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May 19-20, 1983

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PROCEEDINGS OF THE TENTH ANNUAL CONFERENCE ON WETLANDS RESTORATION AND CREATION

May 19-20, 1983

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Frederick J. Webb, Jr.

Editor
Hillsborough Community College
P. O. Box 30030
Tampa, Florida 33630

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Introduction

The Annual Conference on Wetlands Restoration and Creation provides a forum for the nationwide exchange of results of scientific research in the restoration, creation, and management of freshwater and coastal systems. The Conference is designed to be of particular benefit to governmental agencies, planning organizations, colleges and universities, corporations and environmental groups with an interest in wetlands. These Proceedings are a compilation of papers presented at the Tenth Annual Conference.

This year's Conference would not have been possible without the assistance and cooperation of Mr. Roy R. Lewis, III. We are grateful for his untiring help and participation. Thank you is also extended to Alma S. Hires for providing Administrative support for the Conference.

The following people also deserve thanks for contributing to the preparation of the Proceedings for publication: L. Hobbs, C. Fields, H. Nunn, B. Franklin and G. Goldenberg.

The Proceedings could not have been completed without the time and efforts of the authors and reviewers.

To all these people thank you.

INVESTIGATION OF
ENVIRONMENTAL ENHANCEMENT NEEDS AND ALTERNATIVES
FOR THE LOXAHATCHEE SLOUGH/CANAL 18 BASIN, FLORIDA

Stephen R. Adams
Breedlove Associates Inc.
618 N.W. 13th Avenue
Gainesville, Florida 32601

ABSTRACT

INVESTIGATION OF ENVIRONMENTAL ENHANCEMENT NEEDS AND ALTERNATIVES FOR THE LOXAHATCHEE SLOUGH/CANAL 18 BASIN, FLORIDA

The Loxahatchee Slough/Loxahatchee River/Canal 18 Basin encompasses ~1140 km^2 in Martin and Palm Beach counties, Florida. Canal 18 is the major canal in the area controlling water levels within a 265 km^2 area known as the Loxahatchee Slough. The Loxahatchee Slough is considered the headwaters to the Southwest and Northwest forks of the Loxahatchee River. Canal 18, constructed in 1959, was identified as reducing historic flows to the Northwest Fork of the Loxahatchee River and contributing to saltwater intrusion; this has resulted in dieback of the predominantly freshwater vegetation and subsequent invasion by mangroves. Drainage activities in the Loxahatchee Slough have resulted in invasion by upland species and subsequent alteration of its wetland character.

Existing environmental conditions of the area were described through aerial photography, field reconnaissance, literature review, data search, and study of topography and drainage patterns. The current hydroperiod of the Loxahatchee Slough and flow regime of the Northwest Fork were also determined. To prevent further saltwater intrusion, water requirements of the Northwest Fork and a hydroperiod that would enhance wetlands in the Loxahatchee Slough were determined. A simplified mathematical simulation was prepared to evaluate three alternatives for environmental enhancement of the Loxahatchee Slough and Northwest Fork of the Loxahatchee River.

INTRODUCTION

The Loxahatchee Slough/Canal 18 Basin area is presently faced with numerous water and water-related problems. Increasing land use activity in the area is placing greater demands on resources and consequently has augmented public concern in four major water-related areas: 1) flood control, 2) water supply, 3) water quality, and 4) environmental preservation and enhancement.

This investigation was conducted for the U.S. Army Corps of Engineers (ACOE), Jacksonville District, Jacksonville, Florida, under contract No. DACW 17-80-C-0067.

This paper focuses only on the environmental preservation and enhancement aspect of the study which has been identified as a major concern in the project area (ACOE 1980). Of principal interest is maintaining a functional, yet viable, relationship between the Loxahatchee River, Loxahatchee Slough, and Canal 18.

AREA DESCRIPTION

The study area encompasses ~1140 km^2 in portions of southern Martin and northern Palm Beach counties, Florida (Figure 1). Canal 18 (C-18) is the major canal within the study area, authorized and constructed in 1959; it controls water levels within a 265 km^2 portion of the study area including the area known as the Loxahatchee Slough, which is considered the headwaters to the Southwest and Northwest forks of the Loxahatchee River. An automatic control structure (S-46) with manual override regulates discharges of C-18 into the Southwest Fork of the Loxahatchee River.

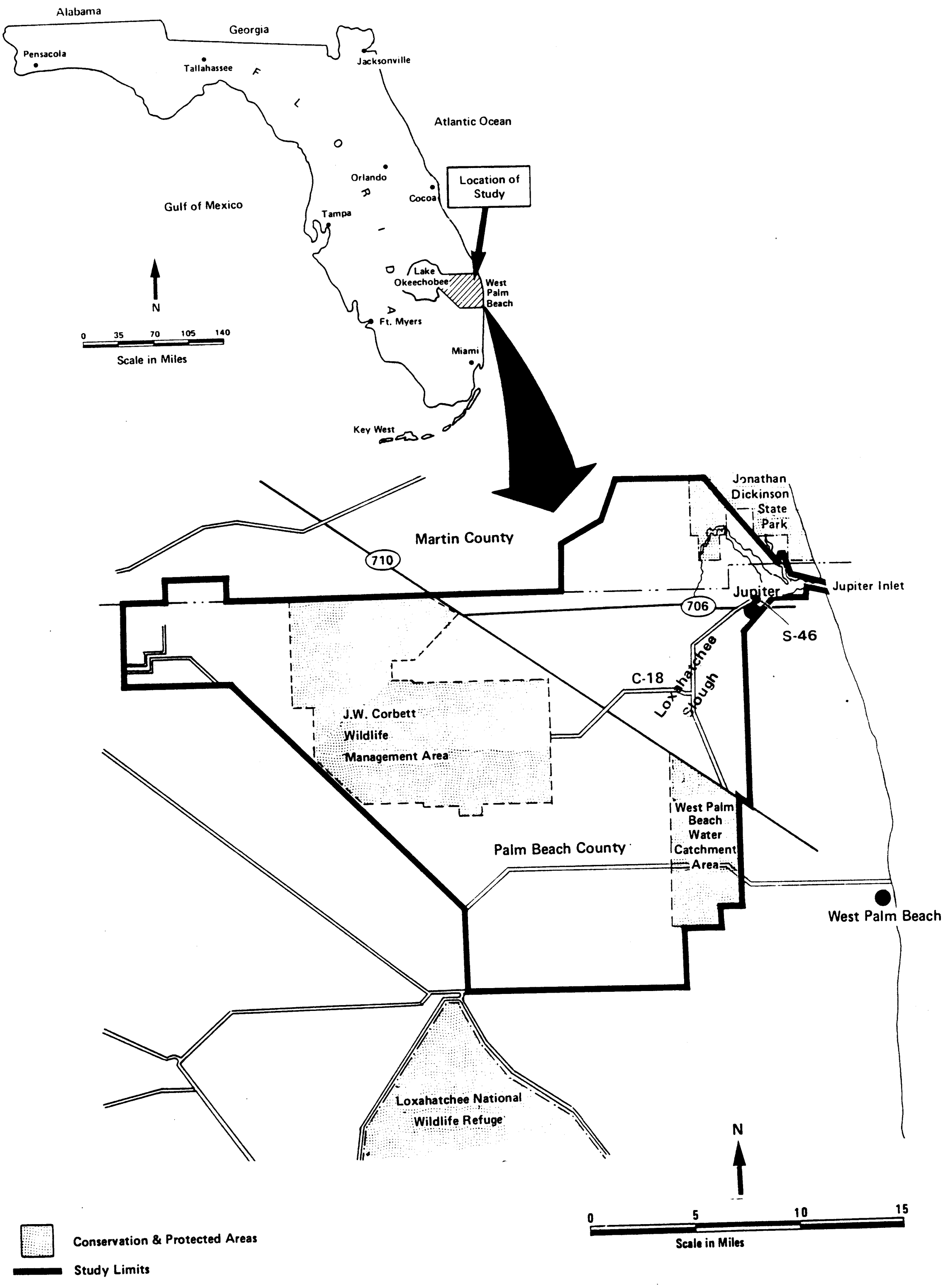

Figure 1. Study area.

Loxahatchee Slough

The Loxahatchee Slough has been affected by drainage activities in the region which have changed the original wetland character of the slough. Pre-drainage vegetation communities included wet prairie, marsh, swamp, and low hammock, with the first two being the predominant vegetation communities in the slough (Richardson 1977). Waxmyrtle (*Myrica cerifera*), saltbush (*Baccharis halimifolia*), Brazilian pepper (*Schinus terebinthifolia*), slash pine (*Pinus elliottii* var. *densa*), and *Melaleuca quinquenervia* are invading the slough area, and the exotic fern *Lygopodium microphyllum* has invaded disturbed wet sites in the slough area (Austin et al. 1977).

Northwest Fork of the Loxahatchee River

The Loxahatchee Slough originally formed the headwaters of the Northwest Fork of the Loxahatchee River. Construction of C-18 has been identified as reducing historic flows in the Northwest Fork and contributing to the deterioration of its scenic qualities. Specifically, maintenance of Jupiter Inlet in an open condition and reduction of historic flows to the Northwest Fork have resulted in saltwater encroachment upstream in the river, altering the predominantly freshwater swamp community and favoring invasion of mangroves.

METHODS AND MATERIALS

This study encompassed a literature search, data collection and professional analysis, and interdisciplinary synthesis of the collected information to identify water-related concerns and formulate management recommendations consistent with the Corps of Engineers planning objectives. A simplified computer simulation model of the basin was developed for testing the feasibility of the formulated plans (Figure 2). The model was not meant for final design purposes, but strictly as a management tool for decision-making. This model was based on existing information and calibrated using three years of actual rainfall and discharge data to insure accurate representation of the hydrological characteristics of the basin.

RESULTS

Loxahatchee Slough

Water levels in the Loxahatchee Slough are affected by several factors. The general lack of slope in the slough and preponderance of local depressions encourage local ponding. Project culverts were installed along the berm of C-18 to facilitate drainage of the slough area. These culverts are equipped with risers and flashboards set at specific elevations (ACOE 1956) to control water levels and prevent severe overdrainage. Topographic data for the slough and flashboard elevations indicate that flashboards in the project culverts are placed 0.6-1.8 m below the average ground elevation (~16 ft NGVD) in the slough, tending to lower the water table.

Water levels in C-18 and control of these levels by S-46 also influence the Loxahatchee Slough. Based on historical records, average water levels in the canal (range ~11-14.5 ft NGVD) have been well below average ground elevations in the Loxahatchee Slough, even with changes in operation schedules of S-46 to provide for more storage and thus higher water levels in C-18. These factors have caused the slough to be overdrained which, in turn, has resulted in altered vegetation patterns within the slough. Existing wet season water table elevations in the slough average ~16 ft NGVD which is about average ground elevation (Schneider 1977). During the

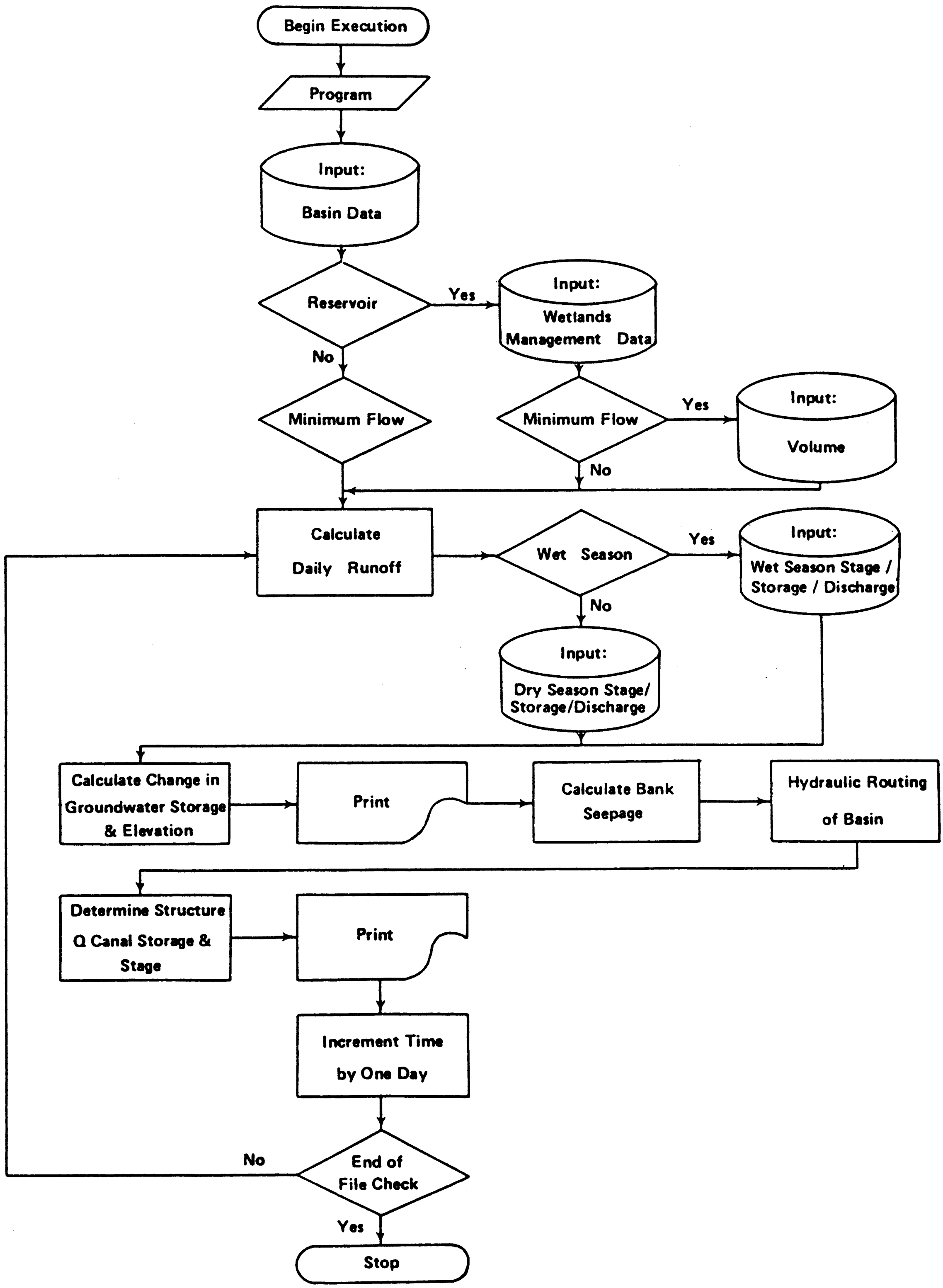

Figure 2. Model flowchart.

dry season the water table is generally well below the average ground elevation of the slough.

