

Transplanting tidal marsh plants

Our propagating trials in random plots indicated that preliminary rooting of marsh plants was unnecessary. Individual shoots gave rise to numerous successful stands with 60 to 80% survival when the transplants were made from November to February. Plant species arranged according to survival trials in descending order are Panicum repens (95%), Juncus roemerianus (95%), Spartina patens (80%), Distichlis spicata (70%), Spartina cynosuroides (70%), Spartina alterniflora (65%) and Phragmites communis (30%). Survival data alone gives incomplete insight into the final result of transplanting and subsequent plant growth. Plant growth in random plots was vastly different among some of the species. In random plots Spartina patens, Spartina alterniflora, Panicum repens, and Distichlis spicata formed closed stands within a year. Spartina cynosuroides and Phragmites communis did not grow appreciably during the entire length of the 3 year study and did not form closed stands in the subsequent eight years of observation, probably because they were crowded by more rapidly growing species. Juncus roemerianus was also a slow growing species which continued to spread although crowded by other species. Juncus roemerianus formed closed stands in five years when transplanted on 1.2 m (4 foot) centers. Other marsh species that need their propagating abilities investigated are shown in Table 1.

From the above described transplanting trials we also learned that entire stands of certain species established from transplants would shift their location and that the species are in a constant state of adjustment. Vegetational zones formed from coalescend transplants have moved 6.1 m-15.2 m (20-50 feet) from their original position. Some species still occupy their original locations, while others have been displaced. And some stands or zones have remained monotypic on spoil over 1.1 m (3.5 feet) above MLW in elevation for over six years. Frequent observations showed that after seven years the mosiac pattern of the "marsh" began to show evidence of succession. Andropogon elliottii, Iva frutescens, Boltonia asteroides, Pinus elliottii and other species began to appear as scattered individuals.

Two restoration projects in Biloxi Bay and Biloxi Back Bay were utilized to test the results from transplanting trials. Information on transplant survival and performance versus elevation were not part of our original transplanting study, nor did we suspect that there would be any problem in restoring plants on a sloping intertidal plane. Elevational data used as transplanting guidelines were taken from adjacent undisturbed marsh and transplants of Spartina alterniflora and Juncus roemerianus, (in these instances 10.1 cm (4 inch) plugs were placed on 1.2 m (4 foot) centers). The gradient planting scheme was to place Spartina alterniflora from 45.7 cm (1.5 feet) below MLW to 54.8 cm (1.8 feet) above MLW and Juncus roemerianus from 54.8 to 91.4 cm (1.8 to 3 feet) above MLW. All of the transplants of Spartina alterniflora made over the lower 2/3 of the zone died at both sites. Transplant death was caused by sediment erosion, frequent wave action and periodically swift currents. Unfortunately we were not equipped with the proper instruments to record wave force and current velocities. Since one site was established in 1972, we were able to observe the reinvasion of this lower zone by surviving transplants of *Spartina alterniflora* from the upper part of the zone. Since 1972 *Spartina alterniflora* has spread over 15.2 m (50 feet) down the intertidal plane to reach an elevation of 45.7 m (1.5 feet) or more below MLW. The zone was about 30.4 m (100 feet) wide.

The second site was covered with transplants in 1977 but the rate of reinvasion appears to be slower than at the first project site probably because of exposure to tidal waters of a much lower salinity. However, the same reinvasion process was observed, as described above and the transplants of *Spartina alterniflora* will probably eventually adjust themselves and spread low on the intertidal plane. Transplants of *Juncus roemerianus* have remained as separated clumps. *Distichlis spicata* and *Spartina patens* have colonized the area between the clumps of *Juncus roemerianus*. A similar process of colonization by these species was also observed on the first site.

An important principle of transplanting has been revealed from these studies; which is, to transplant high on the intertidal plane and let the resulting vegetation zone spread downward. This process corresponds with that of natural colonization (and establishment) on spoil and the seasonal fluctuation in the growth of the lower edge of the *Spartina alterniflora* zone in undisturbed tidal marsh. We recommend transplanting *Spartina alterniflora* sprigs, plugs, seedlings or sowing seeds between 54.8 cm (1.8 feet) and 91.4 cm (3 feet) above MLW on sites in the northern Gulf of Mexico and that the principle be applied accordingly to other coastal areas with different tidal regimes. In Mississippi, Eleuterius and Eleuterius (1979) showed that the tidal amplitude is 54.8 cm (1.8 feet) which fluctuates over a vertical annual range of about 1.5 m (5 feet) .

Another important factor revealed by long term observation was that relatively large areas from 46.4 to 139.3 m^2 (500 to 1500 square feet) could be covered by a single sprig (shoot) sometimes within five years. This clearly shows that a spoil area may be rapidly and densely covered from a few surviving transplants and that a low survival of transplants does not indicate failure, at least for fast growing species.

The results from these transplanting trials and restoration projects where plant growth has been observed over an 8-10 year period, has led to the conclusion that the primary problem is not a biological one involving plants per se but those of engineering and ecology combined. In this regard we mention a new method of spoil containment using hay bales, which was very successfully used in Davis Bay (Eleuterius and Gill, In review). Silty sand and silty clay sediments were contained within a wall of hay bales stacked 2-4 high and held together with wooden stakes 5.1 x 5.1 cm and 3.1 m (2 x 2 inches and 10 feet long) that were driven through the bales and into the bay bottom. Water was allowed to pass through the hay bales during the dredging while a very large amount of the organic material was retained within the enclosed area. Sand and clay was easily contained. The slope of the spoil island was of a low profile and grade, resulting in an excellent habitat for <u>Spartina alterniflora</u>. Colonization of the spoil occurred within six months after spoil deposition. Some transplants of <u>Spartina alterniflora</u> were also

Table 1. Tidal marsh plants. Species needing testing propagation trials to assess their suitability as transplanting stock. These species are selected because of: their prevalence in tidal marshes throughout the northern Gulf of Mexico; their rapid, spreading growth and common occurrence on dredge spoil or other disturbed areas. Furthermore there is an almost total lack of information on the progagation of estuarine shrubs. However, very few of the more than 200 species of vascular plants inhabiting tidal marshes have been adequately investigated as potential transplanting stock.

Scirpus americanus
Scirpus olneyi
Scirpus validus
Eleocharis cellulosa
Zizania aquatica
Zizania milacea
Setaria magna
Sagittaria lancifolia
Batis maritima
Salicornia perennis
Iva fruitescens
Baccharis hamilifolia
Myrica cerifera
Illex vomitoria

made to this area. Vegetative growth from the naturally established colonies (from seed) and transplants has been prolific. Because the hay bales do not form an impervious barrier to tidal waters, the spoil is flooded frequently, without erosion. The hay bales are presently decomposing, but we feel that the vegetative cover of *Spartina alterniflora* will stabilize the spoil island. We recommend the use of hay bales as a containment wall around spoil deposited in shallow waters.

Transplanting seagrass

In contrast to that of tidal marsh plants, transplanting of seagrass involves a great biological problem. Seagrasses do not produce roots and rhizomes as proliferately as do the most abundant tidal marsh plants. Therefore, a smaller percentage of shoots are apt to give rise to new shoots. In our experience, only a small percentage of seagrass shoots (about 15%) will give rise to prolific vegetative growth.

Only about 30% of the transplants (141 out of 500) survived at the end of one year. Of the survivers, ca. 80% had spread sufficiently to be classed as an "established grass bed" (at least 161.3 cm^2 (25 square inches)). Grass bed size ranged from the minimum to as large as 1.8 m^2 (20 square feet). The details of observations and measurements made on these seagrass beds at the end of one year are reported in Eleuterius (1974) and Eleuterius and McClellan (1976). Not enough time was allowed by the time frame of study and final report preparation to properly evaluate this seagrass transplant work. Compressed time frames characteristic of most funded studies may lead to hastily drawn or erroneous conclusions. Low survival of seagrass transplants, for instance, is not that important if a large number of transplants are made. Subsequent observations recorded here have not been previously reported.

At the end of the second year about 100 of the seagrass beds remained alive. However, this was not immediately clear because there were only a very few established beds within the boundaries of the staked transplanting sites. Observations made during and at the end of the first year of growth indicated that the established beds were shifting away from the location where the seagrass were originally transplanted. This was easily determined by linear distance to the buried wire mesh anchors. Established beds were found at increasing westward distances from their original metal anchors. The anchors were tied together trotline fashion, so that they were in lines. Concurrent to observations made in the summer of 1975 several hundred existing and naturally established seagrass beds were marked with metal pins around their peripheries in the shallow waters immediately north of Horn and Petit Bois islands. The results of this study clearly show that seagrass beds in Mississippi Sound are shifting and the preponderance of movement is overwhelmingly westward (Eleuterius, In review). At the end of the third year of observation, most seagrass beds established from transplants had shifted out of the delineated areas and naturally established beds were shifting across the original transplant sites. We estimate that approximately sixty of our seagrass transplants resulted in established beds of a size larger than 3.3 m^2 (36 square feet) with the largest beds covering 92.9 m^2 (1000 square feet) or more. Exact evaluations are difficult to be made because of continous movement, coalescing of two or more beds and separation of a single bed into two or more beds. Seagrass beds established in this study are presently interdispersed in the natural seagrass communities from 16.7 m to 35.1 m (55 to 115 feet) westward of the original transplanting sites.

The dynamic aspect of broken, discontinuous or patchy distribution patterns of seagrass beds characteristic of Mississippi Sound were not known at the time transplanting trials were first carried out in 1971. We now realize that specific or particular area of seabottom may not be readily, permanently, or continuously covered by seagrass. However, this does not mean that transplanting techniques do not have application here. Thousands of acres of seagrass, near Round Island, were destroyed in 1969 by Hurrican Camille. We believe that natural reestablishment in such denuded areas locally is hindered by an absence of viable seed or that no seeds are produced. Therefore, the use of transplanting techniques may be extremely useful by introducing seagrass into these large, barren areas again.

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SHORE EROSION CONTROL DEMONSTRATIONS IN FLORIDA

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ABSTRACT

Two Florida sites, Stuart Causeway-Jensen Beach Causeway and Basin Bayou State Recreation Area, were selected for study under the provision of the federal Shoreline Erosion Control Demonstration Act of 1974 (Public Law 93-251, Section 54). The USDA Soil Conservation Service assisted with the vegetation plan and planting in the demonstrations. Initial results of the study are reported here.

INTRODUCTION

The Shoreline Erosion Control Demonstration Act of 1974 (Public Law 93-251, Section 54) authorized a national low cost shoreline erosion control development and demonstration program under the direction of the U. S. Army Corp of Engineers. The program consists of planning, constructing, operating, evaluating, and demonstrating prototype shoreline erosion control devices, both structural and vegetative. The Act also created the Shoreline Erosion Advisory Panel (EAP) to advise and assist the Corp of Engineers in carrying out the demonstration program.

The Bureau of Beaches and Shores of the Florida Department of Natural Resources (FDNR) nominated two sites for location for demonstration projects; i.e., the Stuart Causeway-Jensen Beach Causeway site, Martin County; and the Basin Bayou State Recreation area, Walton County. These sites were approved by the Shoreline Erosion Advisory Panel.

The USDA Soil Conservation Service was requested to assist with the vegetative plan and planting in the demonstrations. This paper furnishes information on the vegetative aspect of the demonstration.

AREA DESCRIPTION

Causeway Sites

The Stuart and Jensen Beach Causeways are located in the Indian River estuary. The estuary in the area of the causeway is connected to the Atlantic Ocean by the St. Lucie Inlet which is 4.8 km (3 miles) south of the Stuart Causeway and the Fort Pierce Inlet located about 25.7 km (16 miles) north of the Jensen Beach Causeway.

The east island of the Jensen Beach Causeway designated as site 2S was heavily vegetated with Australian pine, *Casuarina equisetifolia* L.

They multiply very rapidly providing a dense sun screen which usually results in denuding the immediate area of many types of native vegetation (COE, 1978a; Craig et al., 1977). The mangrove genus Avicennia, Laguncularia, and Rhizophora are native to the area; however, they are particularly susceptible to such overgrowth and generally will not develop in such areas (COE, 1978a).

The site 2S has no erosion control devices other than broken concrete rubble dumped along the east and west ends of the island. The shore was eroding with root masses of Casuarina equisetifolia exposed along the shoreline.

The most common waves affecting the site are wind-generated. However, there are also boat waves, particularly at the west end of the site which is close to the Intercoastal Waterway, a Federal Navigation project extending along much of the coastal length of Florida. These waves vary in size from ripples to as large as .61 cm (2 feet) in height. Maximum wave heights are limited by relatively short wind fetch and shallow river depths. Wind generated waves are the primary cause of sand losses from the shoreline and cause most of the shoreline damage in the study area (COE, 1978a).

The tide ranges for the demonstration site are 30 cm (1 foot) for mean tide and 36.6 cm (1.2 feet) for spring tide. Site 2S was located in the most protected area and considered the most suitable for demonstrating erosion control using coastal vegetation (COE, 1978a).

Basin Bayou Site

The Basin Bayou State Recreation Area located on Choctawatchee Bay in Walton is a 116.2 ha (287 acres) tract leased from Eglin Air Force Base by the Division of Recreation and Parks, Florida Department of

Natural Resources. The park has approximately 914 m (3000 feet) of shoreline. Tides in this area are diurnal, with a mean tidal range in the bay of 15 cm (0.5 foot) and an extreme, except during storms, of about 45 cm (1.5 feet). Wave action along the shoreline is normally insignificant (COE, 1978b). However, during periods of brisk, southwesterly winds the water level is raised by wind set-up, creating higher, more energetic waves. It is this condition which is primarily responsible for erosion in this area. Chactawhatchee Bay is one of the major bays on the northern shore of the Gulf of Mexico with a total area of about 316 km^2 (122 $miles^2$). The bay averages 6.4 km (4 miles) wide. Approximately 41 km^2 (16 $miles^2$) has a depth greater than 9 meters (30 feet). However, in the demonstration area the water is very shallow with depths of less than 1.2 m (4 feet) extending over 305 m (1000 feet) offshore. A narrow sand beach exists along most of the shoreline with sandy berm approximately 1.5 m to 3 m (5 to 10 feet) high. This berm is being actively eroded by wind-driven waves out of the southwest and a slowing rising sea level. Large trees; both oak, *Quercus virginiana* Mill and pine, *Pinus elliottii* Engelm, are being undermined and toppled into the bay. Erosion and the shifting of the narrow beach by wave action is preventing herbaceous vegetation from growing seaward from the top of the berm.

RESULTS AND DISCUSSION

Jensen Beach Causeway

The existing trees, *C. equisetifolia*, were removed from the shore back to the powerline right-of-way approximately 15.2 m (50 feet) except for 2 control sections. The 396 m (1300 feet) of shoreline was divided into 10 nearly equal sections for planting and observations. One section where the C. equisetifolia were removed was used as a control section with no planting made. Plant selection was based on adaptability to the area and commercial availability. Plants not in great demand are grown for special orders and therefore are not readily available.

Plants selected were red mangrove, Rhizophora mangle L.; black mangrove, Avicennia nitida Jacq.; white mangrove, Laguncularia racemosa Gaertn.f.; smooth cordgrass, Spartina alterniflora Loisel. (Figure 1); salmeadow cordgrass, Spartina patens (Alt.) Muhl; siltgrass, Paspalum vaginatum Swartz (Figure 2); sea grape, Coccoloba uvifera and silverthorn, Elaeagnus pungens Thunberg. C. uvifera and E. pungens were planted at the highest elevation where used for traffic control.

With the exception of S. alterniflora and S. patens all plants were containerized plants. S. alterniflora was planted as plugs and S. patens was planted as sprigs and containerized plants. Tree removal, landfilling and shaping were done by the Martin County Department of Transportation personnel and equipment.

Planting began on June 27, 1979, with completion on July 20, 1979. Mangrove plants were approximately 30 cm (12 inches) tall at planting. Plants of this size were badly damaged by waves and were covered with wash-in sea vegetation.

On September 3, 1979 Hurricane David passed through the area creating waves that destroyed most of the vegetation except for the two upper rows of P. vaginatum (Figure 3).

The area was replanted in March 1980. At this planting the mangroves were 60 cm (2 feet). Fertilizer was applied in the form of a slow release fertilizer (14-14-14 and 7-40-6). S. alterniflora received 56 kg/ha of nitrogen from 14-14-14 and 56 kg/ha nitrogen from 7-40-6.

Figure 1. Planting S. alterniflora plugs, Section 3, Jensen Beach Causeway, July 1979.

Figure 2. Section 3, Planting of S. alterniflora, S. patens, and P. viginatum, August 1979.

All other grasses received 56 kg/ha nitrogen from 14-14-14. Mangroves received 56 kg/ha of nitrogen from 7-40-6 fertilizer. All plants were planted on approximately 60 m (2 feet) centers.

After the March 1980 planting high winds brought 60 cm (2 feet) waves to the site for several days which damaged but did not destroy the planting. On May 28, 1980 the top three rows of _S. alterniflora_ in the tidal area were vigorous while plants below these rows were mostly dead. _S. patens_ plants higher than high tide line were vigorous; those at high tide line were not vigorous. Approximately 20 percent of the mangroves planted away from the intercoastal waterway and somewhat protected by rubble in the tidal zone survived. _P. vaginatum_ grew well in all areas above the high tide line.

Stuart Causeway

Planting and fertilization on the Stuart Causeway was the same as on the Jensen Beach Causeway. The east half of the planting area, at the tide line, was eroding and all the plants within the tide zone were destroyed before they could become established. The west half of the planting area was an accretion area in the tidal zone. The _S. alterniflora_ was being covered up by the accumulating sand but many of the plants are now growing through the sand. _S. patens_ was doing well above the tidal area. _P. veginatum_ had covered the area where planed above the high tide line and by December 1980 the rows were no longer distinguishable (Figure 4).

Basin Bayou Site

Grasses planted at this site were _Spartina alterniflora_ in the tidal zone and _S. patens_ above the tidal zone. _S. alterniflora_ was planted as plugs and _S. patens_ plants were potted. Plants were used on

Figure 3. Remains of S. alterniflora rows, Section 7, Jensen Beach Causeway, November 1979.

Figure 4. Section 7, Replanting of P. vaginatum, S. patens, and S. alterniflora at end of growing season, December 1980.

two different sections, one as the primary erosion control method and the other as a secondary control in conjunction with a permeable structure constructed of concrete blocks tied together with steel rods.

Plants used as the primary erosion control were washed out by wave action and shifting sand before they could become established. Where the plants were used as secondary control S. alterniflora remained in place but was soon covered with sand accumlating inward of the concrete block structure. S. patens which was planted above high tide line was vigorous and doing good at this site.

CONCLUSIONS

Vegetation at these demonstration areas has been at least partially successful to-date. This is especially true where some protection was provided from intense wave action. The plantings are new and time and storm activity will help to determine whether they will increase or deteriorate. We expect that Spartina alterniflora will increase where plants have remained in place long enough for their roots to become established. Without replacement plants it will take several years for these to spread enough to give the desired protection. In areas where there is as much energy as at Stuart-Jensen sites, Rnizophora mangle should be at least 60 cm (2 feet) tall and preferably taller to prevent wave and sea vegetation overtopping when planted in the tidal zone. Paspalum vaginatum established relatively easy and spread rapidly in the Stuart-Jensen Beach demonstration giving good quick protection of the area above the tidal zone. From these observations P. vaginatum should be used more extensively in inland coastal erosion control where adapted.

The U. S. Army Corp of Engineers is preparing a report on all 16 demonstration projects conducted. This report will have information on vegetative and structural costs of the demonstrations.

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VARIATION IN *Thalassia testudinum* SEEDLING GROWTH RELATED TO GEOGRAPHIC ORIGIN

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ABSTRACT

Seedling of <u>Thalassia testudinum</u> Banks ex König from the Florida Keys, Biscayne Bay, and Tampa Bay exhibited distinctive morphogeographical growth patterns under a variety of culture conditions. Root and leaf blade growth was greatest for the southern, Florida Keys population and lowest for the northern, Tampa Bay population; Biscayne Bay seedlings exhibited intermediate growth rates. Leaf widths correlated with growth patterns of cultured seedlings and patterns in the indigenous habitats. Responses to nutrient enrichment were inversely related to shoot growth rates, such that the northern population exhibited the greatest increase in growth, while the southern population responded the least; Biscayne Bay seedlings were again intermediate. These patterns suggest physiological acclimatization of <u>T. testudinum</u> populations to their indigenous environments, a characteristic which may be important when selecting seed stock for revegetation projects. A culture technique utilizing biodegradable compressed peat containers was examined. This method has potential for development of nursery stock and aiding in the anchorage of transplanted seedlings.

INTRODUCTION

Seagrass beds play an important role in coastal marine ecosystems (Wood, Odum, and Zieman, 1969; den Hartog, 1977). Recognition of this importance, and the fact that seagrasses are subject to severe impacts by man's activities in the coastal zone (Thayer, Wolfe, and Williams, 1975), has led to the development of a variety of revegetation methodologies (Knight, Knutson, and Pullen, 1980). Revegetation studies involving the tropical-subtropical species _Thalassia testudinum_ Banks ex Konig have utilized plugs, sods, individaul shoots, seeds and seedlings (Kelly, Fuss, and Hall, 1971; Eleuterius, 1974; Phillips, 1974; Thorhaug, 1974; van Breedveld, 1975; Lewis and Phillips, 1980a). The apparent success of several large scale revegetation projects in Biscayne Bay, Florida, utilizing seeds and seedlings (Thorhaug, 1974, 1979), indicates that the use of seed material may be the most suitable method for mitigation projects involving _T. testudinum_. Lewis and Phillips (1980b) recently proposed that seed material might reduce the cost of seagrass mitigation if large numbers of seeds are periodically available. Compared to destructive transplanting techniques, such as sod or plug removal, the apparent nondestructive nature of collecting seeds suggests that this method should be examined in detail. We stress apparent since the impact of sexual reproduction in maintenance of established beds is unkown.