The U.S. Fish and Wildlife Service (1981) recommended a generalized water regulation schedule for the slough which would aid in restoration of desirable plant communities, particularly wetlands. Their recommendation called for creation of a water regime with 50-80% inundation of land surface during a normal rainfall year, a water depth of no greater than 0.6 m (18 ft NGVD) above average land surface elevation at the end of the rainy season, with a "natural" recession to no surface water by mid-March for a 60-90 day period to allow for plant germination.

A 75% inundation schedule was chosen (Figure 3) as the optimum for the slough. Based on the inundation curve and slough topography, the following vegetation communities and relative occurrence could theoretically be expected in restoration:

Wetland/Plant Community	Percent Occurrence
Transition to upland	2.7
Transition to sawgrass	6.7
Sawgrass and transition to wet prairie	6.2
Wet prairie and transition to aquatic	58.7
Aquatic/open water	25.7

Northwest Fork of the Loxahatchee River

Reduction in the historic flow regime on the Northwest Fork, coupled with maintenance of a continuous open inlet at Jupiter Inlet, has resulted in encroachment of saltwater into the lower reaches of the Northwest Fork. This encroachment has led to dieback of the predominantly freshwater swamp system along the Northwest Fork and invasion of mangroves in the area. Presently, the Northwest Fork is fed by C-14, which is the major north-

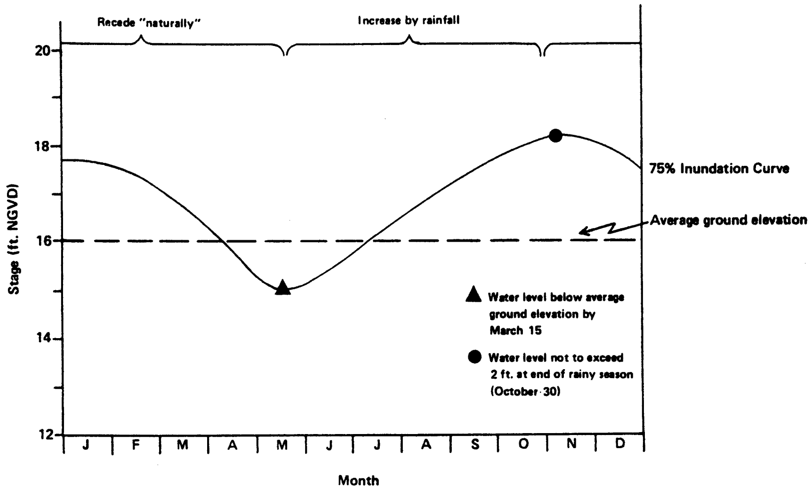

Figure 3. General recommended inundation curve for Loxahatchee Slough.

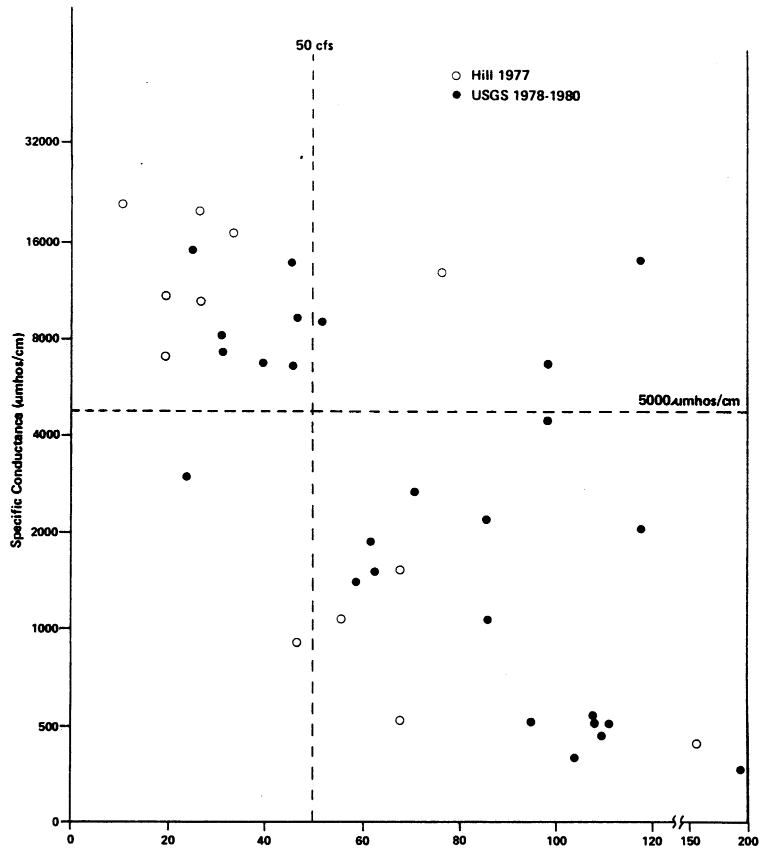

Figure 4. Specific conductance and flow relationships in the Northwest Fork of the Loxahatchee River.

south collection channel for the South Indian River Drainage District, and a diversion culvert from C-18 installed in 1975 to supplement flows into the Northwest Fork (Rodis 1973).

Although no long-term data exist for correlating flows in the Northwest Fork and salinity levels, two short-term studies indicate a relationship between flows in the Northwest Fork and salinity encroachment (Birnhak 1974, Hill 1977). A minimum flow regime of 50 cfs would be needed to maintain the freshwater integrity of the upper reaches of the Northwest Fork (Figure 4). Discharge data for the Northwest Fork indicate that the 50 cfs base flow is not maintained during the dry season (Figure 5), even after installation of the C-18 diversion culvert which was intended to supplement flow to maintain the 50 cfs base flow.

An analysis of discharge data from the C-18 diversion culvert and C-14 revealed that the 50 cfs base flow can be easily maintained except at the end of the dry season (April and May) (Figure 6). However, there are periods when the C-18 diversion culvert does not appear to be operating according to schedule, i.e., supplemental flows have been released from the C-18 diversion culvert when the Northwest Fork flow is at or above the base flow of 50 cfs. However, it still appears that the 50 cfs base flow could not be maintained during the latter months of the dry season even if the structure operated according to the operation schedule.

DISCUSSION AND CONCLUSIONS

Three water management alternative plans were developed to address problems identified in each of the four major water-related areas: 1) flood control, 2) water supply, 3) water quality, and 4) environmental preservation and enhancement.

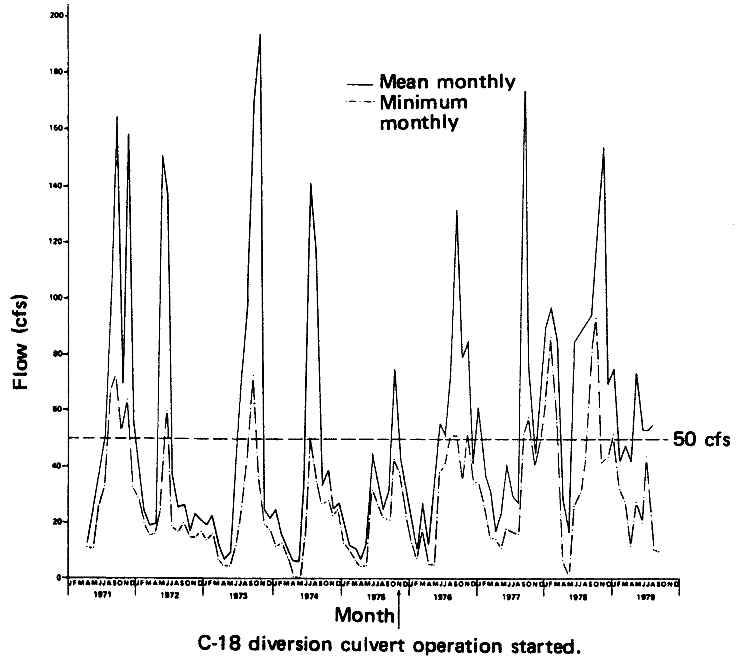

Figure 5. Mean and minimum monthly flows of the Northwest Fork of the Loxahatchee River.

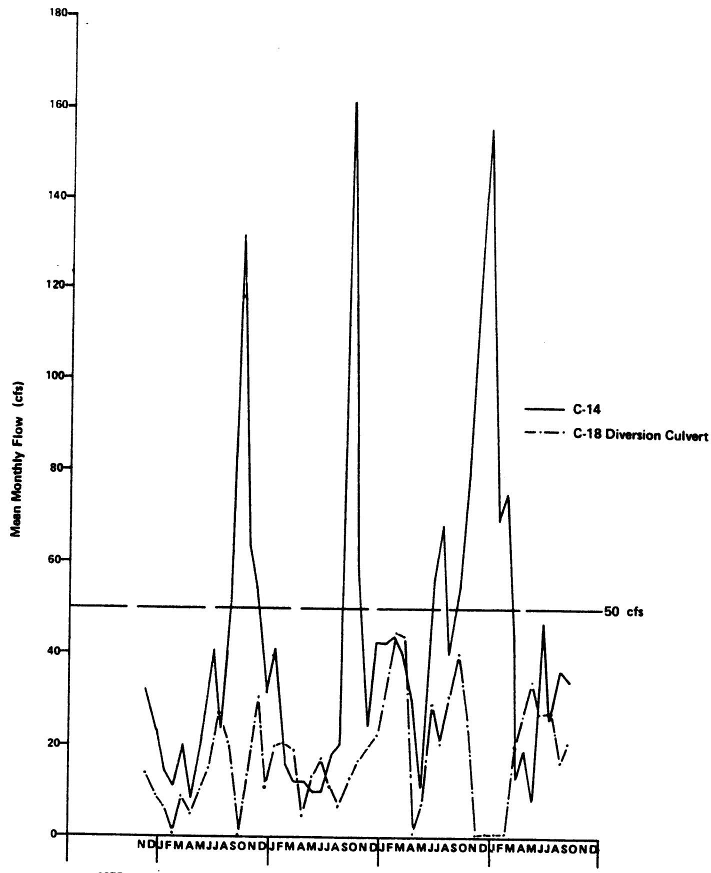

Figure 6. Mean monthly flow at C-18 diversion culvert and C-14.

The first water management plan requires the least degree of physical alteration to the C-18 Basin; the second builds on the alternatives of the first, and the third, the most complex, builds on alternatives of water management alternative plans 1 and 2. From the environmental preservation and enhancement aspect of the project, water management alternative plans 2 and 3 are basically the same; the major difference lies in alternatives to solutions of the area's potential water supply problems. Therefore, only water management alternative plans 1 and 2 will be discussed.

Water Management Alternative Plan 1

Alternative Plan 1 requires minor physical modifications to the C-18 Basin. For enhancement of the Loxahatchee Slough, it was recommended that the flashboards in the project culverts be adjusted and manipulated to create a 75% inundation in the slough. The height of adjustment would need to be closely monitored to maintain flood control in low-lying developed areas. The manipulation of the project culverts to simulate a more natural hydroperiod would aid in maintaining or enhancing existing wetland communities and in reestablishing more wetland communities.

It was also recommended that the C-18 diversion culvert be modified by installing a telemetry system to allow more efficient use of the system. The structure could be calibrated to a salinity monitoring device on the Northwest Fork so that flows released from the C-18 diversion culvert would be related to salinity levels rather than a minimum discharge or base flow. This would possibly aid in more conservative use of available water by delivering supplemental water on a more consistent basis. However, this assumes that water will be available from C-18, and based on historic records, C-18 cannot provide water in the critical period of need (dry season), as stages in C-18 are too low for discharge.

Water Management Alternative Plan 2

This plan includes the creation of a wetlands management area utilizing a levee system along the approximate historic slough boundary (Figure 7). Water level control structures would be needed as well as a perimeter ditch/pump system to capture seepage water and return it to the wetlands management area. The modifications of the C-18 diversion culvert, as identified in Plan 1, would also be incorporated into this plan.

Water levels in the wetlands management area can be managed to simulate the recommended hydroperiod for restoration or enhancement of wetland plant communities. Additionally, water would be potentially available for the Northwest Fork to maintain freshwater integrity in the upper reaches.

The computer model was used to test the feasibility of this plan for simulating the recommended hydroperiod in the slough and maintaining freshwater availability to the Northwest Fork. Two simulations were run, the first for a "normal year" and the second for a "dry year" (Figure 7).

Examination of the graph (Figure 7) indicates that a 50 cfs base flow could be maintained throughout the "normal" and "dry" years. However, based on additional runs, a 50 cfs base flow could not be maintained for two consecutive "dry" years. Examination of stages in the wetlands management area indicates a 69% inundation frequency for a "normal" year, well within the 50-80% inundation range desired by the U.S. Fish and Wildlife Service (FWS) for maintenance of wetland communities (FWS 1981). The "dry" year inundation frequency was calculated to be 32%. This should not be considered detrimental, as undisturbed systems periodically are subjected to low inundation frequencies during "dry" years. Based on the

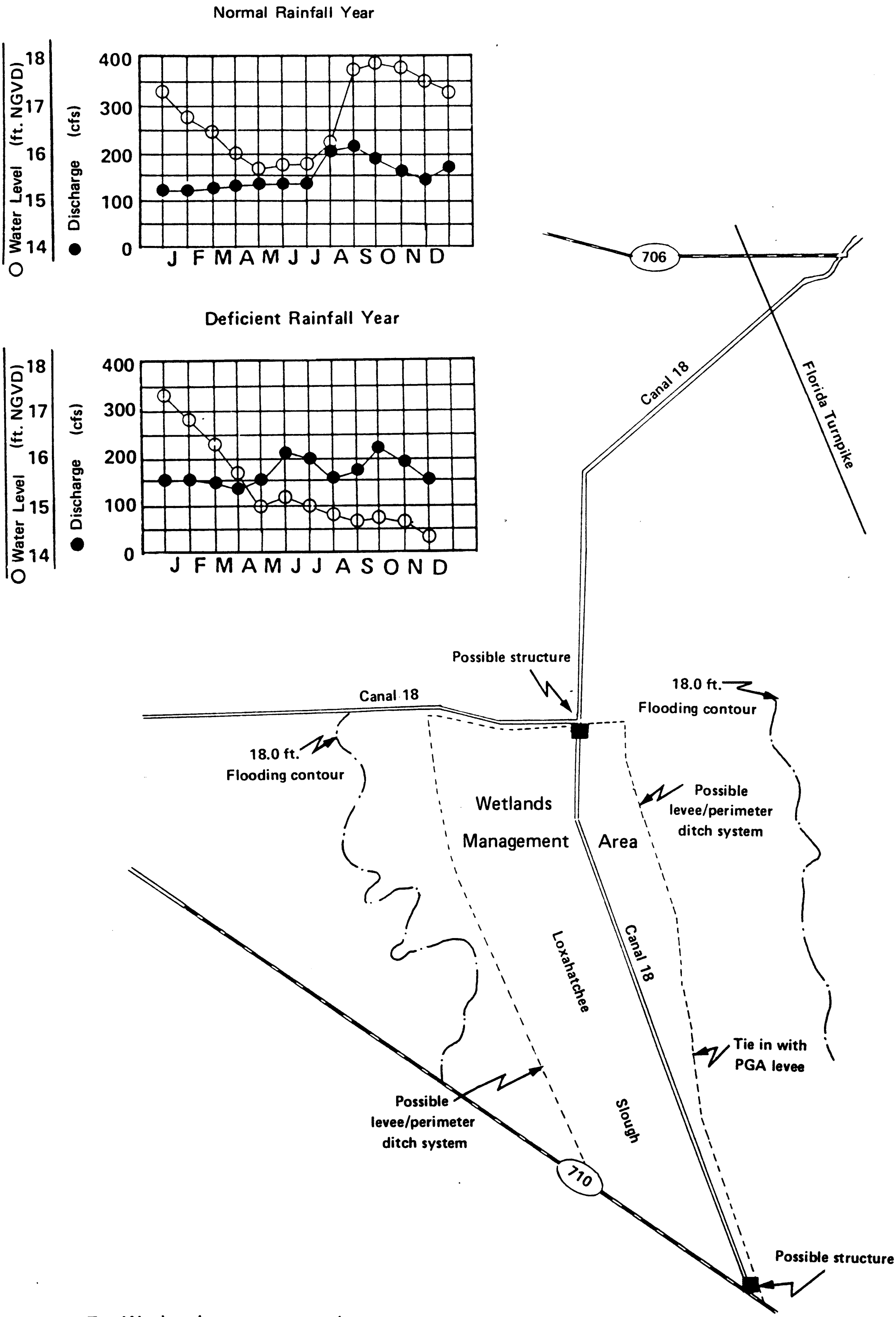

Figure 7. Wetlands management area.

69% inundation frequency and topographic features of the wetlands management area, the following percentages of wetland plant communities could theoretically be established within the wetland management area:

Wetland Plant Community	Percent Occurrence
Transition to upland	2.7
Transition to sawgrass	12.8
Sawgrass and transition to wet prairie	6.9
Wet prairie and transition to aquatic	51.9
Aquatic/open water	25.7

The results of the computer modeling indicate that implementing this alternative would result in potential establishment of desirable wetland communities in the wetlands management area and maintenance of the base flow in the Northwest Fork of the Loxahatchee River.

ACKNOWLEDGEMENTS

The author expresses sincere thanks to Mr. K. Dan Shalloway, Shalloway, Inc., Lake Worth, Florida and Mr. Harry Oleson, Leggette, Brashears & Graham, Inc., Tampa, Florida for their participation and contribution to this project. The author would also like to thank the following individuals and agencies who provided data and constructive suggestions during the course of the project: Mr. Alan Lauwaert and Jed Carlton, Jacksonville District, U.S. Army Corps of Engineers; Dr. Ben McPherson, U.S. Geological Survey; Mr. Joe Carrol and Mr. Joe Johnston, U.S. Fish and Wildlife Service; the staff of the South Florida Water Management District; National Park Service; U.S. Environmental Protection Agency; Florida Department of Natural Resources; Florida Department of Environmental Regulation; Florida Game and Fresh Water Fish Commission; Treasure Coast Regional Planning Council; Area Planning Board of Palm Beach County; Loxahatchee River Environmental Control District; Jupiter Inlet Commission; and the Florida Wildlife Federation.

LITERATURE CITED

Austin, D.F., K. Coleman-Marois, and D.R. Richardson. 1977. Vegetation of southeastern Florida II-V. Fla. Sci. 40(4):331-361.

Birnhak, B.I. 1974. An examination of the influence of freshwater canal discharges on salinity in selected southeastern Florida estuaries. USDI. Bur. of Sport Fisheries and Wildlife. 40 pp.

Hill, G.W. 1977. Loxahatchee River Oyster Bar Removal Project: salinity monitoring phase. Final Letter Report--U.S. Geological Survey, Water Resources Division, Jupiter, Florida.

Richardson, D.R. 1977. Vegetation of the Atlantic coastal ridge of Palm Beach County, Florida. Fla. Sci. 40(4):281-330.

Rodis, H.G. 1973. The Loxahatchee--a river in distress, southeast Florida. Prep. by USGS in cooperation with the Florida Department of Nat. Res. and Central and South Florida Flood Control District.

Schneider, J.J. 1977. Water table contours in Palm Beach County, Florida. USGS Open File Report 77364.

U.S. Army Corps of Engineers (ACOE). 1956. General and detail design memorandum, Canal 18 and Control Structure 46. Central and Southern Florida Project, Supplement 20. Department of the Army, Jacksonville District, Corps of Engineers, Jacksonville, Florida.

U.S. Army Corps of Engineers (ACOE). 1980. Reconnaissance report: review of Canal 18, Jupiter Inlet, and Loxahatchee River--Central and Southern Florida Project. Department of the Army, Jacksonville District, Corps of Engineers, Jacksonville, Florida. 34 pp.

U.S. Fish and Wildlife Service. 1981. Environmental investigation of the Loxahatchee Slough. Draft report. USDI, U.S. Fish and Wildlife Service, Vero Beach, Florida.

ZOOPLANKTON/TROPHIC STATE RELATIONSHIPS AND THE POTENTIAL FOR PREDICTION OF ECOSYSTEM STRUCTURE IN RECLAIMED FLORIDA LAKES

by: James S. Bays
EcoImpact, Inc.
417 S.E. 2 Street
Gainesville, FL 32601

ABSTRACT

Nutrient loading models can provide useful projections of trophic state during the design phase of man-made lakes. This method could be extended to projections of aquatic ecosystem structure by applying empirical relationships determined in this study between the zooplankton and fish biomass and trophic state of 39 Florida lakes.

Positive regressions were obtained between the zooplankton and lake trophic state, with the degree of significance roughly inversely proportional to the size of the plankter. Total fish biomass was significantly and positively correlated with lake trophic state, but the fish community composition varied significantly with increasing trophic state.

Application of these statistical relationships during the design process, and the inclusion of parameters such as zooplankton density in monitoring programs, could be valuable supplements to more conventional water quality analyses, and could yield substantial insight into the changes in lacustrine ecosystems due to human influence.

INTRODUCTION

Statistical models now exist for the prediction of trophic state in Florida lakes (Shannon and Brezonik, 1972; Baker et al. 1981; Huber et al. 1982). Such models are empirical equations that predict: (1) phytoplankton biomass, as measured by chlorophyll *a*, from total phosphorus or nitrogen concentrations; (2) total phosphorus or nitrogen concentrations from their respective loading rates and (3) nutrient loading rates from watershed land use and geological characteristics. Hydraulic loading rates are frequently coupled with phosphorus loading rates or depth in the development of these models.

These statistical tools provide a rough first cut in the prediction of the trophic state of restored or reclaimed lakes based on measured or estimated nutrient loading rates and watershed land use. However, little information and few models exist for the prediction of open water ecosystem structure in lakes in relation to trophic state. The development of such models could contribute towards both the proper management of freshwater fisheries as well as a better understanding of the factors controlling eutrophication in lakes.

Models have long existed for the estimation of fish yield from varying measures of lake trophic state (see Ryder et al. 1982 for a review), and recently, empirical equations have been published for the prediction of standing stock of crustacean zooplankton or percent dominance by large and small phytoplankton from phytoplankton standing crop in temperate zone lakes (McCauley and Kalff, 1981; Watson and Kalff, 1981). Few studies have been done on this subject for Florida lakes (e.g. Kautz, 1980). This paper presents empirical relationships between measures of zooplankton standing stock and lake trophic state in 39 natural Florida lakes, as well as preliminary relationships between fish biomass and community structure and lake trophic state.

METHODS

Crustacean zooplankton, rotifer and ciliated protozoan samples were taken monthly during 1979 from 20 lakes distributed throughout peninsular Florida. This data base was combined with similar data taken by other researchers during the same approximate time period for 19 other lakes of varying trophic state. The main criterion for selection of these lakes was the existence of fish population block net data, some of which has been published by the Florida Game and Freshwater Fish Commission (FGFWFC, 1972; 1975; 1976).

Zooplankton other than protozoans were sampled by vertical tows with an 80 um mesh Wisconsin net at a central station in each lake. Protozoans were collected with a Kemmerer bottle at different depths at the same station and were integrated into replicate samples. Protozoan sampling methodology, as well as preservation, enumeration and biomass conversion information, is given in detail in Bays and Crisman (1983) and Beaver and Crisman (1982).

Lake trophic state was estimated in this study by Carlson's trophic state index based upon chlorophyll *a*, and is referred to here as TSICHL (Carlson, 1977). Chlorophyll *a* was determined by the trichromatic method from a composite water sample taken from each lake (APHA, 1975). TSICHL is the natural log transformed chlorophyll *a* concentration scaled to a value between 0 and 100 with values toward the high end of the scale being eutrophic. For Florida lakes, TSICHL values up to 45 are oligotrophic; between 45 and 55, mesotrophic; between 55 and 70, eutrophic and greater than 70, hypereutrophic (Baker et al. 1981).

Although trophic state may be more accurately estimated by using a multivariate statistic, the univariate statistic based on algal biomass used here is well correlated with more complex measures of trophic state (Brezonik, 1976), is simpler to acquire and process (Carlson, 1977), provides the best regression with zooplankton parameters (Beaver, 1980; Bays, unpublished data), and is a valid estimator of the primary food resources for zooplankton -- phytoplankton and bacteria.

All calculations and statistical analyses were performed using the SAS package (Helwig & Council, 1979). Computing facilities were provided by the Northeast Regional Data Center of the State University of Florida, at the University of Florida in Gainesville.

RESULTS

The regression between log transformed total zooplankton biomass and lake trophic state is positive and statistically significant (r^2=0.72, $p<0.001$; Fig. 1; Table 1). Microzooplankton biomass (rotifers, nauplii, protozoans) was positively and significantly related to lake trophic state (r^2=0.76, $p<0.001$). The relationship of macrozooplankton (cladocera, cyclo-poid and calanoid copepods) to lake trophic state was also statistically significant (r^2=0.15, $p<0.001$), but showed a high degree of scatter.

This information was also expressed by plotting the regression of the percent composition of macro and microzooplankton against lake trophic state (Fig. 2; Table 1). While there was a substantial degree of scatter between lakes over the entire trophic gradient, the regressions are sig-

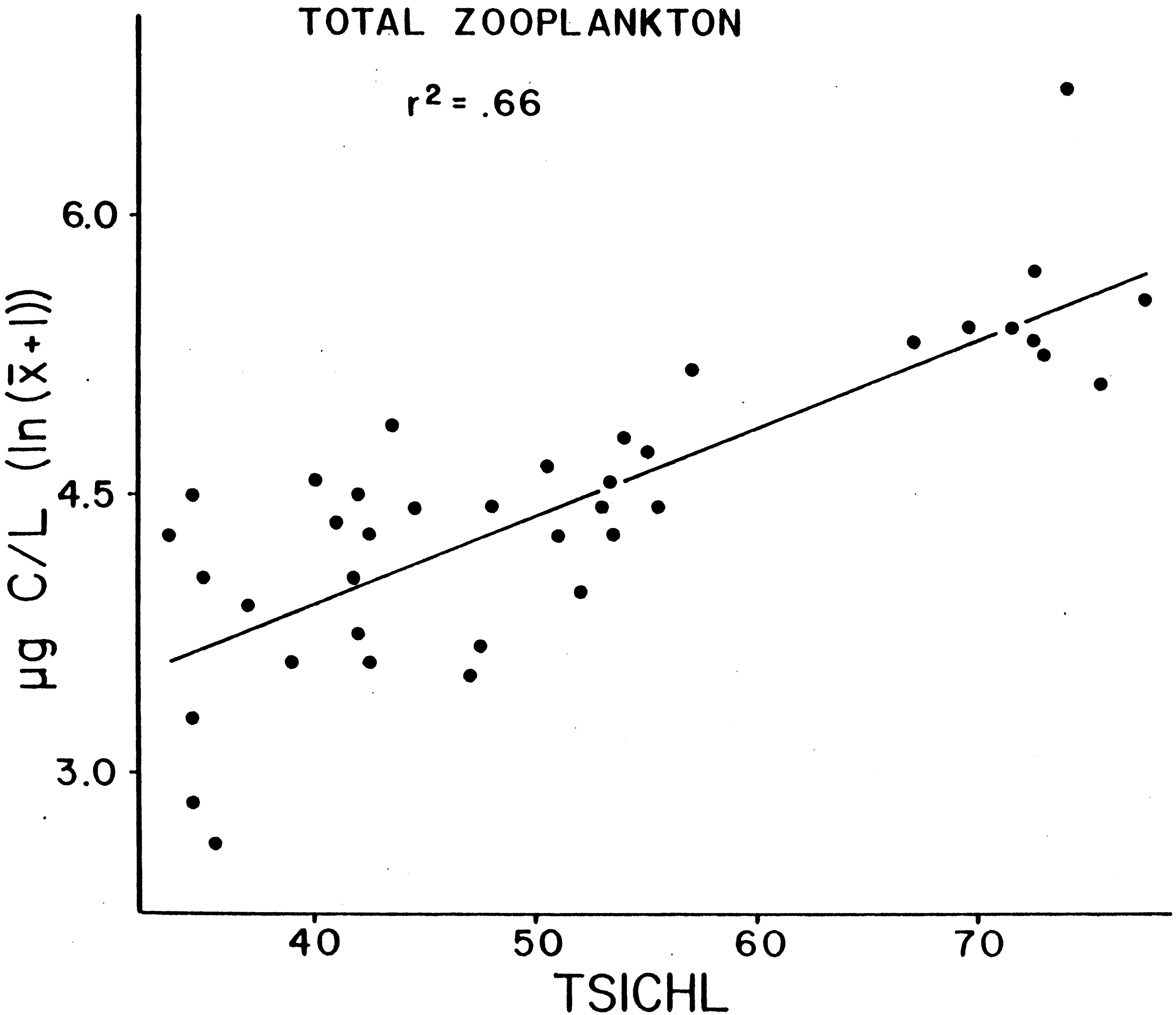

FIGURE LEGEND

Figure 1. Regression plot of total zooplankton biomass (ug C/L) versus Carlson's trophic state index (TSICHL).