When considering seed stock for mitigation projects, the possibility of genetic fixing in local strains may be important (Odum, 1971). Geographically separated _T. testudinum_ populations exhibit ecoplastic limits that are adaptive to local conditions (McMillan, 1978, 1979; McMillan and Phillips, 1979). Responses to the influence of habitat include variation in leaf length and width (Phillips, 1960; Zieman, 1974) and variable reporductive patterns (Grey and Moffler, 1978; Moffler and Durako, unpublished data). In this regard, transplanting seeds or seedlings from remote locations may result in failure or poor success since they lack this factor compensation. The possibility of pathogen introduction utilizing nonlocal seeds also suggests that indigenous seed stock would be preferable for revegetation projects.

Seagrass research at the Florida Department of Natural Resources Marine Research Laboratory has involved studies on transplant techniques (Darovec et al., 1975; van Breedveld, 1975) and aspects of the reproductive ecology and physiology of T. testudinum in Florida (Moffler, 1976; Grey and Moffler, 1978; Moffler, Durako, and Grey, 1981). The aim of these investigations is an understanding of this important resource and the development of economically feasible and environmentally safe techniques for successful restoration of destroyed T. testudinum meadows. This paper reports on the differences in growth patterns of seedlings collected from three locations in Florida, representing a latitudinal gradient. Preliminary results of a novel transplanting system will also be discussed.

MATERIALS AND METHODS

Fruits, seeds and seedlings of Thalassia were collected along the Atlantic shoreline of Grassy Key in the Florida Keys (FK) and Matheson Hammock in Biscayne Bay (BB) on 12 August 1980; and along the Sunshine Skyway causeway and Mullet Key in Tampa Bay (TB) on 13 and 14 August. The material was transported to the laboratory cooled in ice chests. Seeds were excised from fruits within one day of collection. Seeds and seedlings were surface sterilized for 10 min in a 5% sodium hypochlorite-seawater solution and held in separate aquaria filled with synthetic seawater (Instant Ocean) at 32 ppt salinity until needed for culture experiments.

Tube Cultures

Tube cultures utilizing marine agar as a substratum were initiated in order to monitor leaf and root growth of seedlings. Four treatments (4 replicates each) were tested using 60 ml culture tubes with 20 ml agar and 30 ml of liquid media. The two liquid media were Instant Ocean and NH-15 (modified from Gates and Wilson, 1960) at 32 ppt; the two types of agar were nutrient agar composed of 6 g/l phytagar in 32 ppt NH-15 and marine agar composed of 6 g/l phytagar in 32 ppt Instant Ocean. The four treatments were combinations of the two types of agar and liquid media: Nutrient agar - Instant Ocean, Nutrient agar - NH-15, Marine agar - Instant Ocean, and Marine agar - NH-15. Seedlings from the three sites were again surface sterilized for 10 min followed by three one minute rinses in sterile synthetic seawater and placed in the culture tubes. All transfers were performed using a laminar flow hood and aseptic techniques. Larger culture tubes (80 ml) were tested to determine of the size of the tubes affected growth. These tubes contained 25 ml of nutrient agar and 45 ml of Instant Ocean. Illumination was provided by Duro-Test Vita Lites on a 14:10 L:D cycle and the temperature was held between 22-29°C.

Pot Cultures

Seedlings were also grown in 5 x 5 x 12.7 cm plastic pots filled with an autoclaved mixture of builders' sand and aragonite shell hash (1:1). Three replicates from each population were placed in two aquaria:

one containing 71 l synthetic seawater (Instant Ocean) and the second containing 71 l synthetic seawater with von Stosch's enrichment (von Stosch, 1964). Salinities of both tanks were maintained at 32-34 ppt and temperatures were held between 24°-26°C with a 14:10 L:D photoperiod cycle.

The remaining seedlings were planted in Peat Pellets (Jiffy-7) and placed in aquaria containing Istant Ocean. Light and temperature conditions were the same as above. After three months in culture, the rotted seedlings were transplanted randomly within a 1 x 1.2 m quadrat in Boca Ciega Bay, and monitored monthly for survival and growth.

Growth Measurements

Leaf blade growth for all treatments was calculated by measuring the length and width of the leaf blades to obtain a total leaf blade area. Root lengths in the culture tubes were visually estimated using a metric scale placed behind the tube. Leaf and root growth rates of the three populations were compared using analysis of covariance and leaf widths were compared using t-tests, significance was determined at $P<0.05$.

RESULTS

Tube cultures provided a unique opportunity to monitor both leaf blade and root growth without disturbing the seedling (Figure 1). Seedlings growth in the tube cultures indicated populational differences (Figure 2). Clear patterns were not exhibited in response to nutrient enrichment of the agar or liquid media (Table 1), so responses were examined with respect to geographic origin only. Biscayne Bay (BB) seedlings had the greatest total root length (Figure 2b), due to the production of the greatest number of roots per seedling (Table 1). When

Figure 1. *Thalassia testudinum* seedlings in tube cultures.

growth of individual roots was considered, root lengths of the FK population increased the most, followed by BB seedlings with the TB population producing the least growth. Some restriction of root growth was evident in the 60 ml tubes and was also indicated by the significantly greater root lengths that were attained by seedlings in the 80 ml tubes (Table 1). In contrast, leaf blade growth was lower for all populations in the larger tubes. In both cultures leaf growth showed an inverse relationship with latitude of origin; the southern FK population exhibited the greatest growth, the northern TB population showed the least growth and BB seedlings were intermediate (Figure 2a).

Growth studies on the 60 ml tube cultures were terminated after 3 months due to contamination of the FK cultures by bacteria, the fungus

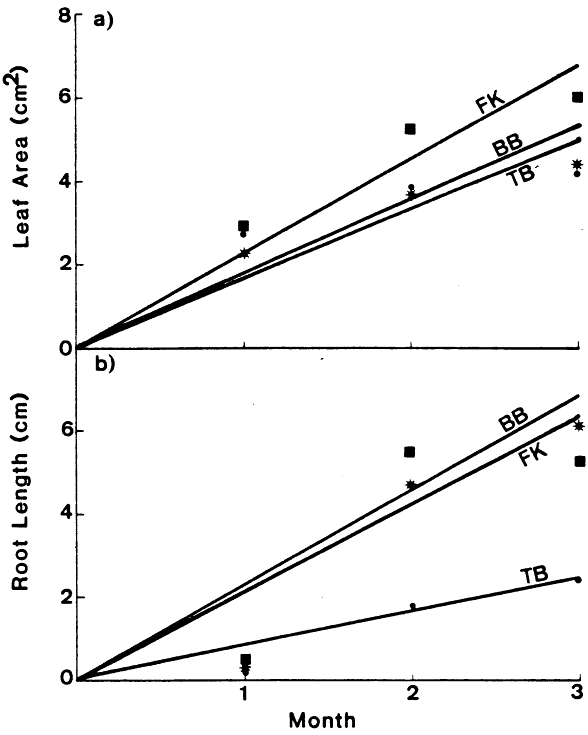

Figure 2. Leaf (a) and root (b) growth of Thalassia testudinum seedlings from Tampa Bay (dots), Biscayne Bay (stars) and the Florida Keys (boxes) in 60 ml tube cultures ($y=a+bx+cx^2$).

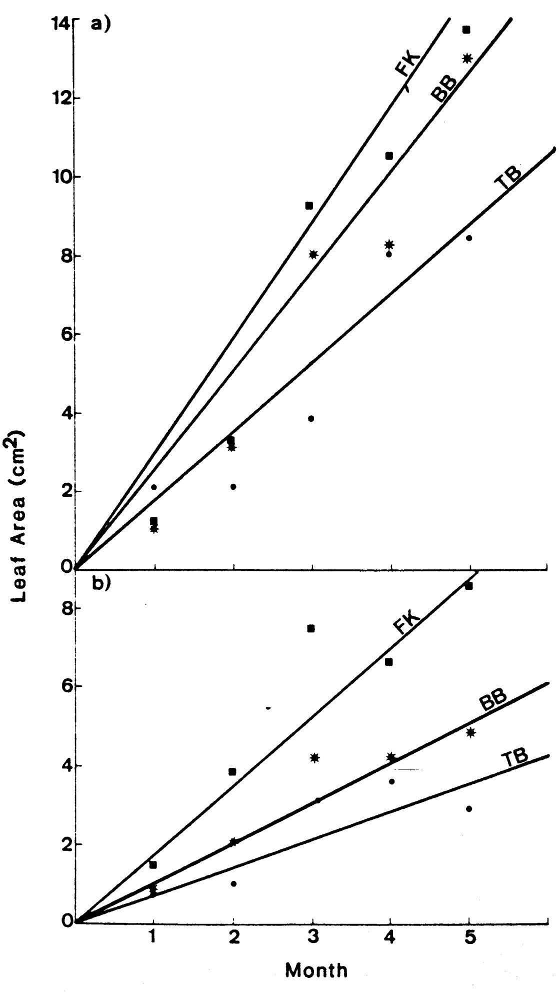

Figure 3. Leaf blade growth of Thalassia testudinum seedlings from Tampa Bay (dots), Biscayne Bay (stars) and the Florida Keys (boxes) in a) von Stosch's nutrient enrichment medium and b) Instant Ocean ($y=a+bx+cx^2$).

Table 1. Root and leaf blade growth of Thalassia testudinum seedlings in agar/seawater cultures after three months.

Treatment	Total Root Length (cm) Tampa Bay	Biscayne Bay	Florida Keys	Leaf Area (cm^2) Tampa Bay	Biscayne Bay	Florida Keys
60 ml tubes*						
NH-15/N.A.	0.95	5.65	4.03	4.32	4.03	6.42
I.O./N.A.	2.48	7.00	6.35	4.48	4.05	6.27
NH-15/M.A.	2.52	6.40	4.75	4.29	4.13	4.58
I.O./M.A.	2.75	5.52	6.00	3.36	5.20	7.00
Mean	2.18	6.14	5.37	4.18	4.35	6.04
Root#/sdling	2.50	3.81	2.47			
Length/root	0.87	1.61	2.21			
80 ml tubes						
I.O./N.A.	3.27	12.10	14.98	3.18	4.21	4.50

*I.O.=Instant Ocean; N.A.=Nutrient Agar; NH-15=Nutrient enriched seawater; M.A.=Marine Agar. Values represent the mean of 4 replicates.

Table 2. Leaf growth rates of Thalassia testudinum seedlings to laboratory cultures.

Treatment	Growth Interval (months)	Shoot Growth Rates (cm^2/mo) Tampa Bay	Biscayne Bay	Florida Keys
Tube cultures				
60 ml tubes	3	1.38	1.47	2.07
80 ml tubes	3	1.08	1.40	1.50
Pot cultures				
Instant Ocean	5	0.72	1.04	1.78
von Stosch's	5	1.76	2.53	2.94
Peat Pellets	3	0.93(1.70)*	1.78(2.52)	1.72(2.80)

*Numbers in parentheses indicate a growth index value where:

$$\text{Growth Index} = \frac{\text{mean leaf area / seedling}}{\text{mean leaf \# / seedling}}$$

Lindra thalassiae Orpurt et al. and high levels of *Leucothrix mucor* Oersted (Dr. Jack Fell, pers. comm.). All of the 60 ml tube cultures had some contamination evident, but only the FK cultures succumbed. The contamination in the FK cultures appeared as a red growth on the seedlings and in the agar. None of the 80 ml FK cultures exhibited this red contaminant and in general, these cultures appeared less contaminated.

Leaf growth of seedlings in the plastic pots repeated the inverse relationship between growth and latitude (Figure 3). Seedlings cultured without nutrients added had lower growth rates than the tube cultures, but the addition of nutrients increased growth rates significantly (Table 2). Responses to nutrient enrichment were inversely related to the levels of growth, such that the TB seedlings exhibited the greatest increase (244%), BB seedlings were intermediate in their response (242%) and the FK population showed the least increase (165%). Nutrient enrichment initiated a phytoplankton bloom which was controlled using diatom filtration.

Leaf growth rates of seedlings in the peat pellets was intermediate between the plastic pot cultures with and without nutrient additions (Table 2). Biscayne Bay seedlings had a slightly higher total leaf area than the FK seedlings, due to a greater number of leaves per seedling, but the growth rates of individual leaf blades (growth index) repeated the patterns of the other treatments. After 3 months in culture, many of the seedlings had roots that protruded through the webbing of the pellets and all were firmly anchored (Figure 4). The seedlings were transplanted in Boca Ciega Bay in November to an area that had supported seagrasses prior to dredging associated with intracoastal waterway maintenance. When transplants were examined one month later it

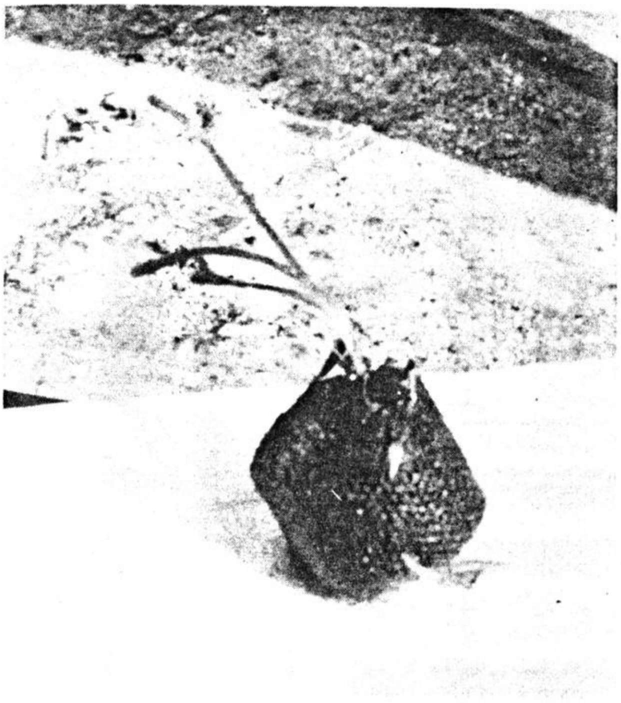

Figure 4. Thalassia testudinum seedling in peat pellet after 3 months in laboratory culture.

was clear that this site was still unsuitable for revegetation using T. testudinum because of high sedimentation rates and associated turbidity. Only three seedlings were still alive, the remainder were coated with sediments. Additionally, the site appeared to have been physically disturbed. In March several peat pellets were retrieved and it was observed that they had maintained their integrity after being submerged for over 6 months.

Leaf blades were generally widest for the FK population and narrowest for TB seedling, with BB seedlings exhibiting intermediate widths (Figure 4). The differences in leaf widths between treatment were not statistically significant and indicated that nutrient enrichment had little effect on this morphological characteristic, so increases in leaf area in response to fertilization could be attributed to leaf elongation.

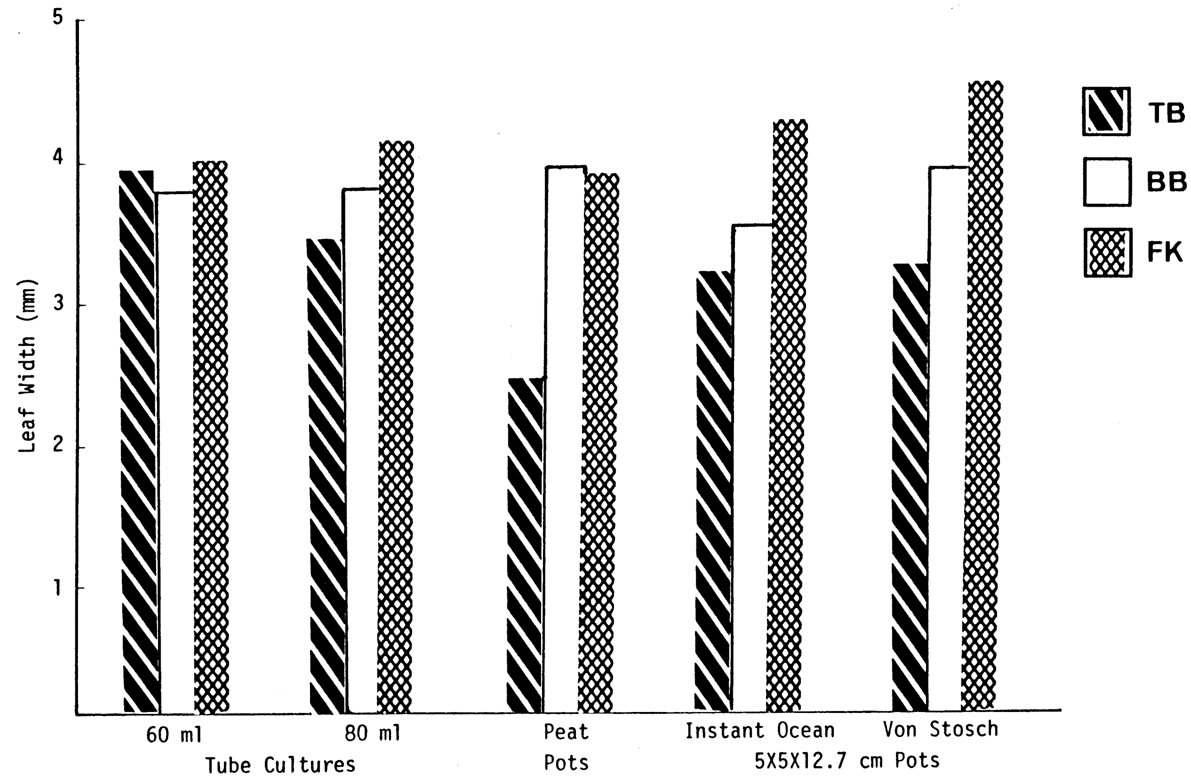

Figure 5. Leaf blade widths of *Thalassia testudinum* seedlings from Tampa Bay (TB), Biscayne Bay (BB) and the Florida Keys (FK) after 3 months in culture.

DISCUSSION

This study demonstrates that *Thalassia testudinum* seedlings from three locations, representing a latitudinal range of only 320 kilometers, exhibit distinctive physiological and morphological features under controlled conditions. Previous investigations have shown leaf widths and cold tolerances to be distinctive in widely separated *T. testudinum* populations (McMillan, 1978, 1979). Morphological variation in leaf width indicates a response to environmental conditions (Phillips, 1960; Zieman, 1974), and maintenance of these patterns under controlled conditions suggests that the limits are genetically modulated (McMillan, 1978). Differing root and leaf growth patterns exhibited by seedlings from Tampa Bay (TB), Biscayne Bay (BB) and the Florida Keys (FK) also suggests genetic differences. Species with wide geographical ranges almost always develop locally adapted ecotypes (Odum, 1971), yet seedlings of *T. testudinum* from diverse habitats show little variation in isozymes (McMillan, 1980). This lack of isozyme variation contrasts with the differentiation found within a single population of *Spartina patens* (Silander, 1979), and between populations of *Uniola paniculata* (David Crewz, pers. comm.).

Root growth in the culture tubes was similar to that reported for *T. testudinum* seedlings growing *in situ* (Thorhaug, 1974). The absence of differential growth in response to nutrient partition suggest either that agar provides an adequate nutrient source over short periods or that seedlings contain sufficient nutrients to support 3 months growth, a supposition that is not supported by the responses in the plastic pot cultures. The smaller 60 ml culture tubes evidently reduced root elongation, and may have increased the deleterious effects of contamination.

Increased root growth in the 80 ml culture tubes, with accompanying reduced leaf growth, indicates that *T. testudinum* allocates a greater proportion of its biomass below ground when growth is not restricted. Dawes and Lawrence (1980) proposed that this improves the anchoring ability of the plant, while Barko and Smart (1978) suggested that this characteristic improves survival in infertile habitats by increasing the absorptive and/or storage capacity. The adaptive significance of the BB population's greater root production is puzzling since this population does not exist in a high energy area where anchorage would be a problem, although it may represent an adaptation to the bedrock substrate which is common in Biscayne Bay (Zieman, 1972; Wanless, 1976).

Infection patterns provided further evidence of populational differences in tube cultures. Florida Keys seedlings were the most contaminated and harbored differing contaminant species than the TB and BB seedlings. The pathogenic nature of contamination in culture systems may be due to stresses of the artificial conditions which are conducive to disease development (Fell, 1976). Yet, the possibility of introducing non-indigenous pathogens when transplanting seedlings *in situ* must be considered. Surface sterilization alone was not effective in eliminating contaminants. This treatment coupled with antibiotics may reduce the possibility of pathogen introduction when transplanting non-local seedlings, and should be considered in pilot mitigation projects.

Leaf blade growth and the widths of leaves in laboratory treatments also showed an inverse relationship with latitude of orgin. McMillan (1978) suggested that differing leaf widths of geographic variants of *T. testudinum* indicate an ecotypic response, but as stated previously this has not been confirmed in isozyme studies (McMillan, 1980). Nutrient enrichment significantly increased leaf blade growth in the shell/sand plastic pot cultures. Fertilization might be used to facilitate establishment and coverage at revegetation sites so less material is initially needed. It is interesting that growth responses to nutrient enrichment moderated populational distinctions. Different T. testudinum populations may have similar photosynthetic or growth capacities, but they are attained under conditions related to their indigenous environments. This phenomenon has been observed in several marine algal species (King and Schramm, 1976; Durako and Dawes, 1980).