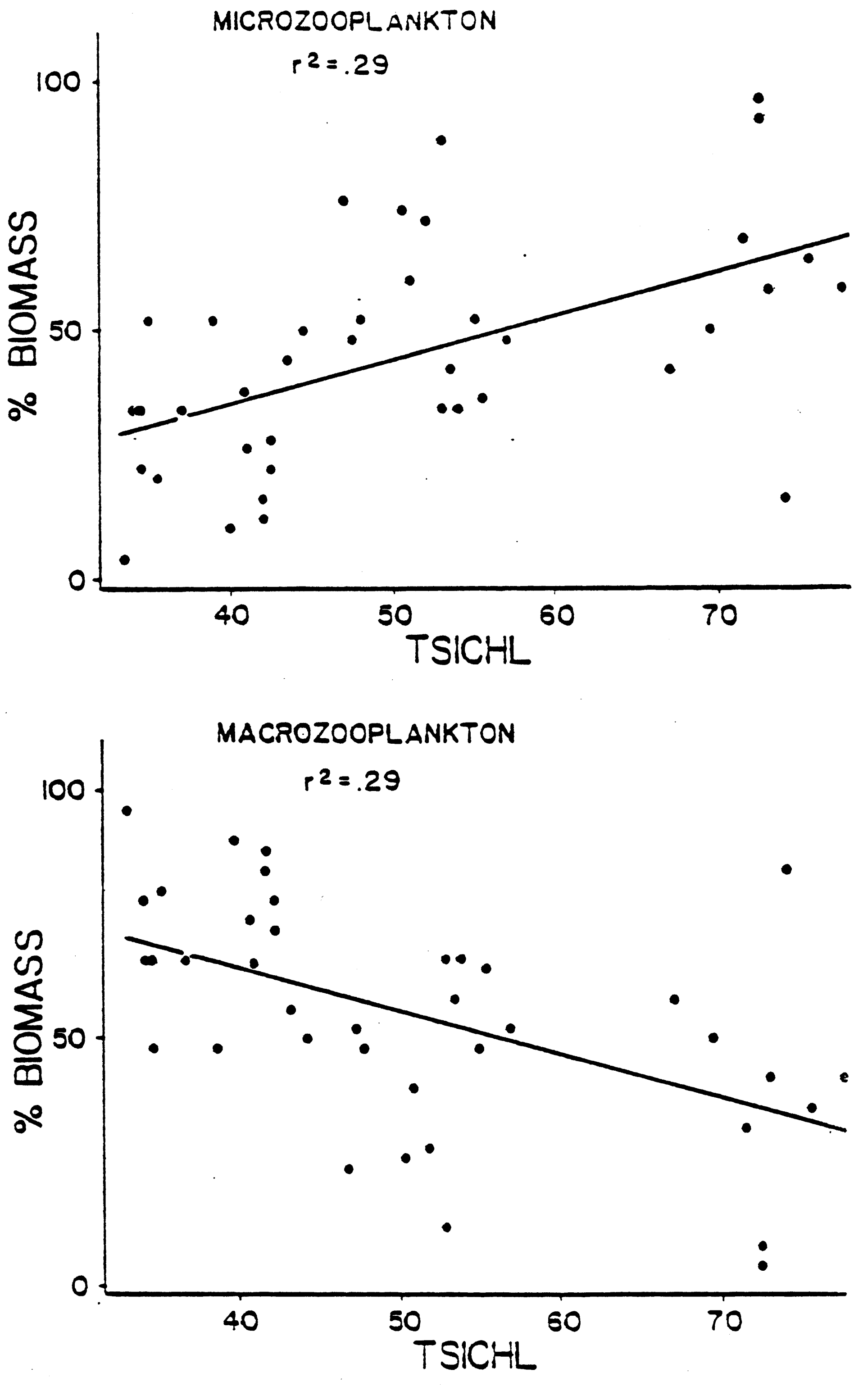

Figure 2. Regression plots of microzooplankton and macrozooplankton biomass (as percent of total zooplankton biomass) versus Carlson's trophic state index (TSICHL). (Bays and Crisman, 1983)

nificant (r^2=0.29; p 0.001), and show the increasing percentage of microzooplankton and the decreasing percentage of macrozooplankton in the zooplankton standing stock with increasing trophic state.

Another aspect of this study was to assess fish community structure and abundance over a trophic state gradient in Florida lakes. Biomass data from approximately one-third of the lakes were combined with similar data from 20 other lakes of known water quality taken from unpublished summaries and annual reports of the Florida Game and Freshwater Fish Commission (FGFWFC, 1972; 1975; 1976) and regressed against Carlson's Trophic State Index. Figure 3 and Table 2 show that total fish biomass, expressed as the natural log of kilograms wet weight per hectare, yielded a significant and positive relationship (r^2=0.42, p 0.001) with lake trophic state, despite a fairly high degree of scatter.

The change in community composition illustrates the well-documented shift in dominance from sport fish to rough fish with increasing trophic state (Fig. 3; Table 2). Sport fish, here primarily represented by bass and bluegill biomass, decrease in percentage with increasing trophic state, even though the biomass increases. Rough fish, such as gizzard shad and gar, show a highly significant (p 0.001) increase with trohic state. Commercial fish (e.g. catfish), which are not depicted here, increase in parallel with the rough fish.

DISCUSSION

Zooplankton-Trophic State Regressions

The relatively low amount of variation around the regression line of microzooplankton and the high amount of variation in the macrozooplankton are related to the effects of food resource composition and predator com-

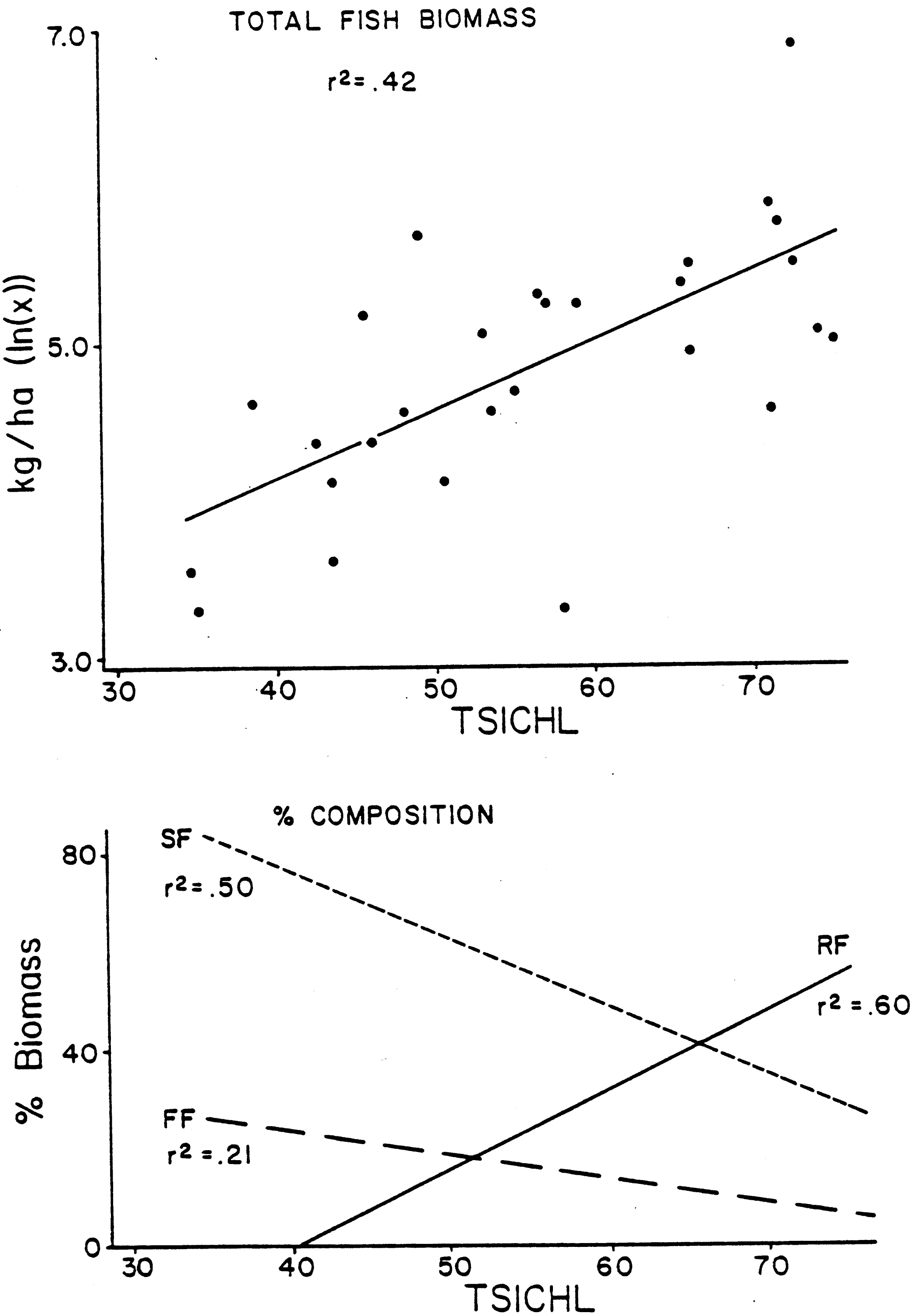

Figure 3. Regression plot of total fish biomass (log kg/ha), upper panel), and predicted percent composition of various fish groups (lower panel), versus Carlson's trophic state index (TSICHL). SF = sport fish, FF = forage fish, and RF = rough fish. See text for details. (Bays and Crisman, 1983)

munity structure on the zooplankton communities. As trophic state increases, the food resources that are available to the zooplankton undergo qualitative changes in composition. Phytoplankton of oligotrophic lakes in Florida may be characterized by the dominance by algal species that are nutritious and edible by most herbivorous zooplankton species. However, the algal composition shifts with increasing trophic state towards dominance by larger filamentous or colonial blue-green algae (Crisman, 1981). Generally, these algae are less edible and nutritious and are less capable of supporting the macrozooplankton populations (Porter, 1977).

Planktonic bacteria populations tend to increase with nutrient enrichment and provide an increasingly abundant food resource for those zooplankton able to utilize them (Porter, 1977). Microzooplankton utilize bacterioplankton at a much greater efficiency than macrozooplankton (Bogdan et al., 1980; Beaver, Crisman and Bays, 1981; Peterson et al., 1978). Macrozooplankton are relatively more efficient at using larger algal forms, but are unable to fully utilize filamentous and colonial blue-greens (Porter, 1977). Macrozooplankton are also subject to predation in increasing amounts with increasing trophic state, due to the increased abundance of zooplanktivorous fish, such as gizzard shad (Drenner et al., 1982). Consequently, the microzooplankton respond positively and significantly to increasing trophic state, due to increasing available food resources, while macrozooplankton tend to increase only slightly and show a less uniform response to nutrient enrichment due to increased vertebrate predation and decreased nutritional acceptability and edibility of food resources.

Fish-Trophic State Regressions

The relative degrees to which the fish biomass of the different categories respond to increasing trophic state is related to a complex of factors that include changes in food quality and quantity, nesting substrate composition, tolerance to low dissolved oxygen conditions and breeding habitat quality and distribution. (See Larkin and Northcote (1969) and Kautz (1980) for good reviews and analysis of this subject.)

The regressions between fish biomass and trophic state presented here were determined as an aid in interpreting the zooplankton regressions, and are preliminary, due to the inherent variability in the fish sampling methodology. However, the utility of this form of analysis is evident and can provide a first cut at prediction of fish community structure.

Application of Biota-Trophic State Relationships to Lake Reclamation

Some attempt at trophic state prediction, with knowledge of nutrient loading rates and watershed land use, should be made in lake design or restoration in order to anticipate potential water quality problems. From this trophic state assessment, given the sort of preliminary empirical equations presented here, it would be relatively simple to predict a range of densities for different zooplankton species, as well as an approximate level of fish production and species composition. Fisheries composition and harvest potential are important recreational and economic considerations, while predictions of abundance and type of zooplankton provide information on aspects of the lake ecosystem function. For example, while nutrient loading, sediment-water exchange of nutrients and morphometry are primary determinants of the overall trophic state, processes within the lake may serve to make nutrients more or less available to algae. Microzooplankton have very short lifespans, and cycle nutrients more rapidly than macrozooplankton (Peters and Rigler, 1973). These nutrients are cycled in forms that are generally taken up rapidly by algae. It is possible that nutrient cycling by such zooplankton could become quite significant, and blooms of algae, or at least high algal populations, would be expected.

Conversely, the zooplankton of lakes that are predicted to be oligotrophic will be dominated by macrozooplankton, specifically the cladocera and copepods. These herbivorous grazers not only cycle nutrients to a lesser degree than microzooplankton, but certain species are effective grazers of algae, and can efficiently consume large proportions of the standing crop, thereby enhancing the recreational qualities of a lake by increasing the transparency of the water (Shapiro, 1979). Microzooplankton are constrained in their use of the spectrum of phytoplankton, and are far less efficient at consuming algae that can cause blooms. Therefore, conditions which favor macrozooplankton populations can be naturally beneficial.

Oligotrophic lakes in this study were all 4-20 meters deep. Lakes that are designed with depths near this range can stratify and will develop hypolimnetic refugia in which vertically migrating macrozooplankton can escape predation by fish, particularly the larval life stages of fish. The mean depth of the eutrophic lakes ranged from 1 to 3 meters. This effectively makes most of the lake depth available for planktivorous fish to prey on zooplankton and causes both a relatively greater reduction in their populations and selection towards smaller sizes. Lower standing stocks of zooplankton or a predominance of microzooplankton in shallow lakes will reduce the effectiveness of such a natural control on algal populations.

Zooplankton are engaged in important roles in the aquatic ecosystem and provide ecological services that help to maintain a stable ecosystem and good lake water quality. Steps can be taken in the design of lakes that will create aquatic habitats that will enhance their populations.

Depending on the nutrient loading rates, deeper lakes will be dominated by macrozooplankton and sport fish populations. Shallow lakes will have an abundance of microzooplankton and rough fish. Consequently, populations of undesirable bloom-forming algae will be subject to natural checks in deeper, more oligotrophic lakes, while in eutrophic lakes, algal populations are more likely to be subject to less control by zooplankton, and if anything, will be enhanced by the activities of the microzooplankton.

This paper presents a series of statistical models that can facilitate the prediction of lake trophic state and certain aspects of its aquatic ecosystem structure, such as the zooplankton community. This allows for an increasingly quantitative element to be introduced into lake design. As models such as these become more sophisticated, it will become possible to work toward an ecological optimization of reclaimed lake characteristics. Until then, more research is required on artificial lakes in Florida, and these statistical models will have to be tested and refined for a wide variety of lakes.

ACKNOWLEDGEMENTS

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TABLE 1

Table 1. Model parameters for regressions of log transformed mean annual zooplankton biomass (ln (x + 1) and percent biomass composition as the dependent variables and Carlson's TSI (TSICHL) based on mean annual chlorophyll a as the independent variable.

Dependent Variable	Slope	Intercept	r^2	P<
Total	0.048	2.003	0.66	0.0001
Microzooplankton	0.069	0.0035	0.76	0.0001
Macrozooplankton	0.024	2.512	0.15	0.0159
Microzooplankton as percent of total	0.898	-0.879	0.29	0.0004

TABLE 2

Table 2. Model parameters for regression models of total fish biomass and percent composition of sport, forage and forage fish against lake trophic state (TSICHL).

Dependent Variable	Slope	Intercept	r^2	P<
Log total Fish Biomass	0.0425	2.452	0.42	0.0002
Percent Sport Fish	-1.374	131.307	0.50	0.0001
Percent Rough Fish	1.593	-64.199	0.60	0.0001
Percent Forage Fish	-0.540	45.582	0.21	0.0120

LITERATURE CITED

American Public Health Association. Standard Methods for the Examination of Water and Wastewater. 14th ed. Washington, D.C. 1193 p.

Bays, J. S. and T. L. Crisman. 1983. Zooplankton-trophic state relation ships in Florida lakes. Can. J. Fish. Aquat. Sci. 40:1813-1819.

Baker, L. A., P. L. Brezonik and C. R. Kratzer. 1981. Nutrient loading-trophic state relationships in Florida lakes. Water Resources Research Center Publication No. 56, University of Florida, Gainesville.

Beaver, J. R. 1980. The distribution of planktonic ciliated protozoans in 30 Florida lakes of varying trophic state. M.S. Thesis, University of Florida.

Beaver, J. R. and T. L. Crisman. 1982. The trophic response of ciliated protozoans in fresh water lakes. Limnol. Oceanogr. 27(2):246-253.

Beaver, J. R., T. L. Crisman and J. S. Bays. 1981. Examination of the relative impact of microzooplankton and macrozooplankton on bacteria in Florida lakes Verh. Internat. Verein. Limnol. 21:359-362.

Bogdan, K. G., J. J. Gilbert and P. L. Starkweather. 1980. In situ clearance rates of planktonic rotifers. Hydrobiologia 73(1-3):73-77.

Brezonik, P. L. 1976. Trophic state indices: rationale, progress, prospects. Report to the Fla. Dept. of Environmental Regulation.

Carlson, R. E. 1977. A trophic state index for lakes. Limnol. Oceanogr. 22:361-369.

Crisman, T. L. 1981. Algal control through trophic-level interaction. In Proceedings of Workshop on Algal Management and Control. Tech. Rept. E-81-7, U.S. EPA/U.S. Army Engineers Waterways Expt. Station. 313 pp.

Drenner, R. W., F. deNoyelles, Jr., and D. Kettle. 1982. Selective impact of filter-feeding gizzard shad on zooplankton community structure. Limnol. Oceanogr. 27(5):965-968.

Florida Game and Freshwater Fish Commission. 1972. Annual Progress Report for Investigations Project 1970-71. 156 pp.

Florida Game and Freshwater Fish Commission. 1975. Fish Management Report, South Florida Region.

Florida Game and Freshwater Fish Commission. 1976b. Fish Management Report, Central Florida Region. 133 pp.

Haney, J. F. 1973. An in-situ examination of the grazing activities of natural zooplankton communities. Arch. Hydrobiol. 72:87-132.

Helwig, J. T. and K. A. Council. 1979. SAS User's Guide, 1979 ed. SAS Institute, Inc., Cary, N.C.

Huber, W. C., P. L. Brezonik, J. P. Heaney, R. E. Dickinson, S. D. Preston, D. S. Dwornik and M. A. DeMaio. 1982. A classification of Florida lakes. Fla. Dept. of Environmental Regulation. Report No. ENV-05-82-1.

Kautz, R. S. 1980. Effects of eutrophication on the fish communities of Florida lakes. Proc. Ann. Conf. S.E. Assoc. Fish & Wildlife Agencies 34:67-80.

Larkin, P. A. and T. L. Northcote. 1969. Fish as indices of eutrophication. In G. A. Rohlich (ed.), Eutrophication: Causes, Consequences, Correctives. NAS, Washington.

McCauley, E. and J. Kalff. 1981. Empirical relationships between phytoplankton and zooplankton biomass in lakes. Can. J. Fish. Aquat. Sci. 38:458-463.

Peters, R. H. and F. J. Rigler. 1973. Phosphorus release by Daphnia. Limnol. Oceanogr. 18:821-839.

Peterson, B. J., J. E. Hobbie and J. F. Haney. 1978. Daphnia grazing on natural bacteria. Limnol. Oceanogr. 23(5):1039-1044.

Porter, K. G. 1977. The plant-animal interface in freshwater ecosystems. Amer. Scient. 65(2):159-170.

Ryder, R. A. 1982. The morphoedaphic index - use, abuse and fundamental concepts. Trans. Am. Fish. Soc. 111:154-164.

Shannon, E. E., and P. L. Brezonik. 1972. Relationships between lake trophic state and N and P loading rates. Environ. Sci. Technol. 6:719-725.

Shapiro, J. 1979. The need for more biology in lake restoration. In Lake Restoration. pp. 161-168; EPA 440/5-79-001.

Watson, S. and J. Kalff. 1981. Relationships between nannoplankton and lake trophic state. Can. J. Fish. Aquat. Sci. 38:960-967.

Wetzel, R. 1975. Limnology. Saunders, N. Y.

TECHNIQUES FOR THE CREATION AND MAINTENANCE OF INTERTIDAL SALTMARSH WETLANDS FOR LANDSCAPING AND SHORELINE PROTECTION

STEVE BEEMAN

Ecoshores, Incorporated
3881 South Nova Road
Port Orange, Florida 32019

ABSTRACT

Techniques are outlined for the creation and maintenance of landscaped intertidal shorelines. Construction of successful and aesthetically positive salt marsh shoreline systems requires that considerable attention be given to the nature of the substrate, slope of the shoreline, elevation relative to mean high - mean low water, types and sizes of transplant material and slope protection on the upper and lower edges of the tidal plane.

It is recommended that a relatively flat slope (5^{o}- 15^{o}) be constructed covering the mean high water - mean low water range with somewhat steeper slopes (20^{o} - 25^{o}) extending landward and waterward. Densely packed substrate (clay, hard pan, etc.) is not considered to be a suitable planting medium. Sand and sandy mud substrates are usually adequate with little or no alteration. The merits of armoring the lower range of the tidal zone and planting the entire intertidal range versus planting higher in the tidal zone to allow eventual colonization of the waterward portion of the range are discussed. Likewise, the utilization of transplants anchored in substrate with well established roots is compared to the use of bare root sprigs with regard to initial success, grow-in time and long-term community survival.

Erosion protection along the upland/wetland interface in the form of a vegetated or armored berm is recommended, where possible. Species composition is also discussed relative to combining visual aesthetics with proper zonation within the intertidal zone.

Maintenance of the system is addressed with respect to the control of counter-productive weeds and destructive human utilization of these intertidal areas. Positive aspects of seasonal pruning following annual seed production are considered.

INTRODUCTION

Estuarine wetland ecosystems are among the most productive in the world (Odum, 1961). They export organic nutrients to their adjoining water bodies and act as nutrient sinks for upland organic runoff (Woodhouse, Seneca and Broome, 1976). De la Cruz (1980) studied the role of estuarine marshes in the exportation of "fattened" animal tissues to the marine and terrestrial ecosystems, while Turner (1977) established a positive relationship between the size of estuarine marshes and the yield of their dependent fisheries.

Considering the evidence documenting the benefits of healthy estuarine marshes, it would be prudent to encourage the utilization of native vegetative shoreline systems for stabilization of riverfront properties. Fortunately, the state and federal permitting agencies have recently streamlined the permit process for waterfront owners who are willing to stabilize their shorelines with intertidal plants. The resultant decrease in red tape combined with substantial cost savings, when compared to conventional bulkhead or revetment construction, favors natural vegetated erosion control systems.

Unfortunately, not everyone considers a diverse salt marsh to be aesthetically appealing. In an effort to encourage the use of vegetative systems, we have combined the construction of productive wetlands with some basic but functional landscaping techniques.

AREA DESCRIPTION

Several sites, between Duval and Palm Beach Counties on the Florida Atlantic coast, large and small have been included in our observations over the past 5 years. All were estuarine shoreline areas adjacent to public parks or residential developments which lacked substantial intertidal vegetative cover prior to the installation of a planted shoreline system. Many were newly created intertidal areas and some were replacements, where high energy erosive forces had denuded the shoreline.

METHODS AND MATERIALS

In the design, construction and revegetation of intertidal wetland shoreline systems, several factors must be considered. Soil composition, elevation and slope of the shoreline, can determine whether a planting program will be successful. The amount of time required for the new plantings to grow together and provide effective erosion control is directly related to the size of the plants, planting density and time of year of installation. Protecting the upper slope of the vegetated area from upland runoff and the lower slope from wave energy, may help to establish a defined intertidal zone and to maintain vegetation throughout the tidal range. It is important to revegetate newly created shorelines with a species zonation that resembles nearby natural areas. Finally some sort of management program may be necessary to insure consistently dense growth from year to year.

Several studies have addressed one or more of these factors and a few are summarized below for the reader's information.

Soil

Christian et al. (1983) examined and compared growth forms of *Spartina alterniflora* in sand and in silt-clay soil. They reported that Spartina growing in silt-clay soil was more robust but had fewer plants per unit area than in sand. The silt-clay soil has smaller interstitial spaces than sand resulting in more restricted water circulation, higher concentrations of organic matter and is generally anaerobic. Stalter (1976) was able to correlate soil salt concentration within the intertidal region to the zonation of naturally occuring tidal marshes.

Woodhouse et al. (1976) reported that *Spartina alterniflora* roots can function effectively under anaerobic conditions where ammonium is the dominant form of nitrogen, but that aeration of the root zone and the introduction of nitrate nitrogen were detrimental to root function.

Slope/Elevation

Aside from salinity, the periodicity of inundation is the most important factor in determining the species composition of estuarine marshes (Salter, 1976). Woodhouse et al. (1976) determined that the range for *Spartina alterniflora* varies according to tidal range; between mean high water and mean low water in areas with little or no tidal fluctuation, and between mean high water and mean sea level on shorelines with substantial tidal influences.

Size/Density of Transplant Material

Banner (1977) recommended the use of plugs of organic material with mature plants when transplanting *Spartina alterniflora*. Woodhouse et al. (1976) suggested that single *Spartina* plants with intact root systems make suitable transplants. They also had greater success with field grown *Spartina* than with nursery stock. They reported that spacing transplants 0.91 meters (3 feet) apart in protected areas resulted in dense growth within one year, while 0.46 meters (1.5 feet) to 0.61 meters (2 feet) spacing was required in higher energy areas to provide effective shoreline stabilization.

Eleuterius and Gill (1981) and Eleuterius and Caldwell (1981) suggested planting plugs, sprigs or seeds of *Spartina* in the upper one-third of the tidal range only. They reported that the lower elevations are subsequently colonized by the high range plants but that transplants installed in the lower two thirds of the intertidal zone die back initially, followed by the same recolonization pattern.

It is questionable whether mangrove seedlings should be planted in conjunction with marsh grasses, since *Spartina alterniflora* seems to act as a trap for propagules and seeds, allowing volunteer colonization by the locally dominant mangrove species (Banner, 1977:Lewis and Dunstan, 1975).

Slope Armament

Eleuterius and Caldwell (1981) reported that the utilization of a hay bale barrier along the lower limits of the tidal range allowed *Spartina alterniflora* to colonize the entire zone.