Anchoring seagrass transplants has proved to be a major problem in revegetation work. The number of anchoring and propagation techniques that have been developed reflects the magnitude of this problem (see bibliography by Knight et al., 1980). Phillips (1980) recommended plugs as the single most important method of transplantation, since many seedlings are lost in the field. However, the plug method is very destructive to the source beds since T. testudinum is slow to propagate into disturbed areas (Godcharles, 1971). This may cause more harm than good if plugging is done on a large scale (Steller, 1976). If a suitable anchoring method were available for use with seedlings a nondestructive revegetation program might be developed. Seedlings become firmly rooted in the peat pellets within 3 months and exhibited leaf growth comparable to the other treatments. The pellets provided anchorage and maintained their integrity in situ for over 6 months. No mortality was observed in laboratory peat pellet cultures. Since the transplant site was unsuitable for growth, the field results were inconclusive and more work needs to be done to test the suitability and cost effectiveness of the technique. Ease of sowing, transport and handling of the pellets and seedlings suggests that they may offer a viable alternative to transplanting vegetative material.

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SEASONAL GROWTH OF SEAGRASSES IN SARASOTA BAY, FLORIDA, WITH NOTES ON HISTORICAL CHANGES AND CONSIDERATIONS FOR RESTORATION

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ABSTRACT

Mappings of seagrass distribution in portions of Sarasota Bay, Florida, revealed significant localized declines of seagrass cover averaging 55 - 65% since 1948. Losses were related to several factors including water quality degradation, erosion, and grazing by sea urchins. Observations on seasonal growth of _Halodule wrightii_ and _Thalassia testudinum_, during 1979-1980, suggested that site-specific factors (e.g. degree of tidal inundation, temperature, drift and epiphytic algal growth, herbivore activity, and sediment stability) influence the maximum biomass attainable at a particular location. Several growth characters, including shoot and blade abundance, leaf area, flowering, and above-ground-belowground biomass, were monitored and used to develop a phenological index for these seagrasses in Sarasota Bay. Comparisons of seagrass standing stock and biomass were made between Sarasota Bay and other regions of Florida, the Gulf and Atlantic coasts of southern U. S., and the Caribbean. A non-destructive means of estimating biomass from _in situ_ measurement of plant parameters was introduced.

INTRODUCTION

The importance of seagrasses in coastal environments has been reviewed by many investigators throughout the world (Phillips and McRoy, 1980). Seagrasses exhibit high productivity, stabilize substrates, promote sedimentation, act as nutrient pumps, and support a diverse assemblage of organisms. Although these benefits are recognized, several paradoxes concerning the role of seagrasses in coastal ecosystems are also apparent (den Hartog, 1980). For one, it seems that although many organisms live in or on seagrasses and use the beds as shelter and stable substrate, few organisms utilize the grass directly for nutritive value. Rather, the great bulk of material present in any seagrass meadow, in any single seagrass plant, at any time is destined for detritus. In fact, as detritus is decomposed by the action of physical break-up and microbial activity, many organisms graze the microbial layer as a source of preferred nutrition (Frenchel, 1977).

The actual fate of the primary production in seagrass ecosystems and in coastal wetlands in general (e.g. internal degradation, utilization and export as particulates, dissolved organics, and living biomass) is one of the most important, yet poorly understood, aspects of wetland ecology (Thayer et al., 1979). The true importance of seagrasses may lie in the realm of those subtle connections to other components of the seagrass community. In this sense, seagrasses may provide a matrix or <u>community frame</u>, which functionally links a great many organisms and processes of otherwise disparate yet adjacent communities.

In order to make assessments and predictions of impacts of man's activities on entire ecosystems or any of their components, we must understand how natural systems operate, including their range of natural variability and the environmental factors which control this variability (Cross, 1979). This will vary between habitats and seasons. There currently exists little or no basis for discriminating between the ecological value of different seagrass species or their life reguirements. Furthermore, there exists the need for a set of monitoring tools which can be used to track the development of seagrass populations.

This paper reports the results of an investigation into the seasonal variation and life history aspects of the dominant seagrasses in Sarasota Bay, Florida: Shoalgrass, *Halodule wrightii*, and Turtlegrass, *Thalassia testudinum*. Further work is outlined dealing with long term changes of seagrass cover in localized regions of Sarasota Bay and the probable cause of the recent declines. The results of these investigations are used to assess the suitability for undertaking seagrass restoration activities.

METHODS AND MATERIALS

Seasonal Growth

Replicate plug samples were harvested from monotypic zones within *Halodule* and *Thalassia* beds at three stations in Sarasota Bay (Figure 1) at approximately 3 - 4 week intervals for a period of one year (March, 1979 - March, 1980). Plugs of plant material, with sediment intact, were removed with a standard post-hole digger ($0.02m^2$ cross section to a depth of 15cm) and washed free of all adhering sediment and epiphytes. *Thalassia* plant material was sorted into several categories: young blades (8mm), mature blades (8mm), attached dead and detritus, short shoot bases, bladeless sheaths, roots and shizomes, and rhizome apical meristems. Replicate samples were processed independently for the

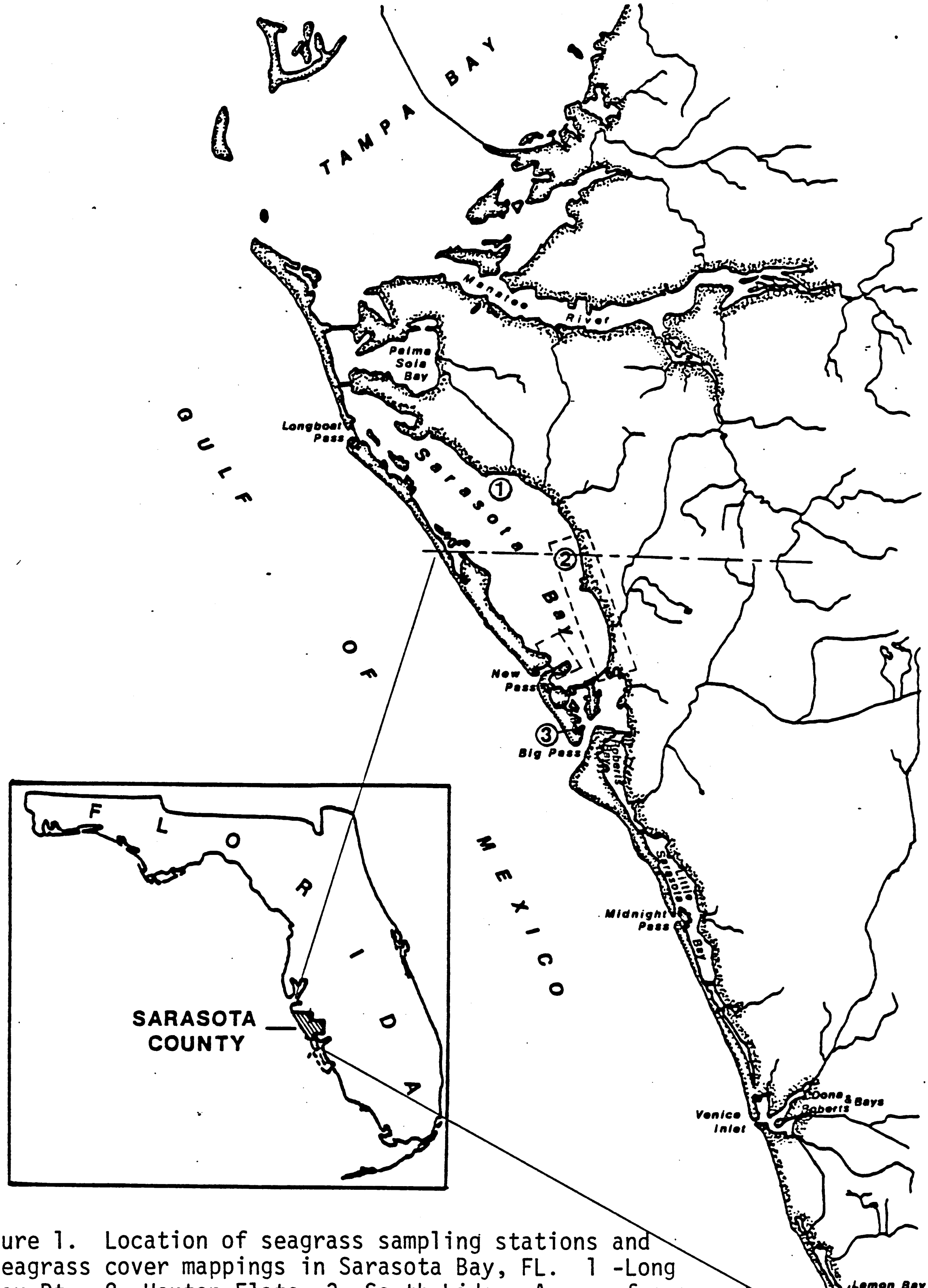


Figure 1. Location of seagrass sampling stations and seagrass cover mappings in Sarasota Bay, FL. 1 -Long Bar Pt., 2 -Horton Flats, 3 -South Lido. Areas of seagrass cover mapping are delineated by dashed line. Detailed site descriptions can be found in Sauers (1980).

following parameters: shoot number, blade number, blade length and width, presence or absence of original leaf tip, rhizome depth, number of leaf scars, intershoot distance, and number of internodes along rhizome. Halodule plant material was sorted into aboveground and belowground material and shoot number were recorded. Biomass was determined after drying samples for 24hrs at 105°C.

Seagrass Cover Mapping

Sources of aerial photographs used to map grass bed configurations were: 1) U.S. Department of Agriculture, ASCS, black and white aerials (1948, 1957, 1969, 1974) at scale 1:7920 (1"=660'); and 2) Mote Marine Laboratory, color aerials (1979) at approximate scale 1:18,500 (1"=154'). Map originals were traced from aerials onto mylar sheets after scale correction to 1:7920 (1"=660'). Area calculations were made using a square grid. Area calculations of seagrass cover shown in Figure 4 were made within 24 arbitrary 1000' sections, which were constructed along the shoreline and adjacent Bay bottom. Area calculations for seagrass cover shown in Figure 5 were made for those grass beds shown in black. Percent change of seagrass cover was calculated using the formula $\% = \frac{X_1 - X_2}{X_2}(100)$ where X_1 = new area and X_2 = old area.

Linear Regression Analyses

Data generated during the seasonal growth study were subjected to linear regression analyses for the purpose of developing a non-destructive means of estimating biomass from it situ measurement of plant parameters (e.g. leaf area and/or shoot density). Data from the period of peak standing crop were used to develop the regressions.

RESULTS

Seasonal Growth

Comparisons of densities and biomass determination for *Halodule* and *Thalassia* at several locations (Table 1) indicate that *Halodule* attained high densities and maximum biomass in Sarasota Bay, which exceeded most published values from other locations. *Thalassia* attained only modest values in relation to other habitats in Florida and the Caribbean. In a subtidal situation, *Thalassia* shoot densities exceeded those found in an intertidal habitat by a factor of 2. On an annual mean basis, *Thalassia* biomass exceeded *Halodule* biomass by a factor of 2. It was found that *Thalassia* biomass varied about 50% throughout the year while *Halodule* biomass varied about 60% between times of seasonal minimum and seasonal maximum biomass.

Seasonal trends of total biomass for *Thalassia* and *Halodule* are shown in Figure 2a. In general, three peaks and three depressions were indicated for *Thalassia* while *Halodule* showed a spring peak followed by a gradual decline throughout the summer and fall months. The apparent anomaly of *Halodule* attaining a peak in August at the South Lido station may be explained as spring epiphytic and drift algae stress delaying the attainment of a biomass peak until later in the season. *Halodule* exhibited none of the multiple seasonal increases and decreases of total biomass as was evident for *Thalassia*.

Seasonal trends of various biomass components of *Halodule* and *Thalassia* are shown in Figures 2b, c, and d. During November, *Halodule* below ground standing crop reached a peak during the time of aboveground standing crop decline (Figure 2c), indicating translocation of energy reserves from aboveground to belowground structures, just prior to

Table 1. Seagrass standing stock and biomass comparisons.

Species	Location	Density (No./m²)	Biomass (g dry wt/m²) Range	Mean	Source**
Halodule wrightii	North Carolina	-	105-200*	200*	Dillon,1971
	Texas	15,118 (shoots)	10-250*	90*	McMahon,1968; McRoy,1974
	Apalachee Bay, FL	-	3-133	40	Zimmerman & Livingston,1976
	Anclote Anchorage, FL	1205 (blades)	-	17	Thorhaug *et al.*,1977
		6766 (blades)	-	289	Zimmerman *et al.*,1972
	Tampa Bay, FL	-	-	334	Levine, 1980
	Sarasota Bay, FL	3500-13,000(shoots)	154-512	320	Sauers, 1980
	Biscayne Bay, FL	-	100-200*	-	Edwards,1977
Thalassia testudinum	Texas	-	60-250	150	Odum,1963; McRoy,1974
	Apalachee Bay, FL	-	408-3204	1240	Zimmerman & Livingston,1976
	Anclote Anchorage, FL	110-475(blades)	-	-	Ford *et al.*,1975
		553 (blades)	-	301	Zimmerman *et al.*,1972
		214 (blades)	-	32	Thorhaug *et al.*,1977
	Tampa Bay, FL	-	-	396	Levine, 1980
	Sarasota Bay, FL	750-3300(blades)	358-1216	740	Sauers, 1980
		400-1100(shoots)	? -1500	530	Morrill *et al.*, unpubl.
	Florida (west coast)	-	75-8100	500-3100	Bauersfield *et al.*,1969 Phillips,1960;Taylor *et al.*, 1973;Zieman,1974
	Florida (east coast)	235-5469(blades)	20-1800	125-800	Odum,1963;Jones,1968; Thorhaug & Roessler,1977; Zieman,1975
	Cuba	-	30-500*	350*	Buesa,1972,1974; Buesa and Olaechea,1970
	Puerto Rico	7376 (blades)	60-560	80-450	Burkholder *et al.*,1959; Margalef & Rivero,1958; Thorhaug & Schroeder,1977
	Jamaica	4055 (blades)	-	-	Greenway,1974

*Biomass of blades only. **Cited in Phillips and McRoy (1980) or see Literature Cited.

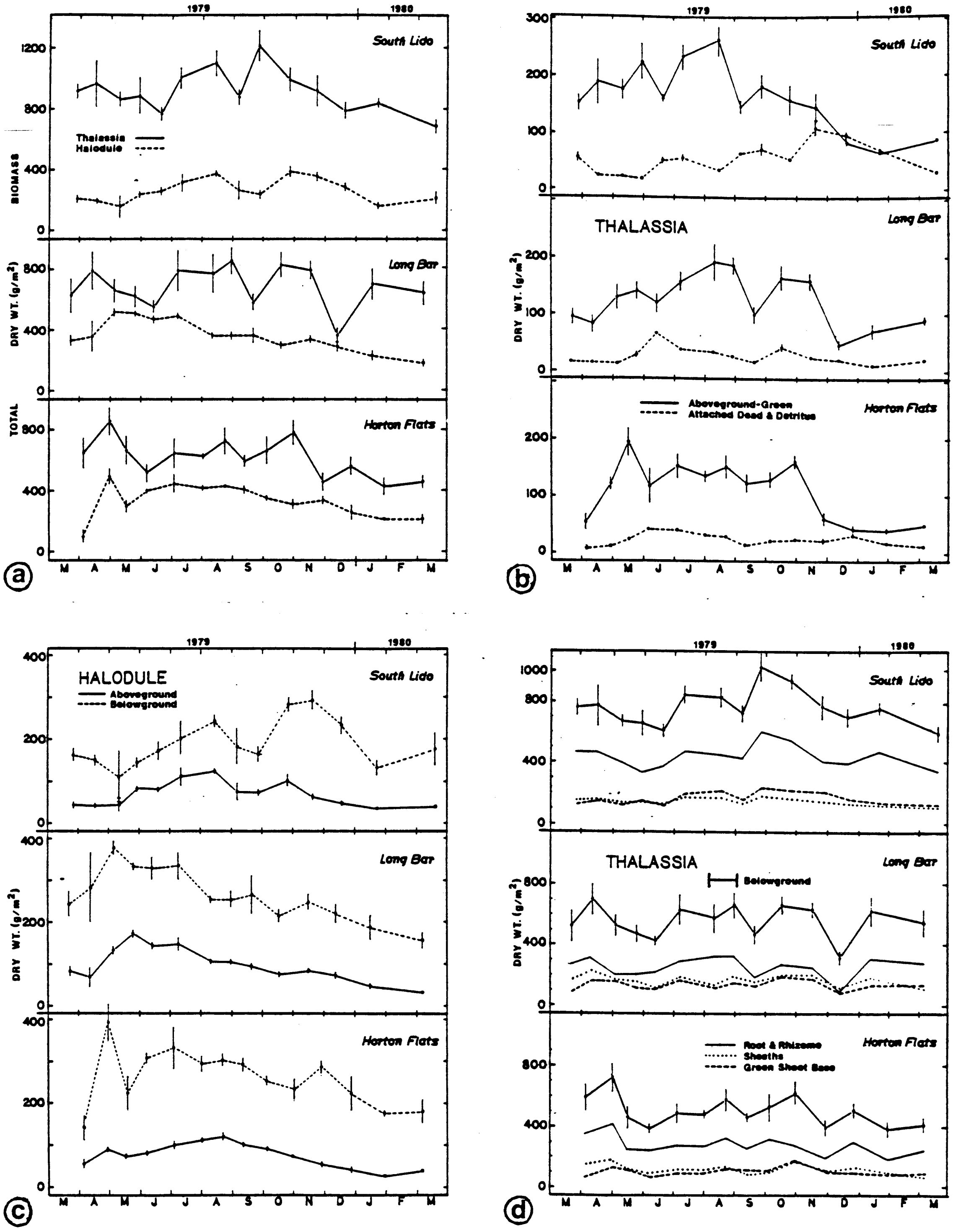


Figure 2. Seasonal biomass changes of Thalassia testudinum and Halodule wrightii at three locations in Sarasota Bay, FL. a) Total Biomass, Thalassia and Halodule. b) Thalassia aboveground (green) blade material and attached dead-detrital material. c) Halodule aboveground and belowground standing crop. d) Thalassia belowground material. Vertical bars represent ±1 SE of the mean calculated for n=5 (Thalassia) or n=3 (Halodule) replicate samples.

winter dormancy. By comparing Figures 2b and d, one can see similar patterns for Thalassia during April-May and December periods. In spring, energy reserves were translocated from belowground to aboveground structures, coinciding with the surge in leaf growth (Figure 3a). In December, energy reserves were moved aboveground to belowground structures.

Thalassia exhibited a characteristic decline in aboveground standing crop (Figure 2b) and leaf area (Figure 3) by June at all location following the spring peak. During the spring peak, blade densities reached maximums of 7 blades per shoot. Following this peak, and coinciding with the declines in leaf area and aboveground standing crop, large accumulations of detached blades were observed on shore adjacent to the study sites. This was precisely the time when the early stages of flowering were observed, as buds were being produced at some locations in mid May and anthesis (blooming) was observed by early June (Table 2). By September, another decrease in aboveground and belowground standing crop was observed at all stations. This decline coincided with a period of high water temperatures, 37°C (98 F), and loss of blade material was indicated at all stations (Figure 3).

Seagrass Cover Changes

Comparison of 1948 and 1979 seagrass cover in the vicinity of Whitaker Bayou in Sarasota Bay, FL is shown in Figure 4 and area estimates are listed in Table 3. This region of Sarasota Bay has experienced a 55% decline in seagrass cover since 1948. It is apparent that the greatest losses of seagrass cover have occurred just south of Whitaker Bayou, north of the Ringling Causeway, and near the Stephens Point boat harbor. These areas of seagrass loss lie within the deeper portions of the Bay. Bathymetry in these areas indicate greater than 4' MLW depths.

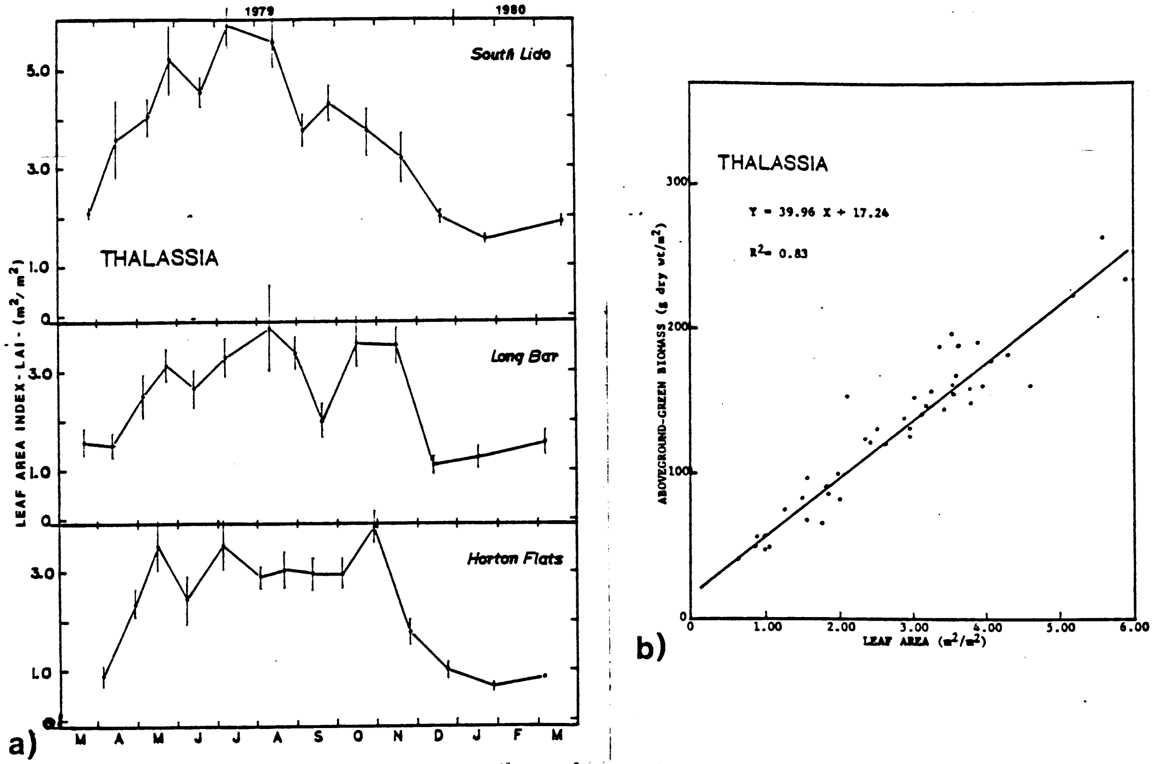


Figure 3. a) Seasonal changes of the Leaf Area Index (LAI) for Thalassia testudinum at three locations in Sarasota Bay. Index calculated as mean green blade area of all leaves in n=5 replicate samples. Vertical bars represent +1 SE of the mean. b) Relationship of leaf area to aboveground (green) biomass for Thalassia testudinum.

In contrast, those sections which show the least declines since 1948 are either distant from the mouth of Whitaker Bayou or lie within shallow regions of the Bay at less than 3' MLW depths.

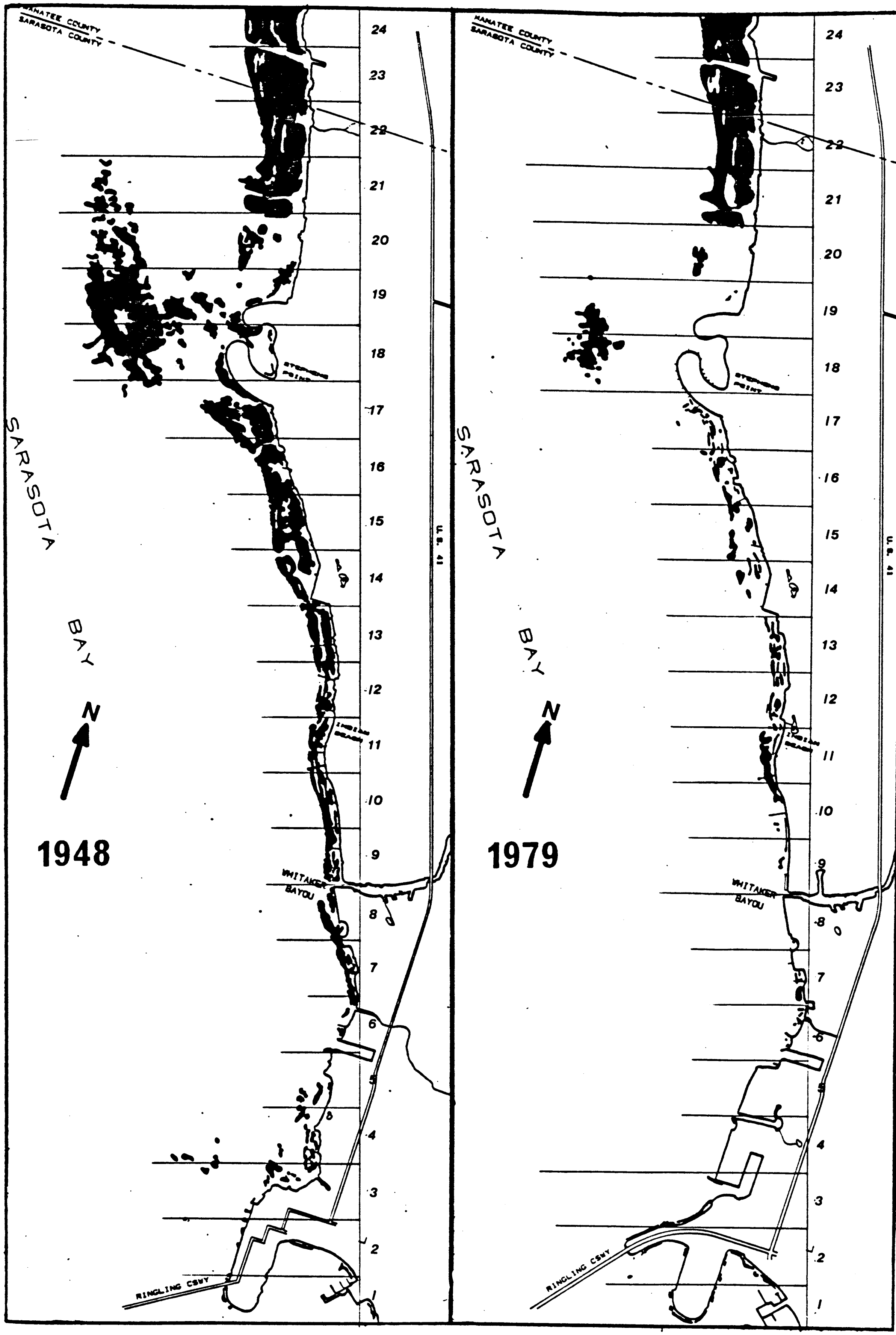
Comparisons of seagrass cover changes in the vicinity of New Pass in Sarasota Bay are shown in Figure 5. Seagrass cover increased approximately 170% between 1948 and 1969 but declined approximately 87% between 1969 and 1979. Sea urchins were abundant in the area in 1974 and were seen to exert grazing impact on these grass beds. In addition, the navigational channel at New Pass was maintenance dredged in 1974 and 1978. Evidence of bottom scour and movement of sediment, perhaps due to increased current velocities in the Pass, can be seen in aerial photographs

Table 2. Phenological Index** for Seagrasses in Sarasota Bay, Florida.

Index Characteristics	*Thalassia testudinum*	*Halodule wrightii*
1. Appearance of new leaves, roots, rhizomes and short shoots	nearly continuous, year-round production, slight winter dormancy indicated	
	new leaf surge in March	
2. Peak of vegetative biomass	site variable May or August	site variable May; July-August or October
3. First evidence of flowering stems	mid May	N.O.*
4. Peak of flowering stem biomass	late May	N.O.
5. First anthesis (blooming)	early June	N.O.
6. First absence of functional flowering stems	late June - early July	N.O.
7. First evidence of fruits	early July	N.O.
8. Peak of seed production	N.O.	N.O.
9. First seed dispersal	N.O.	N.O.
10. Peak of seed dispersal	N.O.	N.O.
11. Autumnal increase in vegetative biomass	late September - October	late October - November
12. Decline of autumn vegetative biomass	December	December
13. First seed germination	N.O.	N.O.
14. Peak seed germination	N.O.	N.O.

*None Observed.
**from Phillips and McRoy, 1980.

(facing page).
Figure 4. Comparison of seagrass cover in the vicinity of Whitaker Bayou; Sarasota Bay, FL; 1948 and 1979. Approximate scale: 1"=2660'. Mapped from aerial photographs taken February 2, 1948 and March 8, 1979.



MANATEE COUNTY
SARASOTA COUNTY
SARASOTA BAY
N
1948
1979
U.S. 41
WHITAKER BAYOU
INDIAN BEACH
RINGLING CSWY
24
23
22
21
20
19
18
17
16
15
14
13
12
11
10
9
8
7
6
5
4
3
2
1

Table 3. Area Estimates of Seagrass Cover in the Vicinity of Whitaker Bayou; Sarasota Bay, FL, 1948 and 1979. Solid line indicates sections near the mouth of Whitaker Bayou. Dashed line indicates sections near Stephens Point.

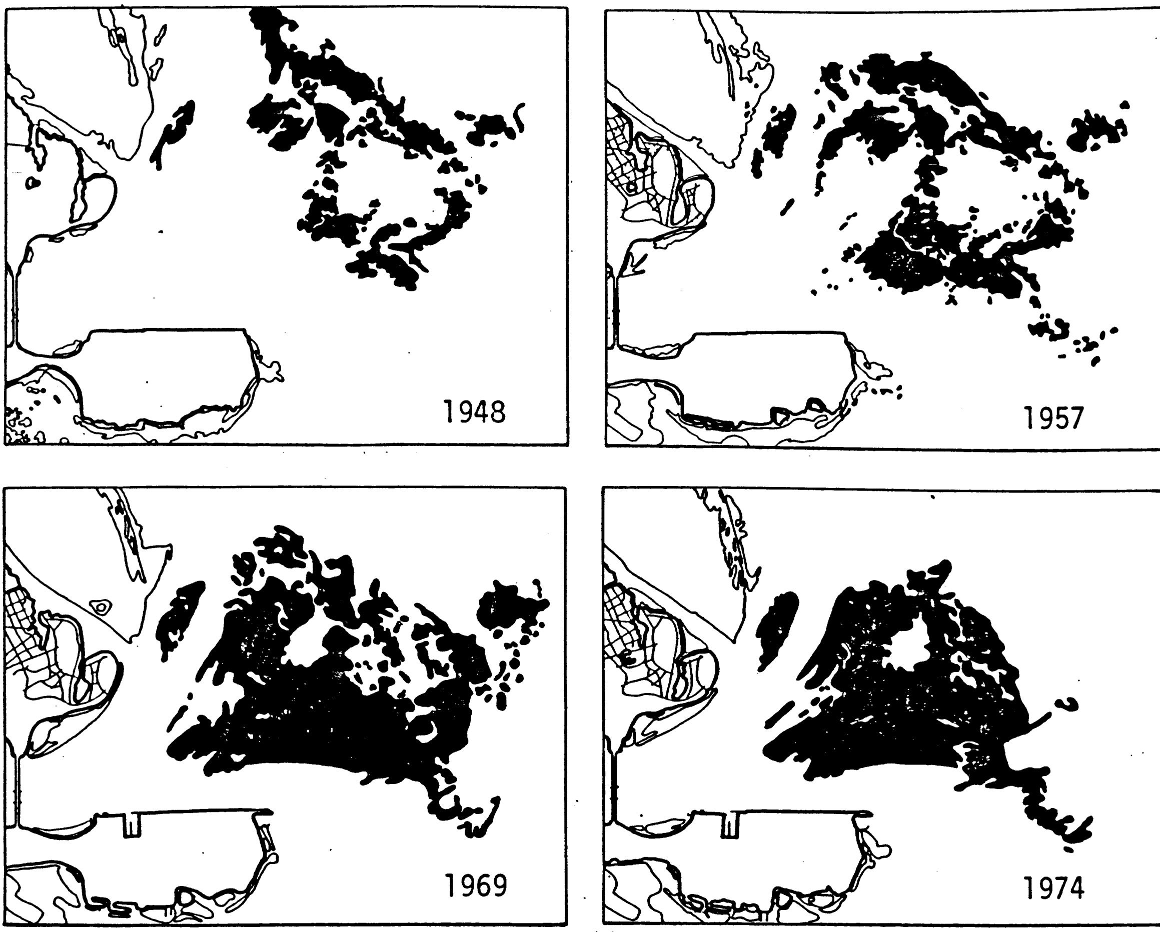
Section	1948 (ha)	1979 (ha)	Percent* Change (Δ%)
1	0.18	0.35	+94.4
2	0.39	0.58	+48.7
3	1.12	0.41	-63.4
4	1.41	0.06	-95.7
5	0.48	0.00	-100.0
6	0.39	0.18	-53.8
7	1.24	0.33	-73.4
8	1.46	0.01	-99.3
9	1.57	0.18	-88.5
10	1.80	0.45	-75.0
11	1.68	1.29	-23.2
12	1.59	0.72	-54.7
13	2.75	0.82	-70.2
14	2.55	0.58	-77.2
15	4.16	0.99	-76.2
16	3.45	0.91	-73.6
17	5.53	0.58	-89.5
18	9.87	2.70	-72.6
19	6.98	1.57	-77.5
20	4.81	0.84	-82.5
21	8.52	4.95	-41.9
22	6.45	5.65	-12.4
23	7.42	6.48	-12.6
24	7.34	7.19	-2.0
TOTAL	83.14	37.82	-54.5

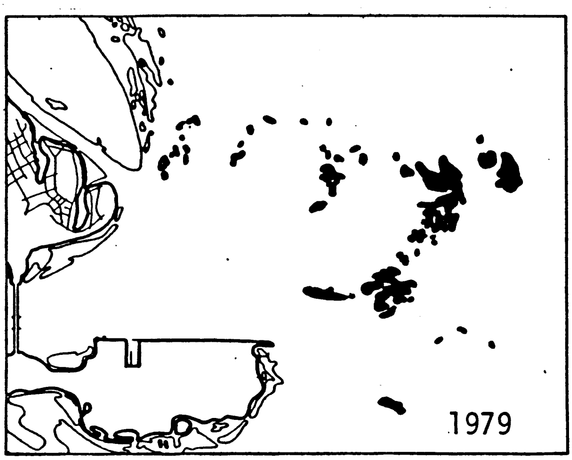
*see Methods.

of the area in 1979. Seagrass cover declines are associated with the loss of suitable bottom habitat (i.e. bathymetry changes) as a result of this scouring.

Non-destructive Biomass Estimates

Data generated during the seasonal growth phase of this study were





Year	Area (ha)	Percent* Change(Δ%)
1948	23.1	
		+50.2
1957	34.7	
		+80.7
1969	62.7	
		-17.9
1974	51.5	
		-84.5
1979	8.0	
1948-1979		-65.4
1969-1979		-87.2

Figure 5. Comparison of seagrass cover in the vicinity of New Pass; Sarasota Bay, FL between 1948 and 1979. See Methods for calculation of percent change*. Area estimates are for those grass beds shown in black. Mapped from aerial photographs taken February 2, 1948; March 23, 1957; December 5, 1969; February 26, 1974; and March 8, 1979.

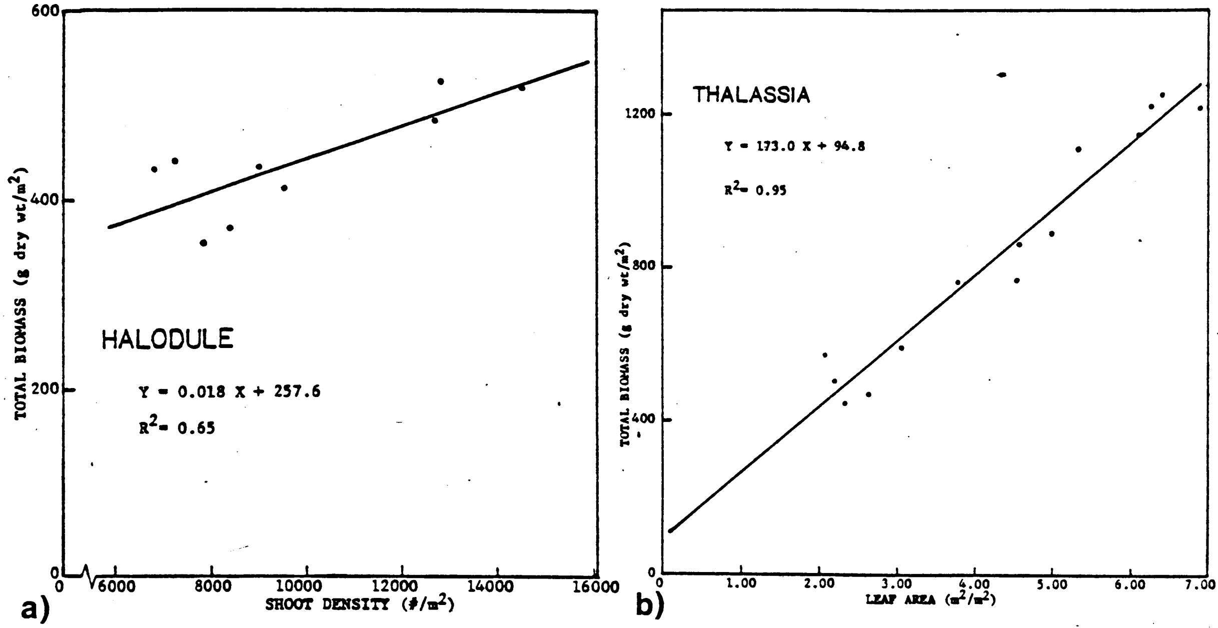


Figure 6. Linear Regression Analyses of Seagrass Biomass. a) Relationship of shoot density to total biomass for Halodule wrightii at peak standing crop. b) Relationship of leaf area to total biomass for Thalassia testudinum at peak standing crop.

subjected to linear regression analyses as depticted in Figure 6. For Halodule, shoot density was regressed against total biomass and the regression equation was significant at the 0.025 level. For Thalassia, leaf area was regressed against total biomass and the regression equation was significant at the 0.0005 level. The coefficient of determination (R^2) for each regression indicated that the fitted regressions accounted for 65% (Halodule) and 95% (Thalassia) of the total variation in total biomass

DISCUSSION

Seasonal Growth

Spatial variation is a typical feature of seagrass growth and it confounds attempts to define patterns of seasonal variation within a population. The high variance associated with sample data makes statistical work-up problematic (Eiseman and Gallaher, 1980). Therefore, a method of making multiple station comparisons and noting whether seasonal trends were _in phase_ at different locations was utilized in this study. It appeared that the seasonal trends reflected species typical variation under the range of environmental conditions encountered during this study.

The late May to early June leaf area depression that occurred at all stations coincided with the time of flowering of _Thalassia_ in 1979. It seems plausible that the switch from incorporation of organic material into vegetative tissue to the shunting of energy into reproductive tissue may explain the depression of leaf area at this time. Although this leaf area depression was seen at all stations, flowering was observed at only two of the three locations. Zieman (1975) observed this same phenomenon in Biscayne Bay, where leaf area decline was seen simultaneously in beds producing flowers and in beds where flowers were not seen. This led Zieman to conclude that the vegetative growth depression may be an innate seasonal rhythm of the plant. It may be that leaf abscission is somehow endogenously programmed to coincide with the shunting of energy reserves into reproductive structures.

Following the June leaf area depression, little increase in leaf area or biomass occurred at the station exhibiting flowering (i.e. Horton Flats). This non-resumption of leaf area increase has been observed at a variety of locations in South Florida (Zieman, 1975).

Seagrass Cover Changes

Declines of seagrass cover have been observed in locations throughout Southwest Florida and in embayments around the Gulf of Mexico.

Lewis et al. (1981) reported an 80% decline in the areal extent of seagrass meadows in Hillsborough County, Florida. Evans and Brungardt (1978) indicated a 25% loss in seagrass cover in Sarasota Bay, between 1948 and 1974. The declines reported for Sarasota Bay in this study seem to indicate the depth distribution of seagrass is shrinking. While seagrass historically extended to the 6' MLW contour, seagrass cover is seldon seen below the 4' BLW contour today in Sarasota Bay. Turbidity, associated with algal blooms and suspended sediment, limits light penetration and may be the dominant factor influencing loss of seagrass cover.

Restoration

Based on the seasonal trends of seagrass growth described in this study, an assessment of optimal planting times for seagrasses can be made. *Thalassia* growth exhibited multiple seasonal biomass variations while *Halodule* growth appeared less erratic with a gradual increase and slow decline throughout the year. From this, one might infer a broad tolerance of environmental extremes for *Halodule* and greater sensitivity to ambient conditions for *Thalassia*. Both species exhibited active vegetative growth in the spring (April-May) and early summer (June-July) while declines in vegetative structure occurred from late summer (August-September) through fall (October-November) and into winter (December-February). The late summer biomass declines are particularly characteristic of *Thalassia*, which may be indicative of heat stress. Therefore, early spring (February-May) appears to be the optimal season in which to plant these seagrasses. Late winter-early spring planting of transplanted stock may be best for taking advantage of the dormant state which seagrasses in this region revert to, thus minimizing disturbance to the normal growth pattern induced by digging, transporting, and planting of seagrass material.

Planting time is not the only consideration that must be given a seagrass restoration project. As this study demonstrates, historical changes add an important dimension to the determination of where and when to proceed with a revegetation effort. By noting the trend of seagrass decline in several areas of Sarasota Bay, one might conclude that ambient conditions are not suitable for successful restoration. Problems with substrate stability and a shrinking depth distribution for seagrass (as revealed by this study) suggest that intensive restoration efforts may not be warranted. Alternatively, restoration efforts may need to be confined to intertidal habitat where light penetration is sufficient for growth. However, stresses associated with exposure of intertidal substrate (e.g. cold stress and dessication) during late winter may necessitate planting later in the spring under more moderate conditions.

With the development of regression analyses that relate density or leaf area to biomass, this non-destructive way of measuring growth can be used to monitor changes in existing seagrass beds as well as to assess the development of revegetated plots.

CONCLUSIONS

In any effort to effectively manage an existing resource, one must <u>define</u> and <u>assess</u> as many aspects of the resource as time and expertise permit. Several tools are introduced in this paper for use in such monitoring (e.g. non-destructive biomass estimates and synoptic mapping). Restoration efforts should include consideration of several aspects of the community in order to determine the suitability for undertaking revegetation activities. Seasonality, tidal fluctuations and temperature regimes, water quality, algal growth, herbivore activity, and physical alterations to adjacent habitat (e.g. dredging) influence the growth of seagrass. All must be given consideration in order to determine which direction the resource is moving and to assess the potential for enhancing the resource.

Efforts should be made to reduce the imput of nutrient-rich wastes to Sarasota Bay, since such imputs support blooms of phytoplankton and algae which contribute to high turbidity levels. These impacts result in declining seagrass stock and loss of benefits associated with the healthy functioning of the seagrass community.

ACKNOWLEDGEMENTS

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A PILOT PROJECT FOR WETLANDS CONSTRUCTION ON THE FLOODPLAIN OF THE ALLEGHENY RIVER IN CATTARAUGUS COUNTY, NEW YORK

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ABSTRACT

Construction of a 21 km section of expressway by the New York State Department of Transportation (NYSDOT) in Cattaraugus County will result in the destruction of 17.5 ha of wetland. As mitigation, NYSDOT will construct 31.5 ha of wetland within the highway right-of-way. It is expected that this project will enhance aesthetic, wildlife, flood protection, nutrient retention, and primary productivity values of the ecosystem.