Species Zonation

De la Cruz (1981) in a comparison between South Atlantic and Gulf coast tidal marshes, noted that classic zonation is prevalent along the Atlantic coast while it is absent in Gulf marshes. He attributed these differences to soil salinities within the marshes (10 - 30 $^{o}/oo$ in Atlantic wetlands, 2-15$^{o}/oo$ in the Gulf of Mexico marshes), and differences in the tidal fluctuation patterns.

Management Programs

Working in Mississippi, with *Spartina cynosuroides* and *Juncus roemerianus*, de la Cruz and Hackney (1983) observed that fire and/or harvest increased primary production values during the following year for both species, but that only the *Spartina* grew back to its original density at the end of one year. The *Juncus* required three years to re-establish pre-disturbance densities.

Reidenbaugh (1983) observed that the upper regions of the low marsh (Spartina alterniflora) are periodically devegetated by tidal wrack consisting primarily of dead Spartina stems. He noted that Spartina culms die back in the fall and winter following seed production in August and September. He also reported that tillers (shoots) arising early in the growing season elongate into culms within that year, while tillers emerging later in the season overwinter and become culms during the second year.

RESULTS AND DISCUSSION

During the past five years we have enjoyed particular success in the establishment of intertidal wetlands when certain procedures were followed.

We have been most successful on shorelines where we constructed an energy barrier (usually coquina boulders) at the waterward edge of a flat slope (5^o - 15^o) between mean high water and mean low water, gradually graded the upper tidal zone (20^o - 25^o) between mean high water and upland elevations, and constructed a berm along the upper edge of the slope. Since we have worked primarily with Atlantic coastal marshes, the classical zonation pattern of Spartina alterniflora, S. patens, Distichilis spicata and Sporobolus virginicus (moving from the water landward) has worked very well. The utiliization of large plugs of each species helps to establish viable communities, especially when planting in sandy areas. Since our primary goal is usually erosion control, most of our systems are planted at a density of 1 plant / $0.12m^2$ (1 plant/ ft^2). This usually provides for coalescence and functional stabilization within two to three months.

The breakwater at the toe of the slope provides protection from wave energy during establishment and protection from flotsam coverage during subsequent years. The berm along the upper edge of the slope prevents runoff erosion from upland sources. We have observed situations where wetland creation projects failed because of high wave energy during low tide levels undermining the vegetated shelf, coverage and shading by weeds, tidal wrack and flotsam, and siltation produced by unrestricted upland runoff.

We have observed that mangrove seedlings planted in conjunction with marsh systems are usually only 50 - 75% successful and that volunteer seedlings frequently outnumber planted seedlings within two to three years if there is a nearby source for mangrove seeds.

It has become apparent over the years, that some sort of management program is beneficial to the maintenance of healthy, dense shoreline vegetation systems. The primary culprit in the disappearance of intertidal plants appears to be shading. Shading can result from flotsam inundation, die-back of the previous year's stems and seed stalks or invasion of creeping high transitional zone plants such as cow pea vines (Vigna luteola). Shorelines with breakwaters at the toe experience substantially fewer flotsam problems that unprotected shorelines. Periodic removal of nuisance species should control the problem of noxious weed invasion. Seasonal pruning of seed heads, stalks and stems is also recommended for each species, following its respective seed production cycle. The export of trimmings to the adjacent estuary, in the form of organic debris, serves as important a function as the export of organic detritus (Reidenbaugh,1983). More importantly, the removal of the dead tops allows sunlight penetration to the emerging tillers. The marshes in which we have observed seasonal pruning, over the past ten years, exhibit consistently dense growth patterns. However, nearby marshes that were not trimmed have exhibited a three to four year cycle of successional growth patterns; from sparse to medium to dense and returning to sparse, following a particularly heavy seed production the previous year.

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LITERATURE CITED

Banner, Arnold. Revegetation and Maruration of a Restored Shoreline in the Indian River, Florida. Lewis, R.R. and D.P. Cole, eds. Proceedings of the Fourth Annual Conference on Restoration of Coastal Vegetation in Florida, Hillsborough Community College, Tampa, Florida: 13-43;1977.

Carlton, J.M. A Guide to Common Florida Salt Marsh and Mangrove Vegetation. Florida Research Publ. No. 6:30pp;1975.

Christian, R.R., J.A. Hansen, R.E. Hodson and W.J. Wiebe. Relationships of Soil, Plant, and Microbial Characteristics in Silt-clay and Sand, Tall-Form Spartina alterniflora Marshes. Estuaries 6(1):43-49; 1983.

Cruz, A.A. de la. Recent Advances in Our Understanding of Salt Marsh Ecology, Fore, P.L. and R.D. Peterson, eds. Proceedings of the Gulf of Mexico Coastal Ecosystems Workshop. U.S.F.W.S./O.B.X.-80/30:51-65; 1980.

Cruz, A.A. de la. Differences Between South Atlantic and Gulf Coast Marshes. Carey, R.C., P.S. Markovits and J.B. Kirkwood, eds. Proceedings of the U.S. Fish and Wildlife Service Workshop on Coastal Ecosystems of the Southeastern United States. U.S.F.W.S./O.B.S._ 80/59:10-20;1981.

Cruz, A.A. de la and C.T. Hackney. The Effects of Winter Fire and Harvest on the Vegetational Structure and Primary Productivity of Two Tidal Marsh Communities in Mississippi. Mississippi-Alabama Sea Grant Consortium M.A.S.G.P. 4:19;1983.

Eleuteruis, L.N. and J.I. Gill Jr. Long Term Observations on Seagrass and Salt Marsh Established from Transplants. Stovall, R.H., editor. Proceedings of the Eighth Annual Conference on Wetlands Restoration and Creation. Hillsborough Community College, Tampa, Florida: 74-86; 1981.

Eleuterius, L.N. and J.D. Caldwell. Colonizing Patterns of Tidal Marsh Plants and Vegetational Seccession on Dredge Spoil in Mississippi. Stoval, R.H. Editor. Proceedings of the Eighth Annual Conference on Wetlands Restoration and Creation. Hillsborough Community College, Tampa, Florida:58-73;1981.

Lewis, R.R. and F.M. Dunstan. The Possible Role of Spartina alterniflora Loisel in Establishment of Mangroves in Florida. In Lewis, RR. editor. Proceedings of the Second Annual Conference on Restoration of Coastal Vegetation in Florida:82-100;1975.

Odum, E.P. The Role of Tidal Marshes in Estuarine Production. N.Y. State Conservationist 15:12-15;1961.

Reidengaugh, T.G. Productivity of Cordgrass, Spartina alterniflora, Estimated from Live standing crops, Mortality and Leaf Shedding in a Virginia Salt Marsh. Estuaries 6(1):57-65;1983.

Stalter, R. The Zonation of Vegetation of Southeastern sal marshes. Lewis, R.R. and D.P. Cole, eds. Proceedings of the Third Annual Conference on Restoration of Coastal Vegetation in Florida. Hillsborough Community College, Tampa, Florida:24-35;1976.

Turner, R.E. Intertidal Vegetation and Commercial Yields of Penaeid Shrimp. Trans. Amer. Fish. Soc. 106(5):411-416;1977.

Woodhouse, W.W. Jr., E.D. Seneca and S.W. Broome. Propagation and Use of Spartina alterniflora for Shoreline Erosion Abatement. U.S. Corps of Engineers, Technical Report 76(2):72pp;1976.

CONSIDERATIONS FOR THE FUNCTIONAL RESTORATION OF IMPOUNDED WETLANDS

Beverly F. Birkitt
Florida Department of Environmental Regulation
2600 Blair Stone Road
Tallahassee, Florida 32301

ABSTRACT

Almost 40,000 acres of mangrove and marsh areas along the east coast of Florida have been impounded for the purpose of mosquito control since the early 1950's. Recently, there has been an increase in the number of developments proposed in or adjacent to these areas. Review of the proposed developments through the state and federal regulatory process has resulted in renewed interest in the rehabilitation of these impounded wetlands. Significant changes in wetland vegetation have resulted from impounding for the purpose of mosquito control. Also, there is some evidence that the cumulative effect of impounding large areas of coastal wetlands might be detrimental to fisheries and other estuarine resources which depend on the marsh/estuary system.

Management plans are being developed for these impounded areas to restore some of the original functions of the wetlands to benefit fisheries and other estuarine resources as well as to maintain mosquito control. Considerations for the development of management plans include water level regulation, connection of the marsh to the estuary, and improvement in water circulation. Information on two sites for which functional restoration will be attempted in the near future is included.

INTRODUCTION

There are almost 40,000 acres of coastal wetlands along the east coast of Florida that have been impounded for the purpose of mosquito control. Impoundments are located in Volusia, Brevard, Indian River, St. Lucie and Martin Counties. These impounded areas have been isolated from the adjacent estuarine systems in many cases for 10 years or more. The construction of impoundments began as early as the 1950's and continued until the early 1970's (Provost, 1977). Recently because of pressures to place condominium developments in impounded wetlands on Hutchinson Island, the condition of these impoundments has been brought to the attention of state and federal regulatory agencies.

Efforts are now underway to restore these impounded wetlands to a functioning part of the marsh/estuary system. Because restoration plans must consider the control of mosquitos in these impoundments, the use of water level management techniques is considered to be the best approach to the functional restoration of wetlands impounded for mosquito control. A Subcommittee of the Governor's Working Group on Mosquito Control has been established to provide technical assistance in the development of management plans which will provide both mosquito control and environmental benefits.

EFFECTS OF IMPOUNDING

The impoundment of coastal wetlands has caused significant changes in vegetation (Harrington & Harrington, 1961, 1982; Neilson, 1961; Clements & Rogers, 1964; Provost, 1967, 1977; Ehrhardt & Herting, 1971; Bidlingmayer & McCoy, 1978; Heald, 1982; Leenhouts, personal communication). Many herbaceous marshes in Brevard County have been transformed into open water bodies by impounding. Pre-impoundment vegetation in Indian River, St. Lucie County and parts of Brevard County consisted of *Batis maritima*, *Salicornia* spp. and black mangroves (*Avicennia germinans*). The impounding and subsequent flooding of the wetlands to prevent oviposition by salt marsh mosquitos killed the black mangroves and low-growing vegetation. Dead black mangroves are still apparent in many impoundments in this area (Figure 1). Although some impoundments are presently devoid of vegetation, other impoundments, most notably in St. Lucie County, have been secondarily revegetated by red mangroves (Figure 2). Submerged vegetation such as *Ruppia maritima* (widgeon grass), *Najas marina* (marine naiid) and *Chara* sp. has become established in some impoundments that remain flooded. In impoundments where salinity levels are low, freshwater species often occur. While the significance of the loss of marsh vegetation is apparent, changes in the type of wetland vegetation may also effect the estuarine system because of changes in productivity or faunal associations.

Isolation of these extensive areas of coastal wetlands has also affected the utilization of the marshes as nursery, feeding and refuge areas by fish and other estuarine organisms (Harrington & Harrington, 1961, 1982; Weaver & Holloway, 1974; Herke, 1979; Gilmore et al., 1982, Lewis et al., IN PRESS). Impounding has effectively blocked the export of forage organisms, detrital material and other nutrients from the marsh. Comparisons of fish collections taken either before and after impoundment or from a closed

Figure 1. Dead black mangroves at the Moorings Development site (Imp. No. 10) in Indian River County.

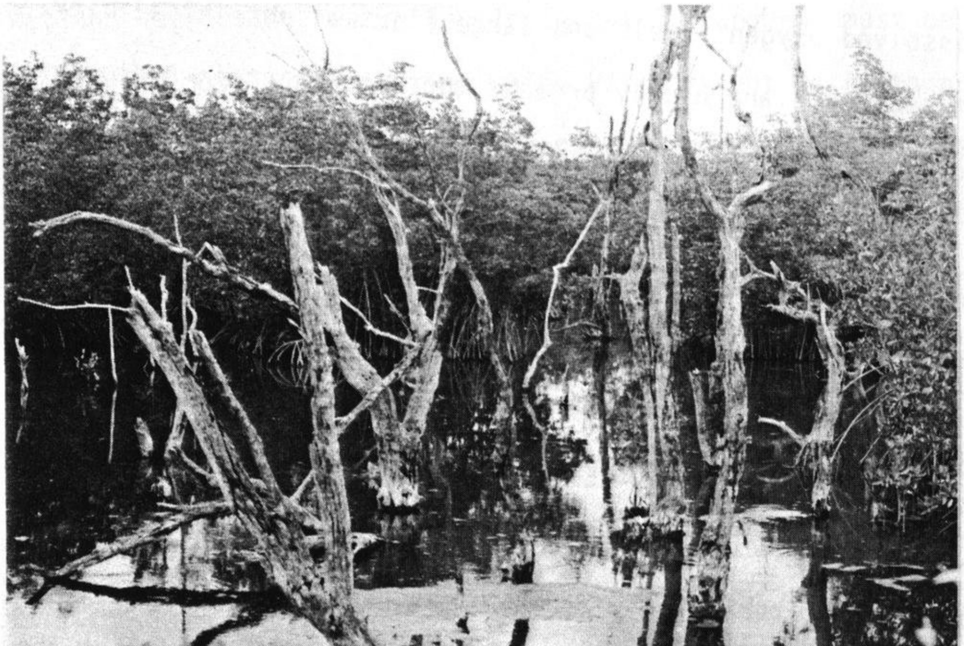

Figure 2. The Campeau/Dardashti site, an impounded area which was previously a black mangrove/<u>Batis</u>/<u>Salicornia</u> marsh currently revegetated by red mangroves. (Imp. Nos. 2, 3, and 4, St. Lucie County).

marsh which will benefit fisheries and other biological resources. impoundment and an adjacent open impoundment indicate the exclusion of many migratory estuarine fish from closed impoundments (Gilmore et al., 1982). Species collected by Harrington and Harrington (1961) from a natural marsh prior to impoundment included snook, tarpon, Irish pompano, white and striped mullet, yellowfin mojarra and ladyfish. Gilmore (1982) collected many of these same species in an adjacent reopened impoundment. Most of the fish collected by Harrington and Harrington (1982) and Gilmore et al., (1982) in the closed impoundment were species such as killifish, mollies and mosquito fish that typically occur in isolated bodies of water and can withstand a great deal of stress.