A pilot project involving construction of 1.8 ha was begun during the 1981 growing season. Native, locally collected emergent macrophytes were planted in 180, 100 m^2 subplots. The effect of water level, substrate, and fertilizer on 16 species is being tested. Control and replication are included in the experimental design. Data has been collected in nearby intact floodplain wetlands to assess water and nutrient retention, primary productivity, alteration of water quality, and production of diversity. These data will be compared to data from the pilot plots.

Highway construction often results in inadvertent construction of wetlands in borrow pits or in interrupted drainage systems. In constrast, the project reported here seeks to construct inland freshwater wetlands, de novo, as a mitigation measure. Singular, salient features embodied in the project are replication and control, use of wetland soil as substrate, and measuring success by comparing functions between experimental plots and natural wetlands.

INTRODUCTION

Environment Consultants, Inc. (ECI) is conducting a pilot Wetlands Development Test Program on the floodplain of the Allegheny River in Cattaraugus County, New York, to implement mitigation procedures as part of the U. S. Department of the Army, Corps of Engineer's (USCOE) requirements under Section 404 of the Federal Water Pollution Control Act Amendments of 1972. This study concerns the proposed construction of a 21 km section (5M, 5N, and 5P) of the Southern Tier Expressway (STE) in the Allegheny River valley between the city of Salamanca and the town of Allegany in southwestern New York by the New York State Department of Transportation (NYSDOT).

Background

NYSDOT plans dating from the 1950s optimistically envisioned traffic flowing over the entire 396 km STE (Hwy 17 between Binghamton, New York and Erie, Pennsylvania) by 1971. The complete construction of this expressway was delayed because of a unique combination of factors: money, problems of traversing the Allegany Indian Reservation, and incomplete environmental data. Early failures to reach an agreement on the design of a program to produce the necessary environmental data resulted in the formation of an STE Steering Committee, an Advisory Committee, and a Task Force with an overall goal to serve as one team to facilitate siting of the STE with the least environmental impact feasible. This culminated in 1979-1980 environmental studies (ECI, 1979, 1980) in design changes, and in a mitigation plan (NYSDOT, 1980) acceptable to all concerned groups. On February 3, 1981, a Section 404 permit was issued.

Mitigation Goals

The permit contains the condition that implementation of all features of the NYSDOT (1980) "Mitigation Plan" be undertaken. As set forth in this Plan, the STE project will directly impact 17.5 ha of wetlands comprising approximately 3.5% of the total wetlands in the study area (ECI, 1980; NYSDOT, 1980). As part of mitigation, NYSDOT is conducting the pilot project described herein in cooperation with representatives from the Federal Highway Administration (FHWA), USCOE, U. S. Fish and Wildlife Service (USFWS), Environmental Protection Agency (EPA), and New York State Department of Environmental Conservation (NTSDEC) to determine the most appropriate approach and techniques to use in developing 31.5 ha of replacement wetlands.

The study (ECI, 1981), which began in early 1981, involves a 3-year demonstration project consisting of the construction of small pilot wetlands (1.8 ha) to be located near two streams--Birch Run and Birch Run Creek--within the STE right-of-way and flood easement (Figure 1).

MATERIALS AND METHODS

Technology for freshwater wetland construction for increasing wildlife populations for New York is well developed (Brumsted, 1954; Atlantic Waterfowl Council, 1959, Emerson, 1961). During the 1970s, considerable effort went into the reconstruction of brackish coastal wetlands while comparatively little effort was expended on freshwater wetland development (Kadlec and Wentz, 1974; Wentz et al., 1974; Garbisch, 1977; Cole, 1979, 1980). The notion of replacing wetland as a mitigation measure, or for maintenance of functional values, such as nutrient cycling and habitat diversity, has evolved much more recently (Erickson et. al., 1978; Good et al., 1978; Swanson, 1979).

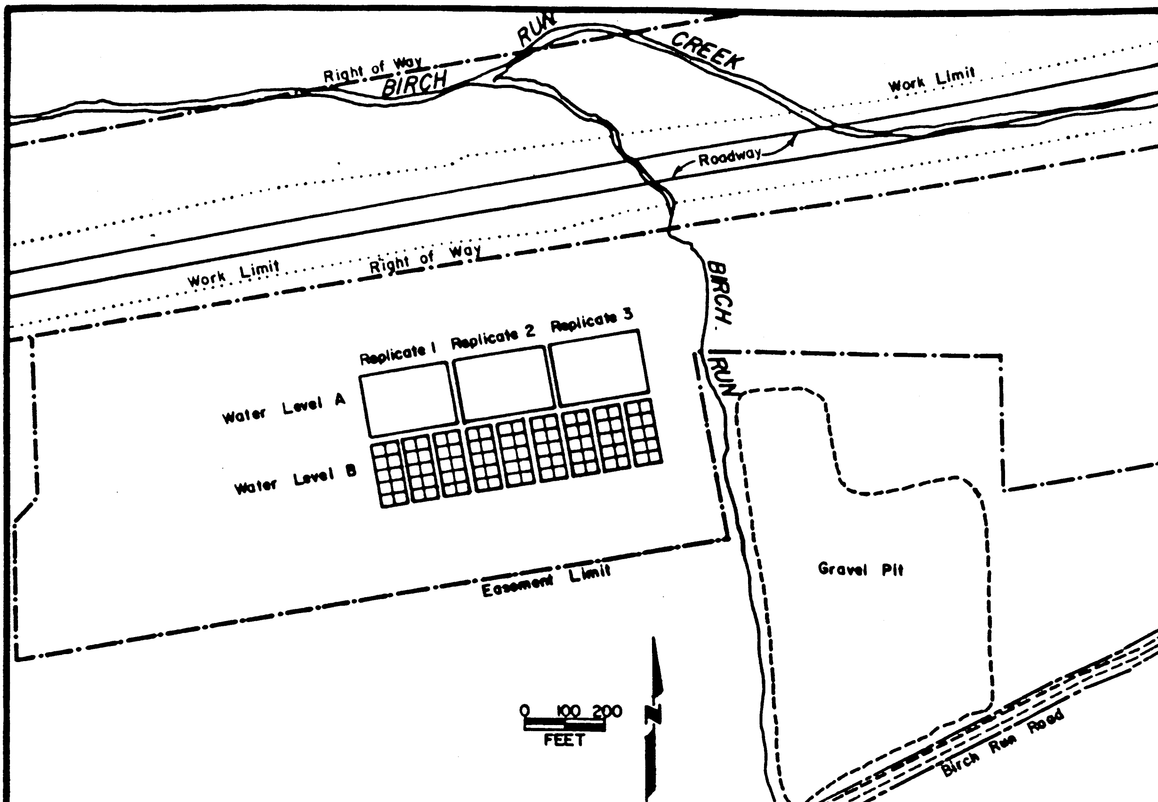

Figure 1. Map of Wetland Development Test Program.

Replication	1,2,3					
Water Level	A			B		
Substrate	1	2		1	2	
Fertilizer	A	A	B	A	A	B
Planting and Species	1-10	1-10	1-10	1-10	1-10	1-10

Figure 2. Conceptual scheme of hierarchical arrangement of plots in the Wetland Development Project. Each replicate contains 60 subplots.

Test Plot Design

Test plots were designed to provide a hierarchial scheme of tests which combines four major variables. Figure 2 shows two water levels, two substrate types, two fertilizer regimes, and various combination of planting method and species choice. In total, this scheme provides 60 different tests which, with replication in triplicate, involves 180 individual test subplots covering 1.8 ha.

The physical arrangement of these plots is shown conceptually in Figure 1. Eighteen plots were constructed, nine with shallow water (30 cm) and nine deeper (61 cm) when full. The 18 plots were divided into 10 subplots each to allow for species and planting method tests. Each plot is 20 x 50 m and each subplot 10 x 10 m.

The literature on freshwater wetland construction indicates the importance of several factors in the design and development of wetlands. These include: water chemistry, water level, draw-down, soil substrate, fertilizer, plant species, planting method and timing, and damage by vertebrates.

Water Chemistry. Dane (1957) and Emerson (1961) have discussed the importance of alkaline waters ($CaCO_3$ and $MgCO_3$ rich) for the productivity of western New York wetlands, and they have emphasized that this reflects soil lime levels. Water, supplying the constructed wetlands, is supplied by Birch Run, which, during the summer of 1981 had alkalinity related to carbonate of 60 ppm. Such waters were considered to be indicative of high fertility by Emerson (1961).

Water Level. Brumsted (1954) has indicated that water depths less than 60 cm are optimal. In this study, spring depths of 30 and 61 cm are planned. Water is supplied from Birch Run through imput canals and culverts.

Draw-down. Draw-down, consistently, has been shown to be an effective technique to stimulate vegetative development in emergent wetlands (Brumsted, 1954; Kadlec, 1962; Linde, 1969). In the constructed wetlands, water loss is expected during the dry mid- to late-summer period. Partial recharge during this period will occur from precipitation and the water table. The wetlands will thus experience annual draw-down during the last half of the growing season, a situation imitating that in many floodplain wetlands within the STE study area. A system of inlet canals, culverts, and drainpipes is used to regulate water level in case of unusually dry or wet seasons.

Soil Substrate. Substrate has been shown to be a significant factor in the vegetative cover in developed emergent wetlands (Emerson, 1961: Shuey and Swanson, 1979). Two types of soil--wetland soil removed from areas to be impacted by the work limit of the STE, and upland soil from the area of the demonstration project--were placed in the ponds.

Fertilizer. One half of the upland soil plots were fertilized with water soluble 10:20:20 which was broadcast at a rate of 250 lb/ha. The ponds were rototilled after broadcasting to reduce leaching of fertilizer into the water. No lime was added because soil analysis showed lime levels in excess of 2,470 lb/ha.

Plant Species. Only native wetland plant species found within the Alleghany River valley ecosystem were planted (Table 1). These were selected because they are aboundant and provide dense cover in their natural habitat. Figure 3 shows the planting scheme employed in each plot.

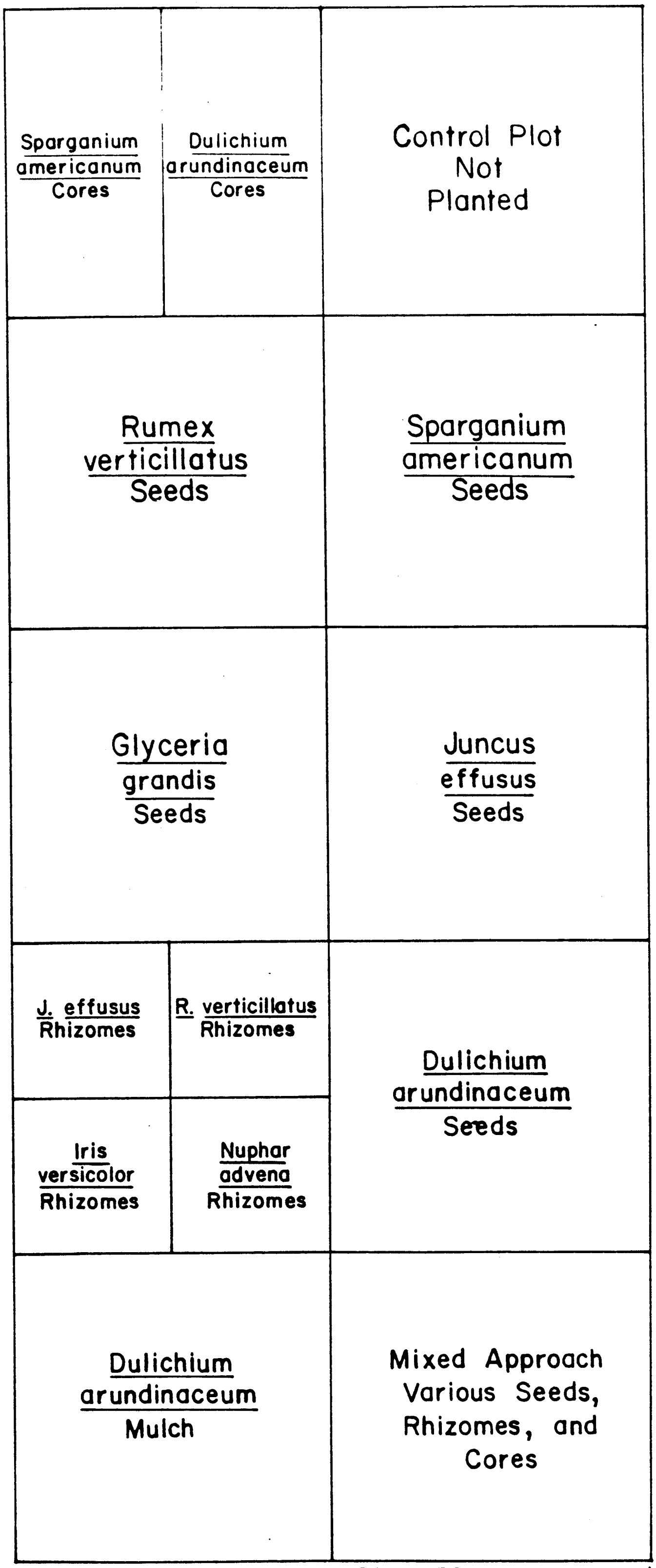

Figure 3. Planting scheme of one 20 x 50 m test pond. Subplots are 10 x 10 m except for subplots devoted to rhizomes and cores.

Table 1.
Hydrophytes and propagation methods for experimental plantings

Species	Mixed Planting	Pure Stand	Propagation Method			
			Seed	Rhi-Zomes	Cores	Mulch
Alisma plantago-aquatica	x		x			
Asclepias incarnata	x		x			
Carex lacustris	x			x		
C. vesicaria	x		x			
Cicuta maculata	x		x			
Dulichium arundinaceum		x			x	x
Glyceria canadensis	x		x			
Glyceria grandis	x	x	x			
Iris versicolor	x	x	x	x		
Juncus effusus	x	x	x	x		
Nuphar advena	x	x		x		
Polygonum hydropiperoides	x				x	
Ranunculus septentrionalis	x		x			
Rumex verticillatus	x	x	x	x		
Scirpus lineatus	x		x	x		
Sparganium americanum		x	x		x	

Planting Methods and Timing. Six planting methods were used: seeds, rhizomes, cores, mulch, natural generation, and a combination approach. seeds and rhizomes were collected locally. All planting was during October and November after hard frost but before soil or water were frozen. Seeds of *Glyceria grandis*, *Rumex verticillatus*, *Ranunculus septentrionalis*, and *Carex vesicaria* were collected in July and stored underwater, following the method of Muenscher (1936). All other seeds were planted within 30 days of collecting and stored dry. Other propagules were collected and planted within hours of collection, or for cores of *Sparganium americanum* stored in the field for one week. Mulching simply consisted of harvesting wetland hay, spreading it on plots, and immobilizing it with a layer of cheesecloth. The papers by Meunscher (1936) and by Kadlec and Wentz (1974) provide much useful information regarding preservation of propagules and planting techniques.

Damage by Vertebrates. Damage by vertebrates, particularly muskrat (*Ondatra zibethica*) which eat wetland vegetation and damage dikes and levees by burrowing, is being controlled or rectified. The best control measure is by exclusion (Linde 1969). The entire test area has been enclosed by a 91 m high, welded-mesh (25 x 50 mm) fence; 30 cm of this fence has been buried. Further muskrat control includes trapping and shooting. Serious depredations by white-tailed deer (*Odocoileus virginianus*) and waterfowl will be rectified by replanting. Damage by carp (*Cyprinus caprio*) is not anticipated.

Other Field Studies

A functional analysis of select in-place wetlands was conducted. This analysis will enable the field team to evaluate the environmental conditions of the constructed wetlands. Although forested, shrub, and emergent wetlands occur within the study area, emergent wetlands were studied, exclusively, because only herbaceous species were planted.

The environmental analysis of the select in-place wetlands included: (1) soil chemistry and structure; (2) water quality; (3) plant species composition and chemistry (nutrients); (4) hydrology; (5) climate; and (6) seed load in wetland soils. The soil analysis included determination of pH, exchangeable H^+, organic matter (%), exchangeable P, K, Mg, Ca, Mn, Fe, Al, NO_3, Zn, and soluble salts, water quality analysis included determination of turbidity, pH, hardness, and macronutrients. Vegetation studies included species diversity, abundance, aerial biomass, and below ground biomass. Plant tissue analysis determined N, P, K, Ca, Mg, Mn, and Fe.

DISCUSSION AND CONCLUSIONS

Highway construction in New York State and throughout the United States often results, inadvertently, in the construction of wetlands in borrow pits or through the interruption of drainage patterns. Many of these uplanned and uncontrolled wetlands are considered to be of low value because they are poorly vetetated, lack diversity, or harbor large quantities of "undesirable" species such as <u>Phragmites communis</u> or <u>Lythrum salicari</u>.

The Wetlands Development Test Program reported here is, by contrast, an attempt to construct inland freshwater wetlands, de novo, as a mitigation measure. Singular, salient features embodied in this pilot project are (1) replication and control, (2) use of wetland soil as substrate, and (3) measuring success with a comparison of the intensities of functions in experimental plots with those in native wetlands. It is expected that this 3-year project will enhance aesthetic, wildlife, flood protection, nutrient retention, and primary productivity values of the ecosystem. Final results will be available in late 1983.

ACKNOWLEDGMENTS

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Transportation or the U.S. Department of Transportation, Federal Highway Administration.

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A HYDROLOGICAL STUDY OF A MOSAIC PINE FOREST/WETLAND

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ABSTRACT

An interdisciplinary investigation is underway in a Gulf Coast estuarine community. This one year study incorporates both biological and hydrological techniques to define the functional value of a complex mixture of upland and wetland species to an adjacent tidal marsh and estuary. Biological evaluations are being made of vegetative species dominance, peak standing crop, and wildlife utilation; hydrologic studies include relative contributions of surface water and ground water to the support of existing vegetation and to export of organic and inorganic nutrients to the adjacent tidal marsh. In addition, data are being collected on the physical and chemical characteristics of soils and water.

The study, which is designed within the framework of an ecosystem model, focuses on the determination of detritus and nutrient export from the forest community into an adjacent estuary, the sources of these materials, and the fluctuations in quantities exported by various transport routes.

Biological studies include vegetative mapping from ground truthed I-R photography, standard plant and animal surveys for species composition, peak biomass determinations, and overall ecosystem description. Hydrologic investigations include monitoring of rainfall, ground water, and surface water fluctuations through appropriate monitoring gages, shallow wells, and a modified Parshall flume. Composite surface and ground water quality measurements are also being collected in a defined drainage system over the course of the study.

INTRODUCTION

This study, initiated to satisfy Corps of Engineers, U. S. Fish and Wildlife Service, and National Marine Fisheries Service data requirements, is designed to provide information regarding the ecological importance of a Gulf coast pine forest, so that those agencies can make decisions on future applications for Section 404 permits to clear and fill portions of this land.

At the time of this writing, early field studies have been completed and are scheduled to continue through early 1982, or until sufficient information is available to characterize the ecological importance of the study area.

Ecological importance will be determined by assessing the value of the area: a) as a source of water, organic and inorganic nutrients, and other materials to an adjacent tidal marsh; and b) as *in situ* habitat for plants and animals; and as c) a buffer zone having flood storage or other wetland values for the surrounding area. The principal focus of the study will be the field monitoring of materials exported from the pine forest to an adjacent tidal marsh and the possible ecological importance of this transfer of matter and energy. The study is designed within the framework of an ecosystem model with the intent of studying ecosystem components and functions using generally accepted investigative techiques.

AREA DESCRIPTION

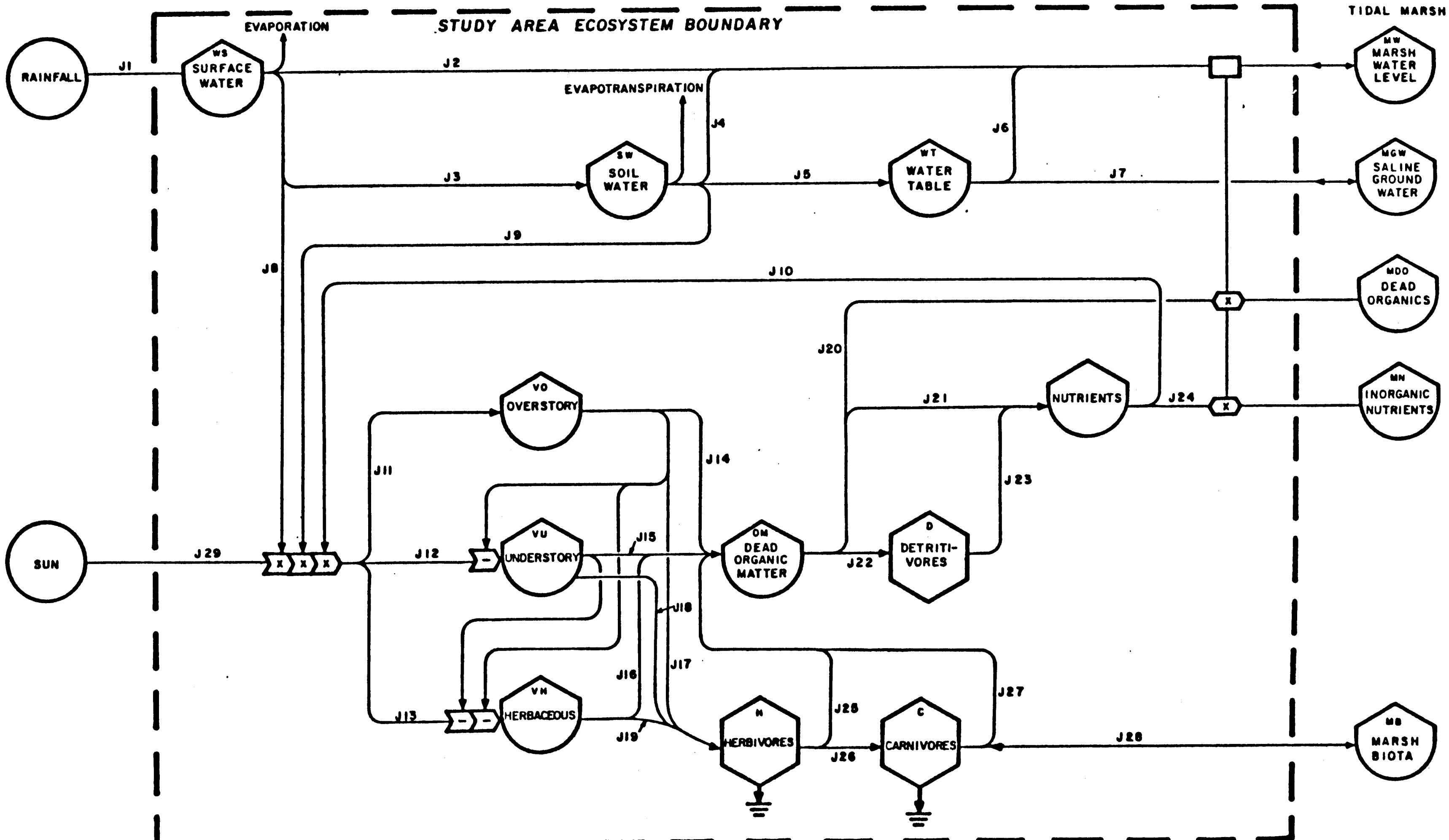
The 81-hectare study area is characterized as a slash pine (*Pinus elliottii*) forest containing a mosaic of numerous intermittent, shallow surface depressions which often remain waterlogged long after precipitation events and commonly support an abundance of hydrophytic plants. These surface depressions form larger, poorly defined swales which provide a means of drainage to the adjacent tidal marsh. The tidal marsh borders 914 meters of the pine forest within the study area and is approximately 1,200 hectares in size (the total length of marsh forest border is approximately 4,000 meters). Black rush (*Juncus roemerianus*) comprises almost 100 percent of the vegetative cover in the tidal marsh although a narow ecotone of salt meadow cordgrass (*Spartina patens*) forms a band 10 to 20 meters in width between the pine forest and the tidal marsh.