Water quality in the impounded marshes is generally poor with low dissolved oxygen levels and large fluctuations in salinity. Bottom sediments are highly organic and often contain hydrogen sulfide. Indicative of a change in water quality, phytoplankton and dinoflagellate blooms are common in the impoundments but not in nearby estuarine areas. The maintenance of water quality in the marshes is considered important to the productivity of the system.

PROPOSED RESTORATION

Although numerous adverse effects on the natural marsh/estuary system can be attributed to the impoundment of these areas for mosquito control, Provost, as early as 1967, suggested that the environmental effects of impounding could be reduced by prudent management techniques. Provost's (1967, 1970) ideas for maintaining wetland vegetation in mosquito control impoundments have been expanded to include possible ways of restoring other aspects of the marsh which will benefit fisheries and other biological resources.

The primary restoration objectives of the management plans being developed are:

1. The maintenance or re-establishement of wetland vegetation.
2. The restoration of biotic exchange between the estuary and the marsh.
3. The improvement of water quality.

Considerations in the design of management plans for restoration of impounded wetlands must take into account many factors including water level regulation, connection of the marsh to the estuary and water circulation. To maintain or allow the re-establishment of wetland vegetation, water levels in the impoundment must be set at the lowest possible elevation which will prevent mosquito production. Although only a few inches of water are needed over the marsh to prevent oviposition by mosquitos, water levels must often be maintained at higher elevations to offset evapotranspiration and to allow for the topography of the marsh. The growth of black mangroves and low-growing vegetation in impoundments can be maintained or re-established by increasing water levels gradually and by carefully regulating flood elevations and duration (Provost, 1970). Impoundment No. 23 in Indian River County which was previously flooded has been revegetated by *Batis maritima* and *Salicornia* sp. (Figure 3). Gumbo Limbo Island in Brevard County is being managed effectively to maintain the black mangrove/*Batis*/*Salicornia* vegetation while controlling mosquito production.

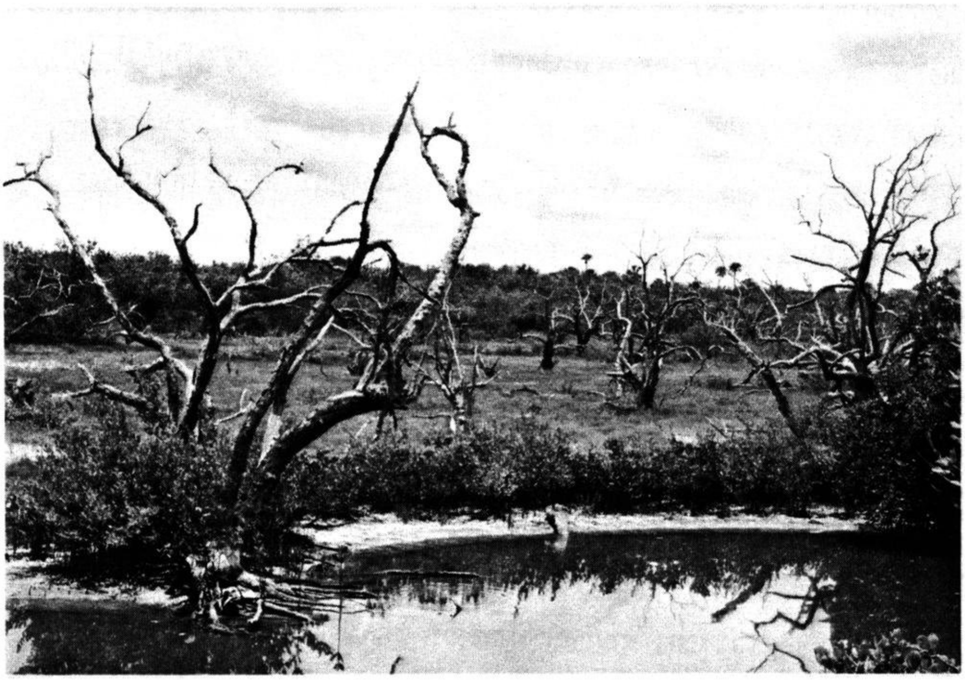

Figure 3. Impoundment No. 23 in Indian River County revegetated by Batis maritima, Salicornia sp. and red and black mangroves.

To restore the biotic exchange between the marsh and the river, impoundments should be opened to natural tidal fluctuations. Because impoundments must be closed and flooded during the mosquito breeding season, seasonal reconnections are considered the best possible alternative. However, timing the opening and closing of an impoundment still presents a problem. Recent work by Gilmore et al., (1982) showed that migratory estuarine fishes including many species important to sport and commercial fisheries will enter an open impoundment during the mosquito breeding season. A decline in the populations of some fish species including snook (Centropomus spp.) has been documented as related to habitat modification resulting from impounding wetland areas (Irby, personal communication). In order to minimize the exclusion of migratory estuarine fish and maintain the marsh habitat, the impoundments should be closed the shortest period of time necessary to provide adequate mosquito control.

The export of nutrients and forage organisms is expected to be increased by the seasonal opening of impoundments and the occasional overflooding and release of water from the impoundments during the closed period. Culverts of sufficient size placed in locations which will allow near-natural tidal exchange during the open period should be used to connect the impoundments to the estuary. Water level control structures placed on the culverts should be designed to interfere with the movement of organisms as little as possible.

Water quality in the impoundments is expected to improve with the increased tidal exchange between the marsh and the estuary and with improved water circulation in the impoundment. The location of pumps, culverts and the placement of structures for routing water flow are important considerations in improving water quality within the impoundment. Additionally, to prevent fish kills from oxygen depletion caused by the release of highly organic water or the resuspension of organic sediments in the river during discharge from the impoundments, water from areas that have been impounded for long periods of time should be released gradually.

Campeau/Dardashti Site

One of the first areas in which restoration will be attempted is the Campeau/Dardashti site which is located on Hutchinson Island in St. Lucie County. This area consists of three interconnected impoundments, Impoundment No. 2, 188 acres, Impoundment No. 3, 168 acres and Impoundment No. 4, 192 acres (Bidlingmayer & McCoy, 1978). The impoundments are bisected by U.S. A1A from north to south. This marsh was originally ditched and then diked and flooded around 1960 (Evans, personal communication). Within the last several years, these impoundments have remained flooded to depths between 1.0 and 2.0 ft. MSL throughout the year and have been isolated from the Indian River (Evans, personal communication). Impoundments Nos. 2, 3 and 4 are vegetated almost entirely by mature red mangroves. Dead black mangroves are still visible among the red mangroves (Figure 2).

Culverts will be installed through the outside dike and between impoundments. Flap-gate riser control structures will be placed at the inside end of the culverts to allow greater flexibility in the management of water levels. A plug will be constructed in the existing perimeter ditch to force improved water circulation throughout the impoundment. Additionally, pumps capable of pumping 6000 gpm will be located at two or three sites in the impoundments on the western side of U.S. A1A (Figure 4). In late spring, at the beginning of the mosquito breeding season, the impoundments will be closed off from the river and pumped with water from the river to the desired flood elevation. Water levels will be maintained at the flood elevation throughout the mosquito breeding season (roughly May-September). The risers will then be removed from the culverts and the marsh will be allowed to exchange freely with the river until the spring mosquito breeding season begins at which time the risers will be reinstalled.

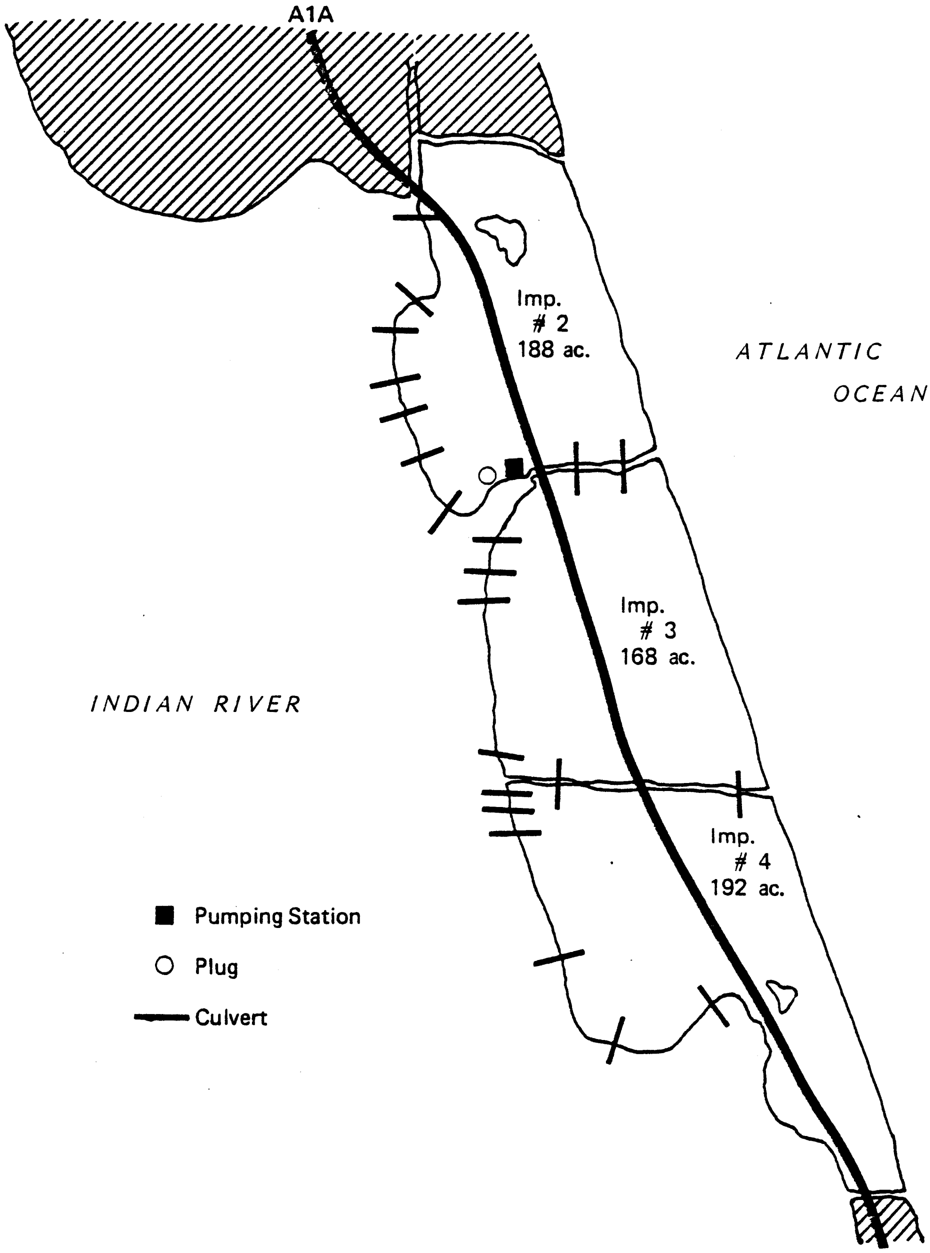

FIGURE 4

IMPOUNDMENTS # 2, 3 &4

ST. LUCIE COUNTY

STRUCTURAL CHANGES FOR MANAGEMENT PLAN IMPLEMENTATION

The restoration plan proposed will not attempt to restore the original vegetation or original conditions of the natural marsh, but will provide increased tidal exchange which should restore the use of the area as a nursery, feeding and refuge area for fish and other estuarine organisms and increase the export of detrital material and other nutrients. Limited monitoring of water quality, water level fluctuations and mosquito production is proposed at the Campeau/-Dardashti site. An attempt is being made to find funding to determine and evaluate changes in fish utilization in these impoundments.

Moorings Site

Another area for which restoration is currently proposed is Impoundment No. 10 at the Moorings Development site in Indian River County (Bidlingmayer and McCoy, 1978). The impoundment is 42 acres in size and is divided into two cells. Before being impounded in 1965 (Bidlingmayer and McCoy, 1978), this area was probably a black mangrove/*Batis*/*Salicornia* marsh.

Today, this impoundment is almost entirely barren. Dead black mangroves are apparent throughout the area (Figure 1). Due to the scarcity of vegetation currently on site and the apparent lack of a nearby seed source, it may be desirable to re-plant vegetation in this impoundment. Funding through an enforcement settlement is available for research at this location. Background data will be collected prior to reopening the impoundment, and a monitoring program will be initiated to document changes in water quality, and vegetation as well as fish utilization resulting from implementation of a management plan.

CONCLUSION

There are many adverse effects from impounding wetland areas for mosquito control; some are very obvious to the casual observer such as the loss of vegetation; others however, are more subtle and perhaps more significant in the long-term such as the exclusion of migratory estuarine fish. These effects can be reduced by prudent water management techniques. The most promising management technique which will restore some of the ecological benefits of the marsh and maintain the required mosquito control appears to be the seasonal reconnection of the impounded marsh to the estuary. Because management decisions must be based on subjective judgement, at least initially, management plans should be set up with flexability to allow for modifications based on experience gained in the operation of these and other impoundments. Monitoring changes in vegetation, water quality and fish utilization will determine whether or not the restoration objectives are achieved.

Restoration by the management techniques described is being planned for several impounded marshes. However, the restoration proposals currently planned include less than 600 acres out of the 20,000 acres of impounded wetlands that could be reopened to adjacent estuaries. An attempt is being made to locate additional money for monitoring restored impoundments and for the development and implementation of management plans for additional impoundments.

Although the seasonal opening and closing of the impoundments is decidedly a compromise between the objectives of 1) mosquito control and 2) maintenance of ecological benefits from the marshes, the management plans being proposed are designed to take into consideration both objectives. In the past little or no consideration was given to the benefits of the natural marsh/estuary system. The connection of these impoundments to the adjacent estuaries is expected to benefit commercial and sport fisheries species and to enhance water quality and productivity in the estuaries through habitat improvement and increased nutrient cycling and export of detrital material and forage organisms.