MATERIALS AND METHODS

Ecosystem Model

Investigation and assessment of ecological relationships in the study area will be performed within the framework of a conceptual ecosystem model. This model will portray those processes considered most significant to processes occurring with the pine forest and to interactions between the pine forest and the total marsh.

A preliminary, simplified ecosystem model is shown in Figure 1, using symbols developed by Odum and Odum (1976). Three types of symbols are used to represent compartments, or storages, of energy and materials: one for plants, another for animals, and a third for non-living materials. Lines connecting these compartments signify the flow of energy and materials. Boxes with an "x" represent a pumping or positive interaction; boxes with a "-" represent a negative interaction. Inspection of the diagram in Figure 1 shows a simple, but logical movement of material and energy within and through the system.



NOTE: THIS IS A CONCEPTUAL DIAGRAM, NOT A MATHEMATICAL MODEL. UNITS OF FLOW AND STORAGE CORRESPOND TO THE MATERIALS IDENTIFIED – i.e., WATER, SOLAR RADIATION, ORGANIC MATTER, NUTRIENTS OR BIOTA.

Figure 1. Ecological Systems Diagram of Study Area.

By conceptualizing the important components within the ecosystem, study methodologies, or tools, can be selected which will be used to investigate the various functional processes.

Water Resources

The study objectives of quantifying material export and identifying ecological importance of matter and energy transfer require both a quantitative and qualitative understanding of the hydrologic cycle and its components. There is little quantitative information concerning the ecological value of the physical components of the hydrologic cycle. Therefore, the present study will define nutrient transport vectors of surface and ground regimes and relate these vectors to specific habitats.

The specific hydrologic functions to be investigated are:

1) groundwater gradients relative to tidal elevations and rainfall;
2) groundwater depths and fluctuations relative to the various vegetative communities;
3) potential for saline water intrusion;
4) relationship between rainfall and the material transport vectors (surface runoff, interflow, and groundwater recharge); and
5) the range of surface, base and ground water flow volumes and rates, and the direction of these flows.

The hydrologic functions will be quantified through assessment of the hydrologic budget. Figure 2 is a conceptual model of the hydrologic cycle. The principle components to be investigated are:

1) Precipitation
2) Surface runoff
3) Interflow

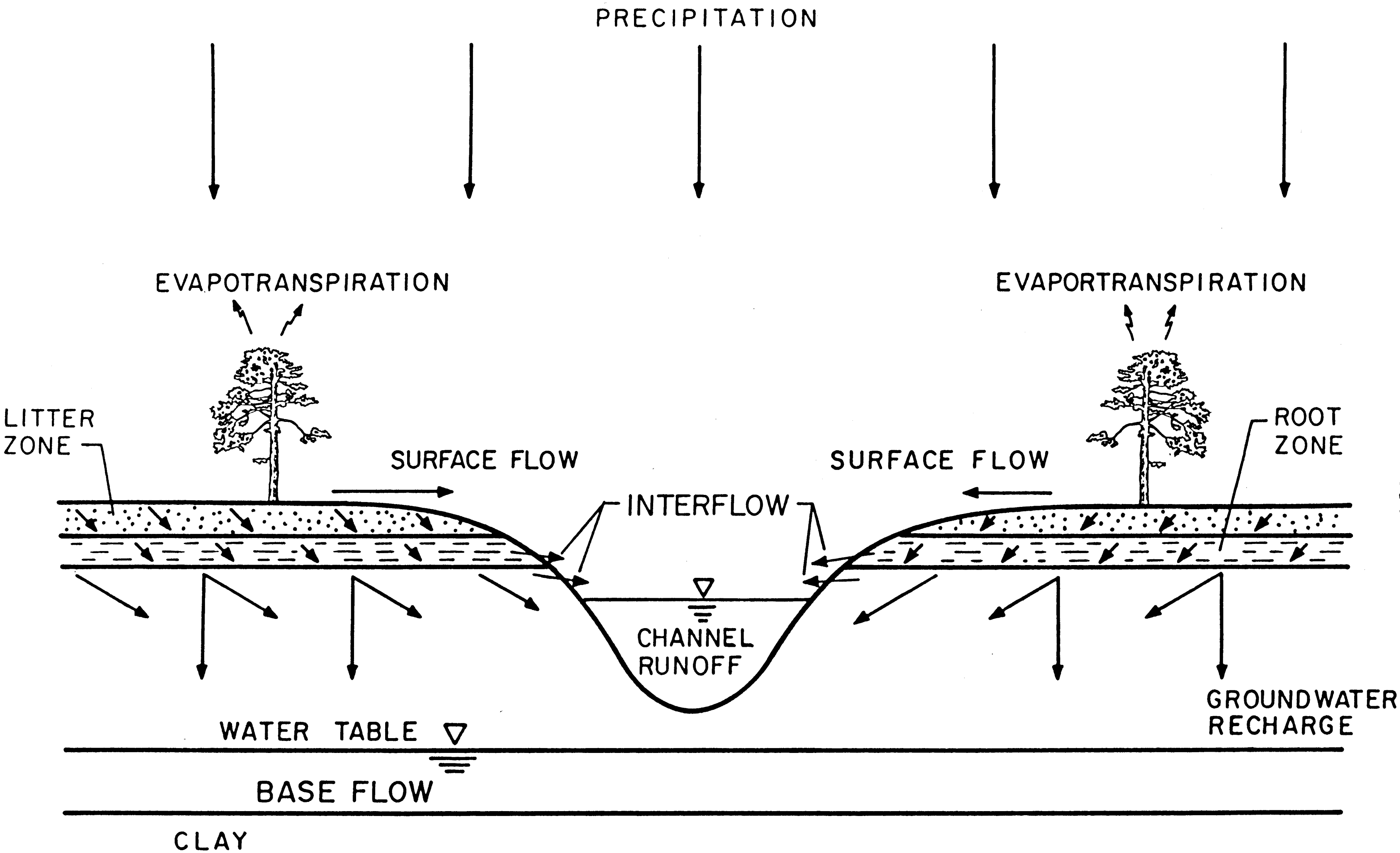


Figure 2. Hydrologic Cycle.

4) Base flow

5) Groundwater recharge

6) Evapotranspiration

A distinction has been made between interflow and base flow. As applied in this study, the term interflow refers to any non-vertical subsurface flow in the absence of boundary conditions. Baseflow refers to nonvertical flow associated with boundary conditions such as a water table or an impervious soil layer above the water table. Analysis of the interflow component will provide an assessment of lateral flow in unsaturated soil produced by anistropic soil conditions.

The above six components of the hydrologic cycle will be investigated through field sampling and analysis of key parameters on a sample catchment area. Extrapolation of these data will then be made to the rest of the study area.

The monitoring system as shown in Figure 3 consists of five groundwater wells, a runoff monitoring flume, a rain gage, and a tide gage. Monitoring equipment will measure rainfall, surface runoff into the tidal marsh, groundwater elevations, and tidal elevations.

Water quality analysis for all surface and groundwater samples will be by standard methods for total organic carbon, dissolved organic carbon, total suspended solids, dissolved solids, nitrate-nitrite nitrogen, total Kjeldahl nitrogen, ammonia introgen, total phosphate, and orthophosphate. In addition, pH, temperature, DO, salinity, and specific conductance will be obtained during each sample collection.

Rainfall, tidal elevation, and surface runoff will be monitored on a continuous basis. Ground water elevations will be monitored manually during both discrete rainfall events and dry periods at four

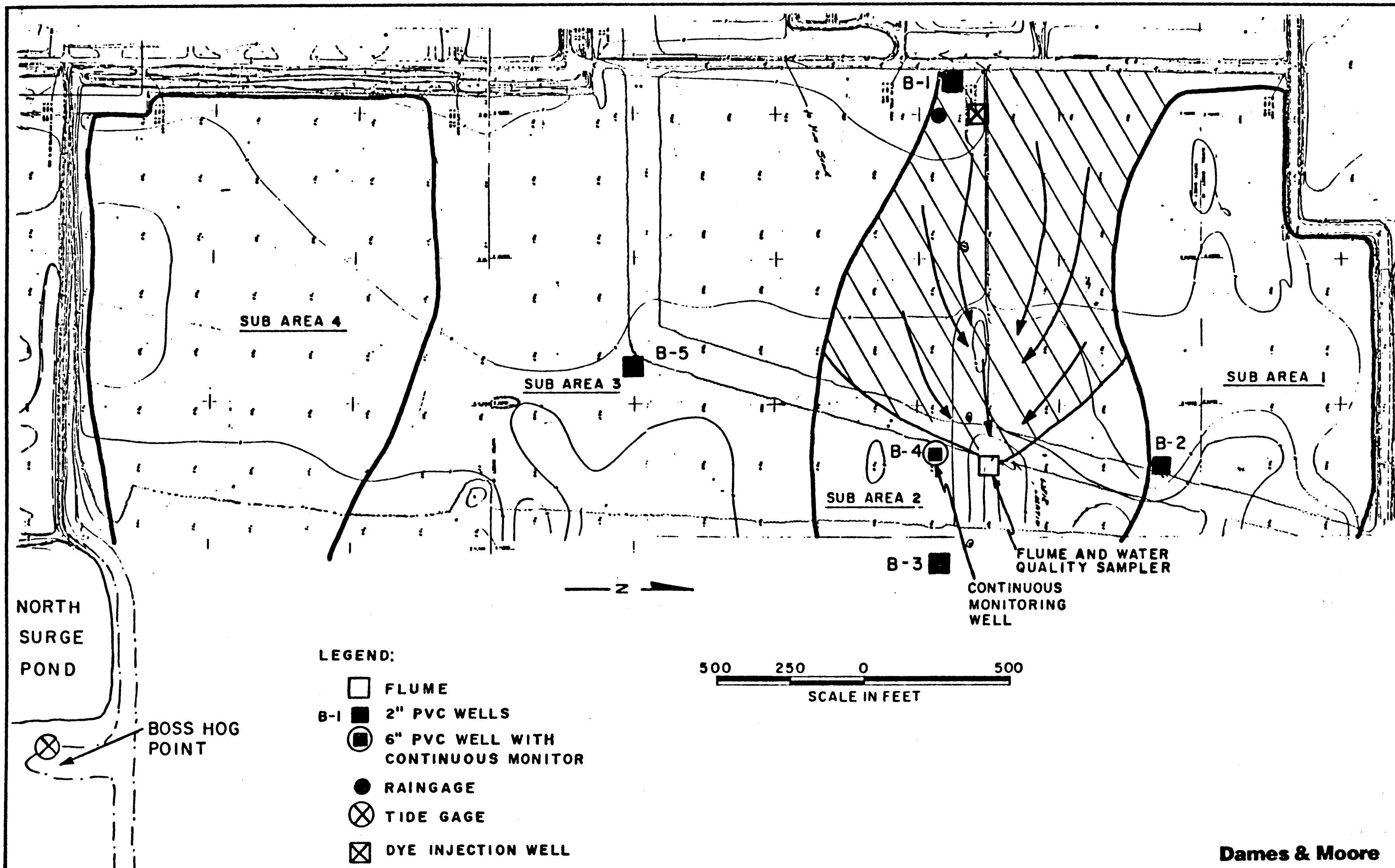


Figure 3. Water Resources Sampling Stations.

wells and continuously at one well.

Permeability tests and infiltration tests will be performed in the field. This information will be supplemented by grain size distribution analyses and density tests on representative soil samples taken in at least four locations in the Study Area, from the surface down to a depth of 3.05 meters.

To obtain an estimate of that portion of the flow which enters the drainageways through the soil (interflow) and the relationship between basin size and runoff, a dye study coupled with daily soil moisture measurements will be performed in the drainage basin upstream of the flume. The dye study will consist of injection of dye into a shallow well (61 cm deep) near Well B-1 and monitoring the dye concentration in the east/west ditch near both Well B-1 and the flume. The purpose of the dye study is to develop an estimate of flow quantities in the upper reaches of the ditch where all inflow is from interflow and thus to determine the relative contribution of surface sheet flow and interflow to runoff reaching the marsh.

Hydrologic field and laboratory data will be analyzed in conjunction with the biological findings to quantify the range of seasonal transport of nutrients to the tidal marsh and to assess the relationships between vegetation types and the surface and groundwater regimes.

The hydrologic information will provide the biological study team with a data base for quantifying the nutrient transport into the tidal marsh estuary from the study area. By combining the surface, base, and groundwater flow magnitudes with the measured organic and inorganic nutrient concentrations, a series of values will be derived for seasonal organic and inorganic nutrient flux into the tidal marsh per unit area and per unit rainfall. In addition, the hydrological information concerning groundwater recharge and fluctuations and surface water flow patterns will provide information useful in defining the relationships between vegetation and the ground and surface water regimes.

Biological Resources

Vegetation Mapping. Using topographic data and color infra-red photography (scales 1:15,000 and 1:6,000) complemented by ground-truthing, a vegetation map will be prepared delineating the major vegetation associations of the study area and of the tidal marsh basin. The size of these associations can be measured by polar planimetry and a preliminary estimate of their productivity can be made from existing literature and field measurments of peak biomass. The estimate of productivity for the whole marsh ecosystem will allow a preliminary estimate to be made of the comparative productivity value of the adjacent pine forest to the marsh.

Vegetative Species Composition Survey. Sampling for species composition will be conducted in late spring to coincide with the reproductive stages of important plant species. Seven transects will be sampled in the study area, comprising up to 30 quadrats each. Transect lines will be perpendicular to drainage patterns in the pine forest (parallel to the tidal marsh) to assure sampling within all habitat types in the Study Area.

Quadrat sampling of the understory and herbaceous layers will be conducted using methods simlar to those described by Daubenmire (1968). Concomitant sampling of the overstory will be conducted using the plotless, point centered quarter (PCQ) method of Cottam and Curtis (1956). Calculations from this sampling effort will include density, frequency, basal area or dominance, relative frequency, relative dominance, and relative density (Mueller-Dombois and Ellenberg, 1974; Phillips, 1959; Cox, 1976). An Importance Value, derived by summing the relative frequency, relative density, and relative dominance (Curtis, 1959), will be used to rate overall dominance by each species for each vegetative layer.

Peak Biomass. Although peak live biomass underestimates true net primary production (Gosselink, et al., 1977), the method requires only one field study, is relatively easy to make, and provides quantitative information from which comparison with other studies of similar habitats can be easily made. Peak live biomass will be estimated from the harvest of 50 to 60, randomly selected 0.25 m^2 quadrats during a time when biomass is expected to be at a maximum. Standing dead litter and root material will also be collected from the same quadrats.

Estimates of peak litter will be obtained based on one autumn collection when leaf and needle litter fall is expected to be at a maximum. Litter fall will be collected in 20, one square meter, elevated screen baskets positioned randomly in the study area. At the end of a two month interval the litter will be removed from the baskets, ovendried, and weighed.

Additional sampling will be conducted in the tidal marsh and in portions of the study area that are not part of the tidal marsh drainage to provide comparative data. By sampling the most mobile components for potential materials exported, an estimate of the area's capacity for detrital and nutrient export can be developed. The values will be compared with both literature values and the values obtained from the field monitoring studies.

Wildlife Inventory. This phase of the project will determine the wildlife usage of the study area in comparison with the nearby salt marsh. The wildlife populations which are dependent upon the study area will be inventoried over a one year period. A variety of census techniques will be employed to provide quantitative field estimates of the use of these habitats by mammals and birds. A series of snap-trapping grids and scent posts will be employed to estimate use of the area by mammals. Since Gulf coast areas experience highly fluctuating bird numbers during migrations, birds will be censused visually by a series of transects during one season and augmented by a roadside census every other week. These data will document the use of the site as a temporary refuge for migrating birds. Surveys of reptiles and amphibians will be based on collections obtained using 30-meter drift fences during the spring.

Soil Surveys. Soil samples will be collected with soil corers to a depth of 20cm concomitantly with the sampling for root material for biomass determination. Collections will be made quarterly along the linear transects used for the vegetative species composition survey. Samples will be analysed for soil oxygen, soil moisture, pH, total organic carbon, total nitrogen, total inorganic nitrogen, ammonia nitrogen, hydrogen sulfide, orthophosphate, and total phosphate. In addition, soil color will be recorded in the field during each collection, using Munsell soil color charts, and soil oxygen will be determined using the method of Patrick (1977).

These determinations will serve to characterize the physical and chemical conditions and degree of aerobic/anaerobic decomposition in the soil. These data will aid in describing wetland functions attributed to the study area.

CONCLUSIONS

Characterization of the ecological significance of the study area may be made in several ways. The most direct method will be based on quantifying key compartments or pathways in Figure 1. For example, what is the annual flow of freshwater (J_2), organic matter (J_{20}) and nutrients (J_{24}) to the tidal marsh? Direct evaluation of _in situ_ value may be made by comparison with existing literature data. Another method of evaluation will be to compare the export of the study area to the estimated total storage capacity of the entire drainage basin. This estimate will be based on selected field measurements and literature data. Thus, the total quantity of dead organic matter in the marsh and the total annual imput (both in terms of marsh productivity and export from all adjacent lands) will be compared to contributions measured as export from the study area.

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BIOTIC RESPONSE TO DRAWDOWN AND REFLOODING IN A CLAY SETTLING POND

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ABSTRACT

About one-third of the total weight of the Central Florida land-pebble phosphate matrix is composed of phosphatic clays ("slimes"), a phosphate beneficiation waste product deposited in dammed retention ponds for consolidation. Clay settling ponds often are characterized as an undesirable element in the mining landscape. However, the wetland and aquatic values accruing to these man-made systems are significant and remain largely unrecognized and poorly researched. Settling ponds are functional wetlands and aquatic ecosystems which offset the loss of natural wetlands during mining and which can increase the regional inventory of such communities after mining. A settling pond in Polk County, Florida experienced a rapid vegetative response to dewatering (drawdown) for stage settling initiated in November 1978. The system was reflooded 11 months later. This paper includes a summary of indigenous biotic resources encountered during the 10 month study interval and details the successional trends and physical changes within the system. Settling pond successional stages beyond terminal consolidation seldom occur, because these areas typically become subject to reclamation or to remining in those cases where settling ponds were constructed on unmined ground. Reclamation options for settling ponds include agricultural, biomass, and natural systems. The worth of settling ponds as potential long-term wetland resources should be reconsidered by the industry, governmental regulators and the environmental leadership.

INTRODUCTION

EcoImpact, Inc., a Florida based ecological consulting firm, was retained by International Minerals and Chemical Corporation (IMC) to perform environmental field studies of a phosphatic clay settling pond designated as N-12A. Field personnel conducted work tasks on and adjacent to the project site during two study phases from July 1979 - May 1980. Phase I of the N-12A study, conducted July - August 1979, coincided with a stage settling drawdown (clay consolidation by dewatering). The purpose of the Phase II research was to document the system's ecological changes in response to subsequent reflooding. The major portion of these studies was conducted from January - May 1980.

Phosphatic clays generally comprise about one-third of the total weight of the Central Florida land-pebble phosphate matrix. Typically, they are discharged from processing plants to settling areas as 2-5 percent solids by weight. Economical methods of rapidly consolidating clay to greater than 25 percent solids remain an important facet of on-going reclamation research. This research project arose from the necessity for a greater understanding of the ecological and wetland values of clay settling ponds while active, and their resource management potential when reclaimed.

AREA DESCRIPTION

The N-12A settling pond is part of IMC's Noralyn Mine Complex and receives phosphatic clay residues ("slimes") from the Clear Springs Mine to the east via pipelines. It lies about 1.2 km due south of the Bartow city limits (Figure 1) in Polk County, Florida (Sections 19 & 20; T30S; R25E).

The total N-12A surface area is approximately 95 ha (230 acres),

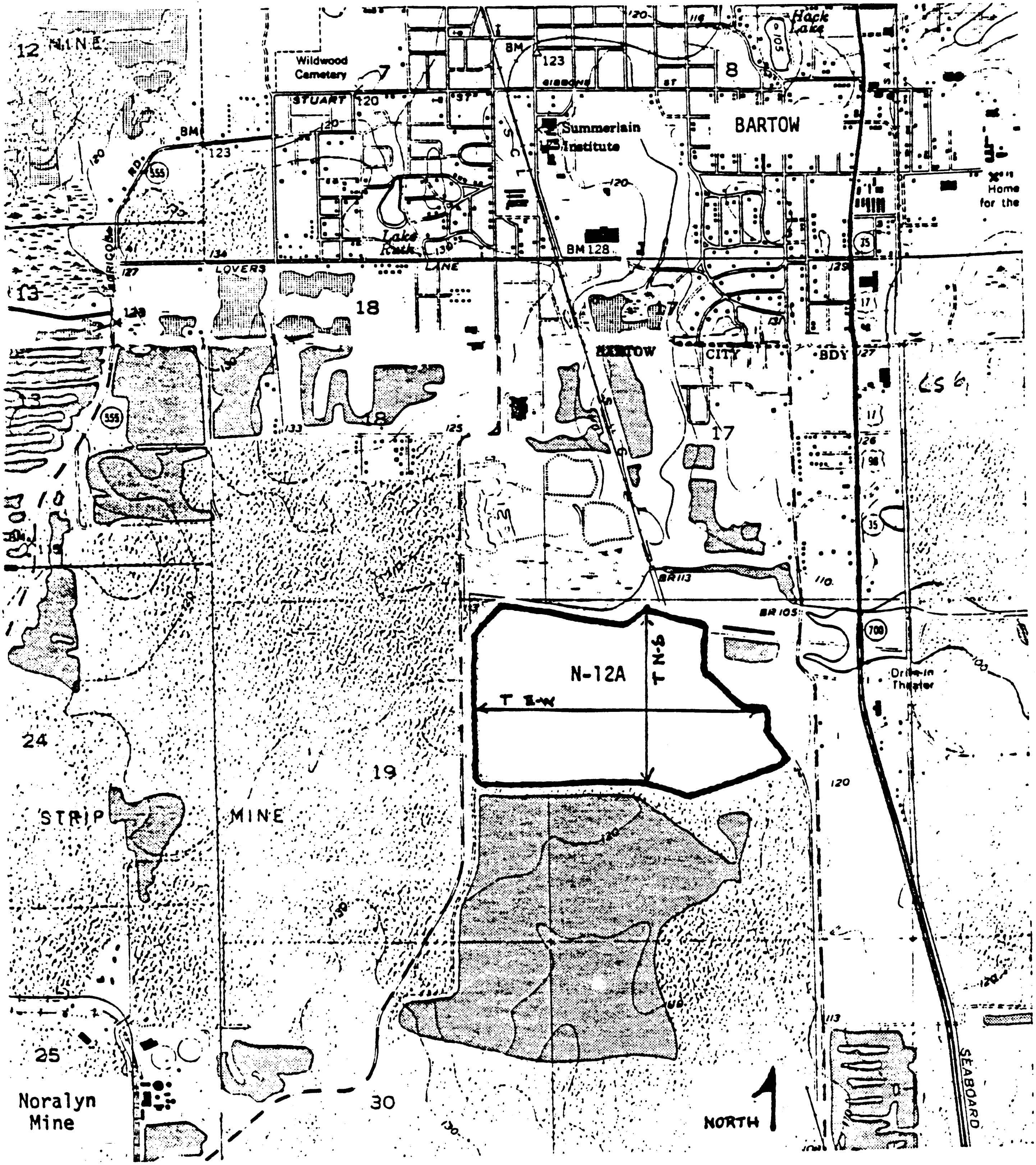


Figure 1. Location Map: N-12A Settling Pond

while the area of the retention dam is about 16 ha (40 acres). The south dam was formed in conjunction with the pre-existing north dam of the adjacent pond N-12, which was completed in 1973. The north, east, and west dams of N-12A were completed in June 1975. The dam was constructed around existing mine pits, resulting in a greater storage capacity than a strictly above-ground pond. The dam rises 12.2 m (40 feet) above an original land surface averaging 35 m (115 feet) mean sea level (msl). The average mining pit depth was 7.6 m (25 feet), creating an average pond depth of about 20 m (65 feet). The dam was constructed of overburden soils with an internal sand tailings drain.

N-12A has been an active clay retention area, sometimes referred to as a "slime pond," since October 1975. Clays are pumped into settling areas by pipeline, and clear water can be discharged from controlled weir spillways with square-frame risers. Settling ponds typically remain active for about 6-10 years, with the first 2-3 years characterized by heavy clay deposition. Thereafter, introduction is more moderate and generally follows periods of consolidation and subsidence.

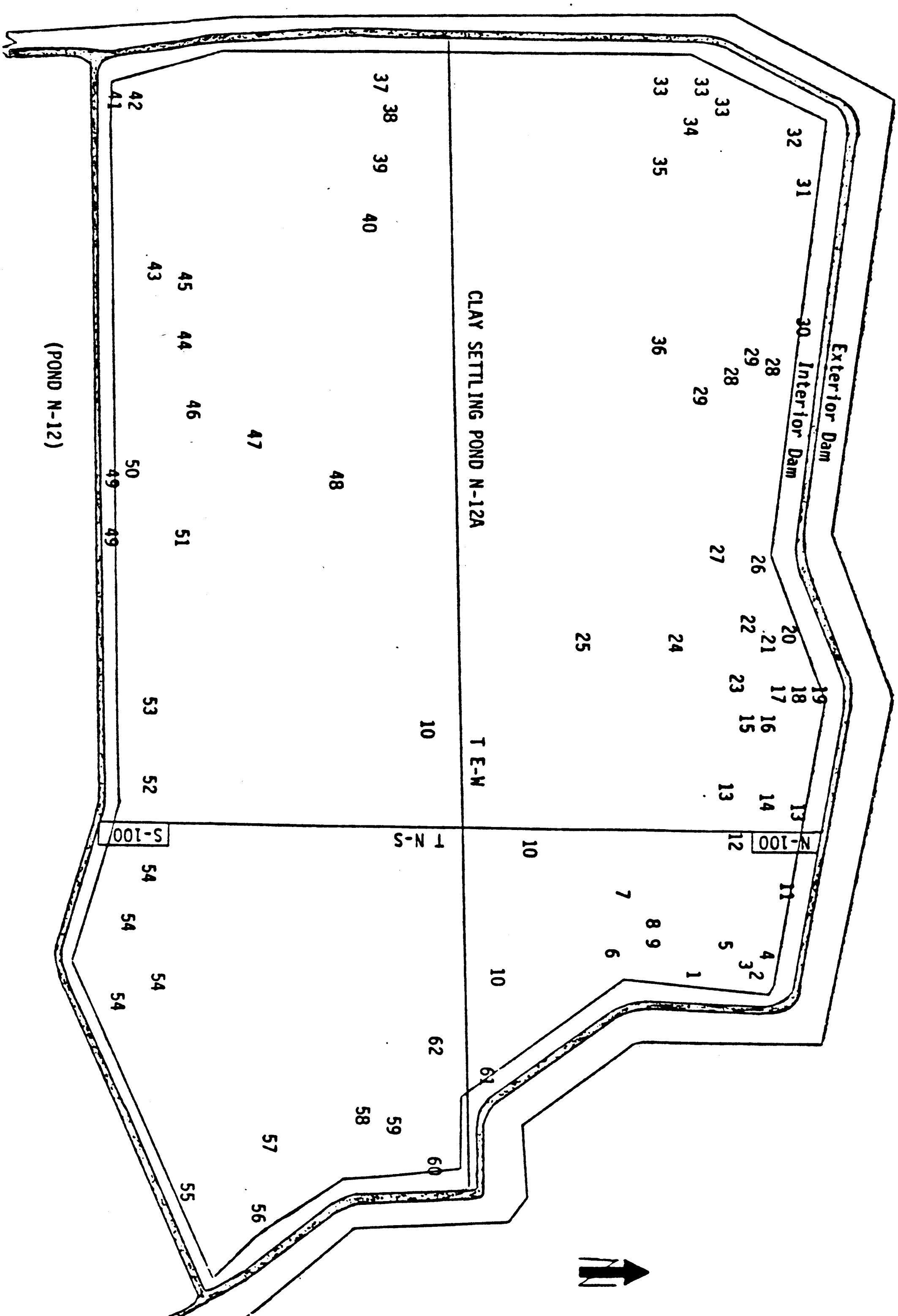
MATERIALS AND METHODS

Phase I - Drawdown Studies

A rapid vegetative response was noted within N-12A to dewatering for stage settling that was initiated in November 1978. This presented an opportunity to study the drawdown from a wetland management perspective and to identify the plant and wildlife response to dewatering. Such important aspects of settling pond natural history generally have been overlooked by all but sportsmen and local naturalists in the past. The findings resulting from this study are an initial treatment of

Figure 2.

N-12A: Observation Stations and Transect Configurations



these considerations.

The specific survey objectives of the drawdown studies were:

1. Describe the plant species composition of the pond.
2. Determine the frequency of occurrence by species.
3. Assess the plant distribution and zonation characteristics by dominant species and/or associational groups.
4. Identify the wildlife species utilizing the settling areas during the field studies.

Sampling was restricted, because the substrate of the settling pond was to unstable to support a person. Two, 30 meter (100 feet) transects were established beginning at the interior crest of both the north and south dams and running out approximately 21-24 meters (70-80 feet) onto the settling pond floor, which was the maximum, safe distance. Plant species occurrence data were gathered at 30 cm (1 foot) intervals along these transects, designated as N-100 and S-100. As a supplement to the shortened transects, 62 observation stations were established around the dam perimeter from which the species composition and relative density of plant associations within the system were described (Figure 2).

In November 1978, after the dewatering of N-12A, field notations of vegetative establishment were recorded at regular intervals by IMC personnel. Within one month the plant assemblage changed from "sparse" to "about 90 percent cover."

Reflooding Studies

The chronology of the biological impacts and responses of reflooding after the 10-month drawdown were investigated at regular intervals from November 1979 to May 1980. The specific survey objectives were similar to those listed for Phase I. Preliminary reactivation surveys conducted during October - December 1979, suggested that water depths, vegetative characteristics and wildlife utilization of N-12A varied from north to south and east to west due to the nature of clay deposition and resultant pond bottom contours.

Phase II quantitative and qualitative studies were conducted on foot and by vehicle, canoe, and motor boat. Specific sampling stations were established at regular intervals along a north-south transect 780 m (2,560 feet) from dam crest to dam crest, and an east-west transect 1,292 m (4,240 feet) from water's edge to water's edge. Stations were installed at 30 m intervals along the north-south line and at 45 m (150 feet) intervals along the east-west line, yielding 25 and 29 stations, respectively. All stations were marked by anchored floats, stakes, or flagging on emergent, woody vegetation.

Data and samples gathered at the monitoring stations included: water depth to the drawdown clay surface; depth of newly introduced clays, where applicable; a bottom detritus grab sample; a benthic dredge sample; a description of emergent plant growth within a 6 m (20 feet) radius of the sampling site; and observations concerning wildlife, fish, and invertebrate populations, as well as water conditions in the immediate vicinity. The benthic samples subsequently were analyzed for biotic occurrence.

Line-intercept plant transects, replicating the location of similar transects in July 1979 (N-100 and S-100), were run for both the north and south dams. Total transect lengths from the interior dam crest to the existing water's edge were 8.5 m (28.1 feet) for the north and 11.0 m (36.4 feet) for the south. A continuous linear measurement of cover for each species intercepted by the line was recorded, in addition to data gathered at 30 cm intervals.

Water samples collected by IMC in the receiving waters of N-12A outfall complied with existing Florida Department of Environmental Regulation 17-3 water quality standards and the "best available technology" standards of the NPDES program. Generally, all parameter values (SS, Turb., Spec. Cond., D.S., P-S, P-T, Ca, Cl and Fe) were stable throughout the September 1976 - October 1977 sampling interval.

RESULTS AND DISCUSSION

Drawdown Studies

A total of 41 vascular plant species were identified on the interior dams and settling pond floor (Table 1). Six of these plants were early succession hydric or mesic woody species, with the remainder categorized as herbs, sedges, and grasses. Nine plants of the total are faculative hydro-phytes, while at least three are essentially obligate hydrophytes, including bulrush, cattail, and a species of galingale (*Cyperus odoratus*).

Quantitative transect data obtained were sufficient to make an inference about the frequency of occurrence of the most common species within the first 30 meters from the dam crests (Table 2). Thirty-one species were encountered along the two line-intercept transects, with N-100 yielding 18 species and S-100, 23 species. Plants of only ten species grew along both the North and South transects.

The most common species, in descending order, found along transect N-100 were: an unidentified sterile herb; *Typha domingesis* (cattail); *Pluchea odorata* (fleabane); *Conyza parva* (horseweed); *Cynodon dactylon* (Bermuda grass); *Paspalum notatum* (Bahia grass); *Cyerus odoratus* (galingale); and *Urena lobata* (Ceasar weed). For transects S-100, the follow-

Table 1.
Plant Species Observed Within Settling Area N-12A
July - August, 1979

SCIENTIFIC BINOMIAL	COMMON NAME
Aeschynomence americana	Shy-leaves
Amaranthus australis	Southern amaranth
Ambrosia artemisiifolia	Common ragweed
Andropogon glomeratus	Bushy beardgrass
Baccharis glomeruliflora	Saltbush
Bidens pilosa	Beggar ticks
Cassia faciculata	Partridge pea
Chenopodium album	Mexican tea
Colacasia esculentus	Wild taro
Conyza parva	Horseweed
Cynodon dactylon	Bermuda grass
Cyperus ordoratus	Galindale
Digitaria sanguinalis	Crabgrass
Echinochloa walteri	Barnyardgrass
Eclipta alba	Yerba-de-tago
Erechtites hieracifolia	Fireweed
Eupatorium compositifolium	Dogfennel
Heterotheca subaxillaris	Camphor plant
Indigofera hirsuta	Indigo plant
Ipomoea spp.	Morning glories
Lantana camara	Shrub verbena
Lepidium virginicum	Pepper grass
Ludwigia erecta	Ludwigia
Ludwigia octovalvis	Ludwigia
Mikania scandens	Climbing hempvine
Morrenia adorata	Morrenia vine
Panicum purpurascens	Papa grass
Panicum virgatum	Switch grass
Paspalum notatum	Bahia grass
Phytolacca rigida	Pokeweed
Pluchea odorata	Marsh-fleabane
Rhynchelytrum roseum	Natal grass
Ruellia caroliniensis	Wild petunia
Salix caroliniana	Coastal-plain willow
Schinus terebinthifolius	Brazilian pepper
Scirpus validus	Bulrush
Scoparia dulcis	Sweet-broom
Setaria geniculata	Foxtail grass
Typha domingensis	Narrowleaf cattail
Urena lobata	Caesar weed
Sterile Herbs**	

** Two fairly common dam bank plants were not in flower or in seed (sterile herb), and could not be identified in either the Botany or Range Plants Herbariums at the University of Florida.

Table 2.
Plant Species Frequency of Occurrence
Transects N-100 and S-100 (N-12A)
July - August, 1979

	PRECENT FREQUENCY	
	N-100	S-100
Aeschynomene americana	--	0.99
Ambrosia artemisiifolia	--	6.93
Andropogon glomeratus	0.99	--
Baccharis glomeruliflora	--	1.98
Bidens pilosa	--	--
Cassia fasciculata	--	0.99
Chenopodium album	--	0.99
Colocasia esculentus	--	0.99
Conyza parva	9.90	8.91
Cynodon dactylon	9.90	--
Cyperus odoratus	8.91	1.98
Digitaria sanguinalis	--	0.99
Eupatorium compositifolium	4.95	9.90
Heterotheca subaxillaris	--	6.93
Indigofear hirsuta	0.99	--
Ipomea spp.	--	0.99
Lantana camara	--	7.92
Lepidium virginicum	2.97	--
Ludwigia erecta/octovalvis	1.98	31.68
Mikania scandens	3.96	--
Panicum purpurascens	--	1.98
Panicum virgatum	--	1.98
Paspalum notatum	9.90	--
Pluchea odorata	15.84	11.88
Rhynchelytrum roseum	3.96	3.96
Ruellia caroliniensis	0.99	--
Salix carolinina	1.98	7.92
Scoparia dulcis	--	3.96
Typha domingensis	18.81	29.70
Urena lobata	7.92	0.99
Unidentified sterile herb	29.70	0.99

ing species were noted as common: Ludwigia peruviana (primrose willow); Typhas domingensis; Pluchea odorata; Eupatorium compositifolium (dog-fennel); Conyza parva; Lantana camara (shrub verbena); Salix caroliniana (southern willow); Ambrosia artemisiifolia (ragweed); and Heterotheca subaxillaris (camphor weed).

Descriptions from the 62 observation stations revealed that the rivulets, which flow westward across the site due to the internal drainage pattern, were dominated by the woody plants, willow and ludwigia, while most of the central interior was dominated by varying-aged cattail stands. A more heterogeneous association of herbs and sedges occured within 100 meters of the dam, but could not be quantitatively assessed due to substrate instability.

Based on our field studies and aerial photography review, we estimated the percent of settling pond bottom occupied by each of the five major cover types after the 10-month drawdown. These estimates represent the areal vegetative cover, excluding the dams, prior to reflooding:

Table 3.

% Cover	Cover Type	Acres
60-70	Cattail	138-161
15-25	Willow	36-58
5-10	Transtitional herbs, grasses	12-23
5-7	Ludwigia	12-16
3-5	Open (water and exposed clay)	7-12

This N-12A vegetative complex is interpreted as consisting of an integrated, fresh-water marsh/swamp community.

A total of 23 bird species, 5 mammal species, and 3 reptiles were observed utilizing the study site during the mid-July through early August drawdown period. The most commonly noted bird species included Red-winged Blackbird, Coot, Common Gallinule, Least Sandpiper, Florida Yellowthroat, Great Egret, Snowy Egret, Louisiana Heron, Green Heron, Little Blue Heron and Great Blue Heron. Black-necked Stilts, Pied-billed Grebes and Glossy Ibis also were observed feeding with the settling pond.

Bobcats, raccoons, and opossums were sighted along the dams. The abundance of prey, based on sign and sightings of animals such as marsh rabbits, rice rats, and invertebrates, along with the shelter and escape cover provided by the dense, dam-site vegetation, may substantially meet the habitat needs of these omnivorous predators. This is especially true if remnant natural system shelter and breeding habitat coexist nearby, such as the Sixmile Creek system adjacent to N-12A on the north.

Reflooding Studies

The physical and biological changes in the N-12A ecosystem as a result of reflooding and new clay deposition were most intensively investigated during the January, March and May study intervals. Water level fluctuations during the period varied about 0.75 m in response to mining operation water usage, while average water depths ranged from 1.5 - 2.5 m overall. Within six months, all plant species except willow were eliminated from the settling pond basin when flooded to a depth of 1.5 m or more. Willow association canopies formed island-like thickets where surviving individuals had greater than 0.5 m of emergent growth. Changes in the pond bottom configuration due to differential settling were common, with undulations and depressions occurring even in areas unaffected directly by new clay deposition due to loading pressure.

Variable decomposition of the herbaceous bottom cover established during the drawdown apparently was indicative of the relative fiber content of plant species tissues. Only detritus from woody species and some cattail was encountered in the bottom grab samples by May. There was an overall subsidence of the settling pond floor of as much as 0.5 m by the end of the study.

Active settling ponds attract nekton, plankton, and sessile organisms which attach to objects such as plant stems and water control structures in the water column. When inactive, and as the clays consolidate and the bottom stabilizes, niches become available to other plants and animals that apparently cannot survive in an active settling pond environment.

The diversity and abundance of settling pond bethnic (bottom-dwelling) organisms is undocumented in existing literature. Past examination of settling pond sediments has identified an abundance of blind mosquito larvae in the upper strata of clay substrates. Our N-12A collection of bottom samples taken six and nine months after reflooding revealed the presence of a low abundance of midge larva from seven common species of chironomids. All identified species often are associated with pioneer aquatic communities.

All collected individuals had extra gills or gill-like structures which enable them to tolerate very low oxygen tensions. This phenomenon may have been a response to an unusually high biological oxygen demand generated by the abundance of plant material decomposing on the bottom after reflooding. The nature of the substrate and shifting bottom topography seemingly is unsuitable for development of a mature and diverse benthic population, at least until active clay deposition ends.

During the reflooding studies, 70 bird species, 8 mammals, and 10 reptiles and amphibians were identified on the project site on the basis of species-specific signs and/or sightings. Overall, the most regularly abundant bird species, excluding migratory ducks, were: Coot; Snowy Egrets; Red-winged Blackbirds; Double-crested Cormorants; Great Egrets; Ring-billed Gulls; Green Herons; Common Crows; Little Blue Herons; Boat-tailed Grackles; and Pied-billed Grebes. The most unusual birds encountered at N-12A were the Black Tern, Smooth-billed Ani, Rusty Blackbird, Black-necked Stilt (nesting), Black Skimmer, White Pelican, and Limpkin. Of the duck species, the most commonly sighted were: Ruddy Ducks; Blue-winged Teal; Ring-necked Ducks; and Shovelers. A brood of Florida Ducks was successfully reared on the adjacent pond N-12. Otters were the most commonly observed larger mammal, the marsh rabbits and cotton rats also seen regularly. Signs from raccoons and bobcats were common, and a bobcat with two young was seen in the shrub cover of the interior south dam. None of the reptiles and amphibians was commonly encountered except cooter turtles, although many soft-shelled turtles, alligators, pickerel frogs, water snakes, and green anoles were observed.

<u>Vegetative Transect Data</u>. A total of only 19 plant species were encountered on the north and south interior dams, with 12 and 14 species per north and south transects. The south dam, older by two years than the north, had substantially more mid-slope and up-slope woody cover. The north dam was dominated by graminoid species. Plant cover also was more dense on the older south dam, with only about 5 percent of the area occupied by bare soil and detrital matter, compared to 25 percent for the north. The roughly 2.5 m increase in the water level since the July 1979 sampling eliminated the biologically active toe of the dam

transition zone, as well as the herbaceous pond bottom species. Thus, the vascular plant species composition of N-12A was reduced about one-half by reflooding.

CONCLUSIONS

The ecological monitoring study of the drawdown and reflooding of settling pond N-12A, which is summarized in this paper, suggest the following:

1. Clay settling ponds can function as important habitat for aquatic, wetland and terrestrial wildlife populations. The study site was biologically active and apparently offers new and replacement niches for many floral and faunal species during and after mining. Typically, settling areas are reclaimed to land uses other than wetlands.
2. Within a limnological framework, settling ponds should be considered man-made impoundments with many of the attributes of natural lacustrine ecosystems and their adjacent wetlands. This perspective generally is not recoginzed by the public at large.
3. The industry's active settling ponds are similar in their ecology and wetland management potential to man-made waterfowl production areas. Periodic drawdowns seemingly can enhance the wetland wildlife values of settling ponds.
4. The worth of settling ponds as potential long-term wetland resources should be reconsidered by the industry, governmental regulators, and the environmental leadership.
5. Settling ponds can be considered as existing wetland and aquatic habitat which offset the loss of natural wetlands during mining, and which can increase the regional inventory of these systems after

mining.

6. Our research findings indicate that plant and animal utilization is high and that the rich assemblage of birds and other vertebrate animals utilizing these ponds is diverse and regionally important.
7. Aquatic and wetland plants with wind-dispersed seeds, such as cat-tail and willow, tend to dominate the interior settling pond plant assemblage.
8. A variety of highly desirable wetland plant species conceivably can be introduced by men to improve plant diversity in settling ponds.
9. Phase II studies show that N-12A willows have survived and exhibited normal phenological behavior for more than eight months in water depths varying between 5 - 8 feet.
10. The water, from either active or reclaimed settling areas, is not polluted by state standards.
11. Reclamation of settling ponds should embrace a multi-use land policy, including a mix of cultural, agricultural and natural systems. More innovative approaches to their management should be researched.

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CONSIDERATIONS FOR DESIGN OF AN ARTIFICAL MARSH FOR USE IN STORMWATER RENOVATION

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ABSTRACT

An artificial marsh and a filtration impoundment have been designed to function in series to renovate urban stormwater entering Megginnis Arm of Lake Jackson. The stormwater is sediment-laden with excess nutrients, heavy metals, and hydrocarbons absorbed to the clay fraction. The sediments pose the greatest problem, but dissolved pollutants contribute as well. The function of the artifical marsh is to trap the clay fraction and provide uptake for the dissolved pollutants. Discussion will encompass criteria for vegetative component selection; particularly growing cycle characteristics, nutrient uptake capacities, periphyton host suitability, hydroperiod constraints, morphology as it relates to sediment trapping tendencies, and compatibility to substrate. This project has been funded by the U. S. Environmental Protection Agency's Clean Lakes Program and the State of Florida Department of Environmental Regulation.

INTRODUCTION

Lake Jackson, nationally renowned for its trophy large-mouth bass, is located in the Florida Panhandle near Tallahassee. The lake encompasses 1618 ha (4000 ac) at altitude 25.9 m (95 ft) within a 11.9 sq. km (43.2 sq. mi) drainage area. The majority of the lake's watershed is either undeveloped or in low-density residential development. The urbanized portions of the watershed are almost entirely within the drainage area of Megginnis Creek, the primary tributary to the Megginnis Arm of Lake Jackson (Figure 1). Stormwater entering Megginnis Arm is virtually uncontrolled and has resulted in eutrophication of the arm.

The effects of urban stormwater on Lake Jackson during the past 15 years is quantified in several studies (Harris, 1974; Schamel, 1974; NWFWMD, 1977; Byrne, 1980). The recent eutrophication of the arm is in sharp contrast to the general oligotrophic nature of the main body of Lake Jackson. Symptoms of this problem are extensive and persistent sediment plumes which appear in the arm after rain events, as well as considerable filamentous algae growth throughout, reflects the hypernutrification of the water column. These sediment plumes and algae mats have begun to encroach into the main basin of the lake, thereby raising concern over its impact on existing aesthetic and recreational values.

Aside from the effects on large scale introduction of urban sediments to the lake, the sediment fraction of the stormwater, particularly the fine clay particles and colloidal material, have significant quantities of phosphorus, nitrogen, heavy metals, and petroleum hydrocarbons adsorbed to them. When these particles are subjected to anaerobic conditions and many of them are subsequently resuspended, many adsorbed constituents are released to the water column, intensifying the eutrophic conditions in Megginnis Arm.

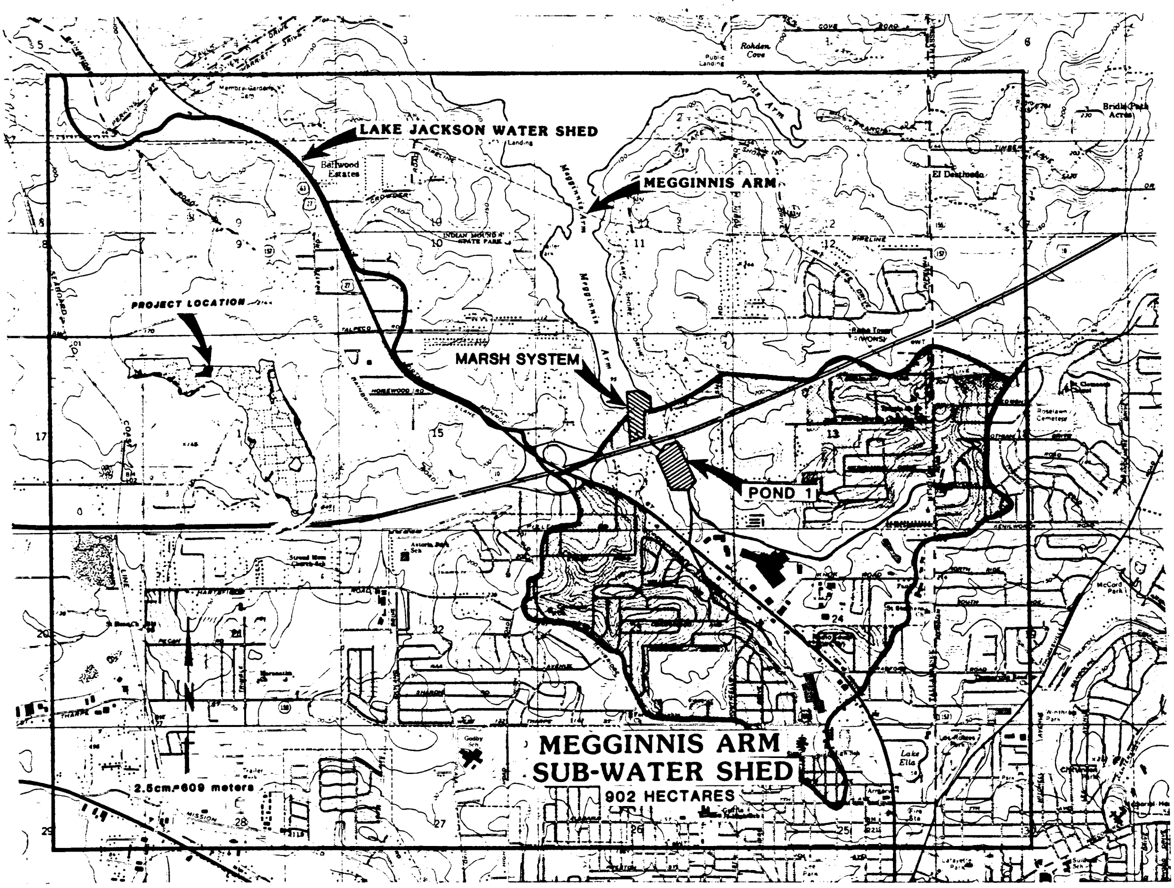

Figure 1. Location of Lake Jackson watershed.

The objectives of this project are to remove the sediment load and excessive levels of dissolved nutrients from the stormwater of the Megginnis Creek watershed. To achieve this goal, a dual system consisting of a filtration impoundment and an artificial marsh system has been designed. The filtration impoundment will serve to remove the majority of the sediment fraction from the stormwater directly upstream of the artificial marsh. The artificial marsh will serve to further remove fine clay particles, colloidal material, and dissolved constituents from the water column directly upstream of Megginnis Arm.

AREA DESCRIPTION

Megginnis Arm (Figure 1) of Lake Jackson has a surface area of 27.5 ha (68 ac) at altitude 25.9 m (85 ft). Its primary source of surface water is Megginnis Creek, which drains a watershed of 902 ha (2230 ac) with altitudes ranging to 89.1 m (279 ft). This watershed is largely urbanized, with most commercial and roadway development occurring during the past 15 years. The large increase in impervious surfaces has promoted a higher runoff coefficient, greatly increasing the introduction of sediments and other contaminants to the Arm.

The marsh site is adjacent to the creek-run and is approximately 4 ha (10 ac) in area. The southern half of the project site is currently vegetated by a sweetgum stand, with the northern portion covered by blackberry, willow, and sumac. The land currently ranges from 35.6 m (88 ft) to 40.9 m (101 ft) in elevation. The topsoil is clay sand which grades to clay several feet below the surface.

MATERIALS AND METHODS

Major Design Criteria

The major criteria to be addressed relate to hydrologic considerations, appropriate community types, and plant species selection for optimization of sediment retention and dissolved fraction uptake.

The primary hydrologic consideration involves achieving the greatest amount of dentention volume within the available space. During the initial planning stages of this project, approximately 10 ha (24 ac) of land was thought to be available for the marsh system; however, recent complications have led to the reduction of this figure to only 4 ha (10 ac). Of these 4 ha, about 0.6 ha are not suitable for inclusion in the marsh. After completion of the engineering design for the dike system, 2.6 ha (6.5 ac) of land remained as actual planting space. These developments affirmed the need to re-evaluate the storage volumn per unit area concepts. In this re-evaluation, the volumn of runoff from the watershed and the surface area of the marsh are the fixed parameters, with depth and geomorphological considerations the remaining variables.

After decisions were reached concerning the system's morphology, community types were reviewed to determine which type was most appropriate for achieving significant sediment retention and dissolved constituent uptake. Stem density was the primary consideration for sediment retention. Gleason (1979) found that stem density is directly proportional to sedimentation. Additionally, community types which might occur in meso-eutrophic environments were considered relative to their uptake of the dissolved constituents in stormwater. Such plant assemblages might be dominated by reed, floating leaves, or plankton (Hutchinson, 1975).

Finally, criteria for individual plant species within the chosen community types were considered. The primary criteria of depth and inundation tolerances were utilized in relation to the marsh's morphologic and hydrologic parameters. Other criteria were specific soil tolerances (i.e., will selected plants tolerate a range of soil types similar to the available soil), colonization habits (i.e., are selected plants aggressive enough to effectively colonize this site, yet not become a nuisance in adjacent areas), and morphological adaptations for effective sediment retention (i.e., high stem density, or special capabilities for uptake of dissolved elements).

Space Allocation

Consultation with staff hydrologists determined that the most effective morphological concept for the available land would be to compartmentalize the system (Figure 2). Stoplog spillways and dikes be constructed to create three compartments, in order to minimize constructions costs. In this way, each compartment can be designed to store varying depths of water, allowing greater diversity in storage volumns and plant communities. Some degree of diversity is considered desirable due to the diversity of contaminants in stormwater. Additionally, such compartments would be conducive to sheet flow and therefore more complete mixing and renovation of the stormwater. In order to maximize volumn capacity, each segment (Figure 3) will be designed to have a normal depth of 0.75 m (2.5 ft). This has been observed locally as a good to optimum depth for the communities chosen. By adjusting the number of stoplogs in the spillway, the water level within the entire system can be altered, with extremes being a total drydown of the septem or a depth 1.3 m (4 ft) greater than normal pool level.

Community Type Decisions

The first compartment will consist of a sawgrass (Cladium jamaicense) marsh, a relatively monotypic community type. This system has been chosen to perform a sediment retention function. The second compartment will be comprised of a bullrush (Scirpus californicus) marsh, which is also a relatively monotypic community. The Scirpus community will provide both sediment retention and dissolved pollutant uptake functions. The final segment shall be inhabited by a flag marsh, with emergent vegetation (Pontedaria sp. and Sagittaria sp.) dominating the shallow portions and floating leaf species (Nuphar luteum) occupying the deep

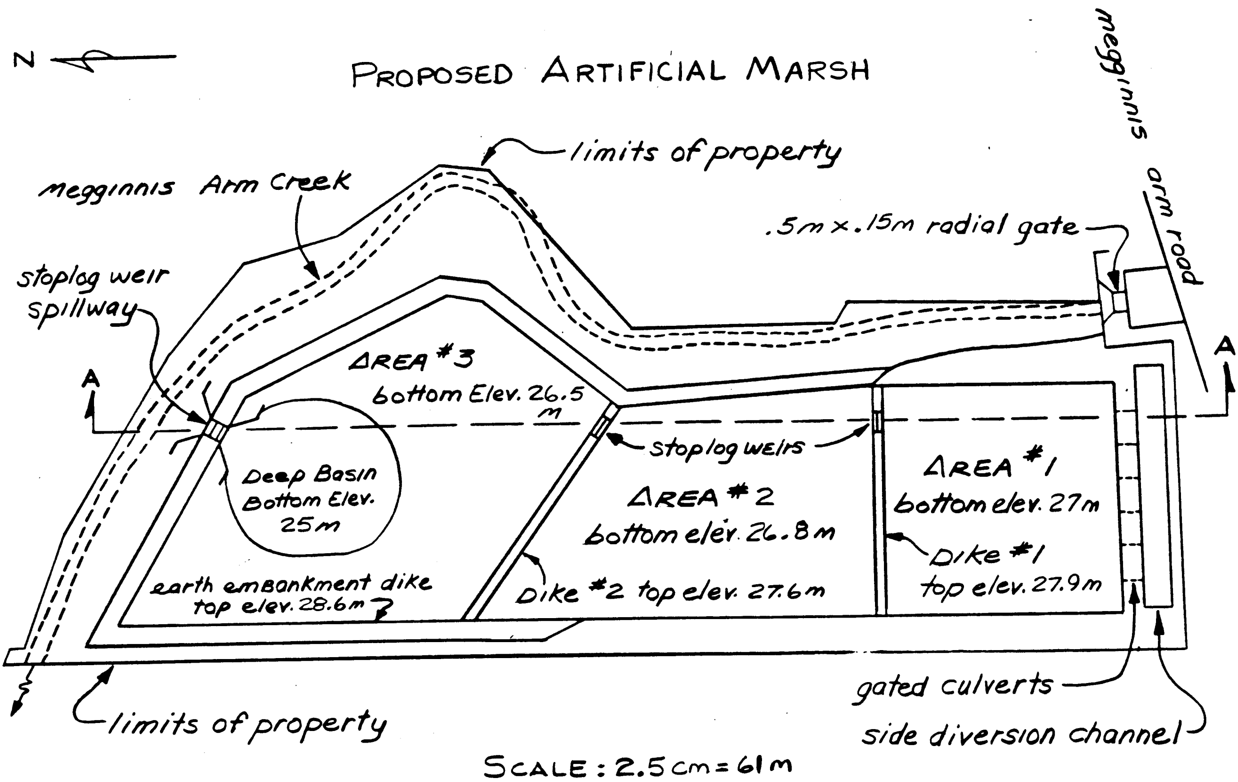

Figure 2. Plan view of proposed marsh

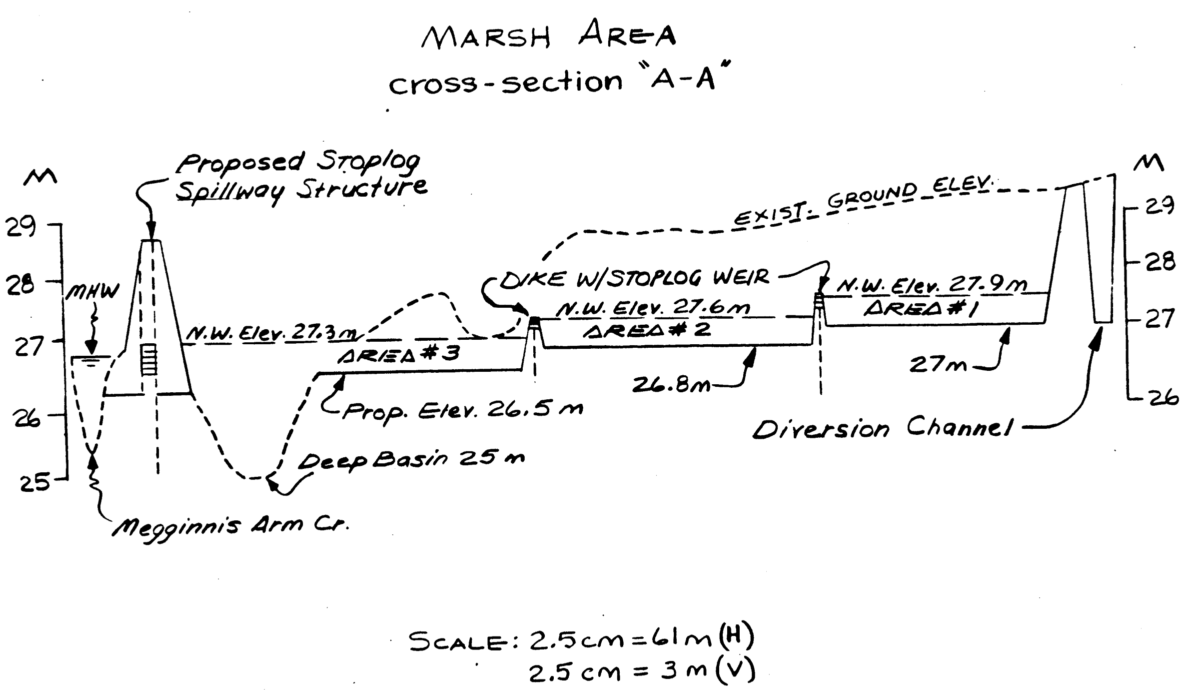

Figure 3. Profile of proposed marsh

basin. The functions of this segment will include sediment retention, but will be primarily nutrient uptake.

Individual Species Selection

The primary reason for selecting Cladium jamaicense is its high stem density, based on the fact that the major source of problems in Megginnis Arm is sediment-related, and considering that plant species having high stem densities are, morphologically, best suited for promotion of sediment retention (Gleason, 1979). Additionally, stands of C. jamaicense at the St. Marks Wildlife Refuge exist in 0.75 m of water most of the year, with some seasonal fluctuation occurring. C. jamaicense forms dense stands (Steward, 1975), largely via rhizomatous spreading and will grow on soils ranging from sand to marl to muck (Forthman, 1973), making C. jamaicense a practical choice regarding establishment. Its tendency to crowd out competitive species (Steward, 1975) increases its usefulness from the sediment retention viewpoint. Finally, this plant will persist throughout the winter, so that the seasonal effects upon renovation will be less pronounced.

The choice of Scripus californicus for the second compartment was made for virtually the same reasons, as Cladium jamaicense. Three additional features made this species attractive for our purpose. First, it was hypothesized that increasing the diversity of plants utilized would be wise. Secondly, Seidel (1976) has studied Scirpus lacustris extensively, documenting its ability to cleanse industrial and other types of wastewater. S. californicus is very similar to S. lacustris (Hutchinson, 1975) morphologically, anatomically, and ecologically (Godfrey, 1979), suggesting a likelihood of analogous chemical interactions in their environment. Lee (1976) asserts the Scripus spp. can be used for removal of dissolved pollutants and bacteria from wastewater. Finally, successful plantings of S. californicus were made at St. Marks Wildlife Refuge some time ago in habitat very similar to that of the proposed marsh. This, coupled with the fact that Scirpus spp. are known for rapid growth (Lee, 1976), should prove beneficial in the establishment phase.

The third compartment includes emergent, suberged and floating leaf species. Each type has unique advantages for this system.

The emergent plants will generally be represented by Pontedaria sp. and Sagittaria sp. The attractive features of these plants are their tolerance of long inundation periods and depths which range from zero to 1.3 m. These species also provide an excellent periphyton substrate. Periphyton and phytoplankton are expected to provide a large measure of dissolved nutrient uptake (Biederman, 1981).

The submerged plants will consist of Bacopa sp., Utricularia sp., and others. These are expected to derive dissolved nutrients from the water column (Sculthorpe, 1967), as well as from the soil, and to promote sediment retention due to the high stem densities. Of course, their preference for prolonged inundation and the depths provided makes them suitable for inclusion.

Finally, the floating leaf community will be represented by Nuphar luteum. The salient features of this species include its preference for continuous inundation, affinity for 1 m to 2.7 m depths, and ability to absorb significant amounts of phosphorus from the system throughout the year (Twilley, 1977).

CONCLUSIONS

Several questions are of interest upon consideration of the system. First, will this marsh system facilitate sediment retention? Second, will the system provide for adequate nutrient uptake? Third, will excessive nutrients be generated within the system?

There have been a few projects utilizing wetlands for stormwater. A few wetland treatment projects have just been completed in Minnesota; however, existing wetlands were utilized. Two artificial marshes have been constructed in New York, but are much smaller in scope. Although similar in design, these systems are not strictly comparable. Perhaps the most salient features of marsh systems are their relatively low cost and aesthetic value. Upon completion of this proposed system, a comparison of the renovation results with those of other systems can yield information which local governments can use to address their stormwater needs.

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