ACKNOWLEDGEMENTS

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LITERATURE CITED

Bidlingmayer, W. L. and E.D. McCoy. An inventory of the saltmarsh mosquito control impoundments in Florida. Unpublished Report to Fish & Wildlife Service, U.S. Department of Interior. 103 pg., 173 maps, appendices I-III. 1978.

Clements, B.W. Jr. and A.J. Rogers. Studies of impounding for the control of salt marsh mosquitos in Florida, 1958-1963. Mosquito News 24(3): 265-276. 1964.

Ehrhardt, W.R. and G. Herting. Tomoka Marsh Survey. Annual Report, East Volusia Mosquito Control District. pp.1- 43. 1971.

Gilmore, R.G., D.W. Cooke, and C.J. Donohoe. Comparison of fish populations and habitat of open and closed salt marsh impoundments in east-central Florida. N.E. Gulf Sci. 5(2):25-37. 1982.

Harrington, R. W. and E. S. Harrington. Food selection among fishes invading a high subtropical salt marsh: from onset of flooding through the progress of a mosquito brood. Ecol. 42(4):646-666. 1961.

Harrington, R.W. and E.S. Harrington. Effects on fishes and their forage organisms of impounding a Florida salt marsh to prevent breeding by salt marsh mosquito. Bull. Mar. Sci. 32(2):523-531. 1982.

Heald, E.J. Preliminary assessment of proposed development project on the Hoyt Tract, Hutchinson Island, St. Lucie County, Florida. Unpublished report. Study of impoundments No. 3 & 4 in St. Lucie County. 1982.

Heald, E.J. Resource description of the Hoyt tract, Hutchinson Island, St. Lucie County Florida and assessment of the proposed Green Turtle Beach Development Project. Unpublished final report. 1982.

Herke, W.H. Some effects of semi-impoundment on coastal Louisiana fish and crustacean nursery usage. In: Third Coastal Marsh and Estuary Management Symposium. L.S.U., Baton Rouge, J.W. Day, D.D. Culley, R.E. Turner, and A.J. Mumphrey, Jr. (eds.), pp. 325-346. 1979.

Lewis, R.R., R.G. Gilmore, Jr., D.W. Crewz and W.E. Odum. The fishery resources of mangrove forest in Florida. IN PRESS.

Neilson, E.T. On the habits of the migratory butterfly, Asciamonuste L. Biol. Medd. Dan. Vid. Selsk. 23(11):1-79. 1961.

Provost, M.W. Managing impounded salt marsh for mosquito control on estuarine resource conservation. LSU Marsh and Estuary Symposium. pp. 163-171. 1967.

Provost, M.W. Water management plan for an island in Sections 16 and 21 of T-25-S, R-37-E to be impounded for mosquito control. pp. 1-11. 1970.

Provost, M.W. Source reduction in salt-marsh mosquito control: past and future. Mosquito News 37(4):684-698. 1977.

Weaver, J.E. & L.F. Holloway. Community structure of fishes and macro-custaceans in ponds of a Louisiana tidal marsh influenced by weirs. Contrib. Mar. Sci. 18:57-69. 1974.

ANCHORING STABILITY OF NEW SEAGRASS PLANTINGS

by

B.A. Christensen
Professor
Hydraulic Laboratory
Department of Civil Engineering
Gainesville, FL 32611

and

David Tackney
President
Tackney and Associates, Inc.
Naples, FL 32302-10464

ABSTRACT

Seagrass beds are among the most desirable systems in our shallow coastal waters. Others are mangroves, mud flats, coral reefs, etc. Revegetation of areas damaged by development and in some cases natural events such as hurricanes should therefore be quite common.

Runners from healthy and well established beds are transplanted to the damaged areas after the bottom elevation has been restored. A spacing of 0.5 to 1 m between the transplanted runners is usually used. Runners may be attached to the bed by use of steel staples or other anchoring devices, and the roots pushed into the sand. Areas revegetated in this way exhibit a lower resistance to flow than heavily vegetated areas under similar flow conditions. Therefore, new beds are exposed to higher velocities and bed shear stresses than the established areas, often resulting in failures caused by disruption of the bond between plant runners and sand bed or simply by a massive erosion of the sand bed itself.

The paper discusses these processes and develops methods for quantification of the flow impact on the bed. Testing of live transplanted seagrasses (Halodule) in the Hydraulic Laboratory's research flume is discussed and a special model law for transfer of flume results to prototype conditions is developed.

Model tests referring to a case near Cudjoe Key in the Florida Keys (Niles channel restoration project) are described.

INTRODUCTION

Newly transplanted seagrasses such as Halodule are exposed to higher erosive forces than well established beds due to their lower hydraulic roughness and thereby reduced resistance to flow. Failure may be caused by one of two phenomena, a general erosion of the sand around the transplants or flow induced motion of the transplanted runners resulting in disruption of the root/sand contact. A combination of the two is of course also possible. The general erosion of sand beds is considered first.

Erosion of sand beds

Due to its importance in littoral processes, the question of when incipient motion of sand takes place has been treated in the literature quite frequently during the last couple of centuries. Two schools of thought have evolved, the critical velocity approach and the tractive force approach. Both are used today. However, the tractive force approach does seem to be the most attractive.

The critical velocity approach is based on a *critical velocity* at which a granular cohesionless bed material will begin to move. A formula for this velocity was - according to Forchheimer (1914) - proposed by Brahms in 1753. It is postulated to be proportional to the dry weight of a singular grain to the one sixth power corresponding to Sternbergs (1875) later formula stating that the critical bed velocity is proportional to the square root of the grain-size. Many similar formulas have been presented since then. They are discussed and referenced by Forchheimer in his 1914 book and by Graf (1971). The location of the critical reference velocity is not always well defined in the contributions. Sometimes it is a surface velocity, other times the vertically averaged velocity, the cross-sectional mean velocity or the bed velocity as used by Brahms and Sternberg.

In the early part of this century Fortier and Scobey (1926) published an extensive study on "Permissive Canal Velocities" that is giving such velocities in tabular form. Fortier and Scobey refer to the cross-sectional velocity.

Although Fortier and Scobey's tables still are used today by some engineers and coastal morphologists it is Hjulström's (1935) diagram, Fig. 1, that is used by most of the critical velocity proponents. This diagram which is based on numerous observations of incipient erosion in Swedish rivers gives the critical cross-sectional mean velocity, v_m, as a function of the grain-size, d_e.

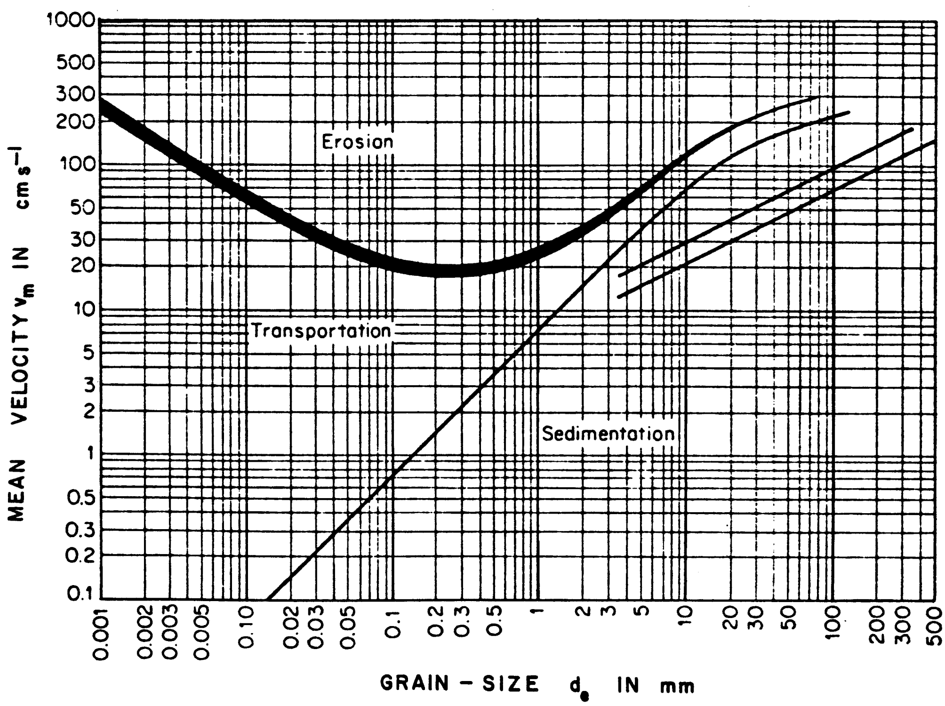

Figure 1. Hjulström's diagram for critical mean velocity as function of grain-size of bed material. Horizontal bed. Hjulström (1935)

The *critical tractive force* approach was initiated by du Buat (1786) (Forchheimer (1914)) at about the same time as Brahms did his pioneering work with critical velocities. Du Buat states that a bed material can withstand a certain bed shear stress, called the critical bed shear stress, beyond which erosion will take place. This critical shear stress is considered a property of the bed form and material while the bed shear stress it must be compared to of course is a flow property proportional to the local depth and slope of the energy grade line.

The best known results of critical tractive force research are those provided by Shields (1936) and given in his now famous diagram shown in Fig. 2. In Shield's diagram the entrainment function for a horizontal bed E_h is defined as indicated on the ordinate axis where γ_s = unit weight of the grain material, γ = unit weight of water, d_e = grain-size and $\tau_{cr.h}$ = the critical shear stress of a horizontal bed. The abscissa, the wall Reynolds number, is defined as the product of the friction velocity v_f and grain-size d_e divided by the kinematic viscosity ν of water. v_f is a measure of the bed shear stress and defined as the square-root of the ratio of bed shear stress to density of water.

The critical bed shear stress seems to be the most rational measure for a bed's ability to resist scour. However, a majority of the formulas for this quantity are deterministic. Since both the turbulent flow that causes scour and the composition of the bed material are of a stochastic nature it should be expected that the formulas for the critical bed shear stress must be stochastic rather than deterministic and relate the critical shear stress to a probabilistic risk of erosion.

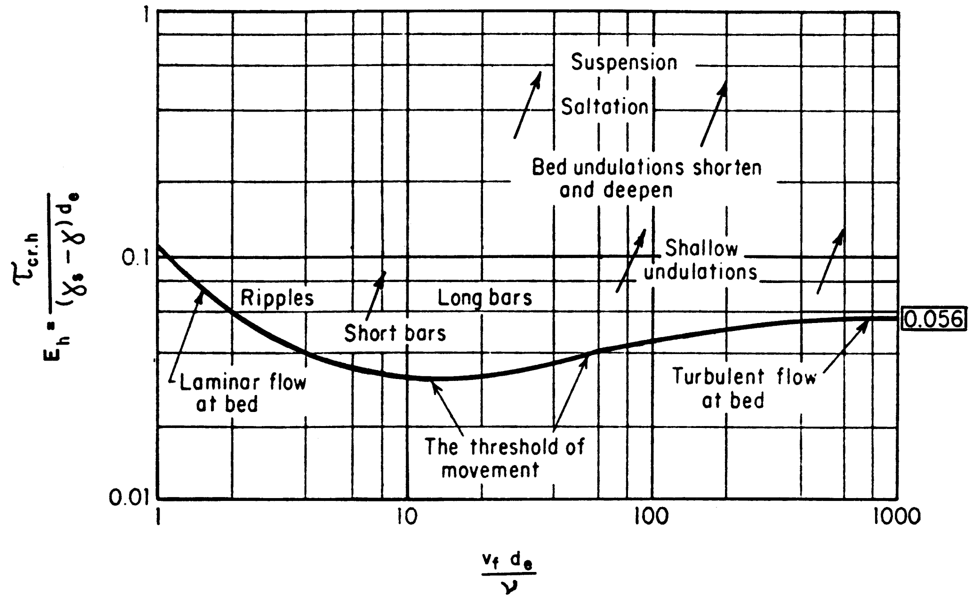

Figure 2. Shield's diagram for critical value of entrainment function as function of Wall Reynolds number. Horizontal bed. Shields (1936)

In the case of nonuniform deposits an effective grain-size, d_e, of the bed material may be used. This grain-size is defined as the particle size of a bed of a uniform material which will experience incipient motion at the same bed shear stress as the nonuniform material it represents. When the grain-size distribution of the nonuniform material is known the effective grain-size d_e may be evaluated. According to Christensen (1969) this may be done by the integration

$$d_e = \frac{1}{\int_0^1 \frac{df}{d_s}} \quad \ldots\ldots\ldots\ldots (1)$$

where f = fraction of weight of the bed material which is smaller than the size d_s.

CRITICAL SHEAR STRESS OF A SAND BED

Erosion of a sandy horizontal ocean bed will take place when the local bed shear stress exceeds the critical bed shear stress $\tau_{cr.h}$, i.e. when

$$\tau_{cr.h} = \gamma dS > \tau_{cr.h} = E_h (\gamma_s - \gamma) d_e \quad \ldots\ldots\ldots\ldots (2)$$

where d = the local water depth and S = the local slope of the energy grade line. As indicated for instance by Streeter and Wylie (1975) S may be related to the mean flow velocity v_m and depth by the classic Manning formula

$$S = \frac{v_m^2 n^2}{d^{4/3}} \quad \text{(S.I. units)} \quad \ldots\ldots\ldots\ldots (3)$$

where Mannings n may be found from the roughness k of the bed by

$$\frac{1}{n} = \frac{25.8}{k^{1/6}} \quad \text{(S.I. units)} \quad \ldots\ldots\ldots\ldots (4)$$

according to Christensen (1983a).

At values of the wall Reynolds number in excess of about 500, Figure 2 indicates that E_h is constant and equal to 0.056. Since erosion is caused by turbulent flow of a highly stochastic nature it does not seem likely that E_h should be the same constant for all cases. Dependency on the probability of erosion appears to be more realistic. This was shown by Christensen (1983b).

The stochastic formula for the entrainment function found in that study is plotted in Figure 3 where the indicates values of shape factors β_1, β_2 and the angle of repose ϕ are characteristic for sandy beds in Florida's coastal waters.

A *longitudinal* bed slope in the direction of flow does have an influence on E_h, but this influence is usually negligible in Florida's coastal waters.

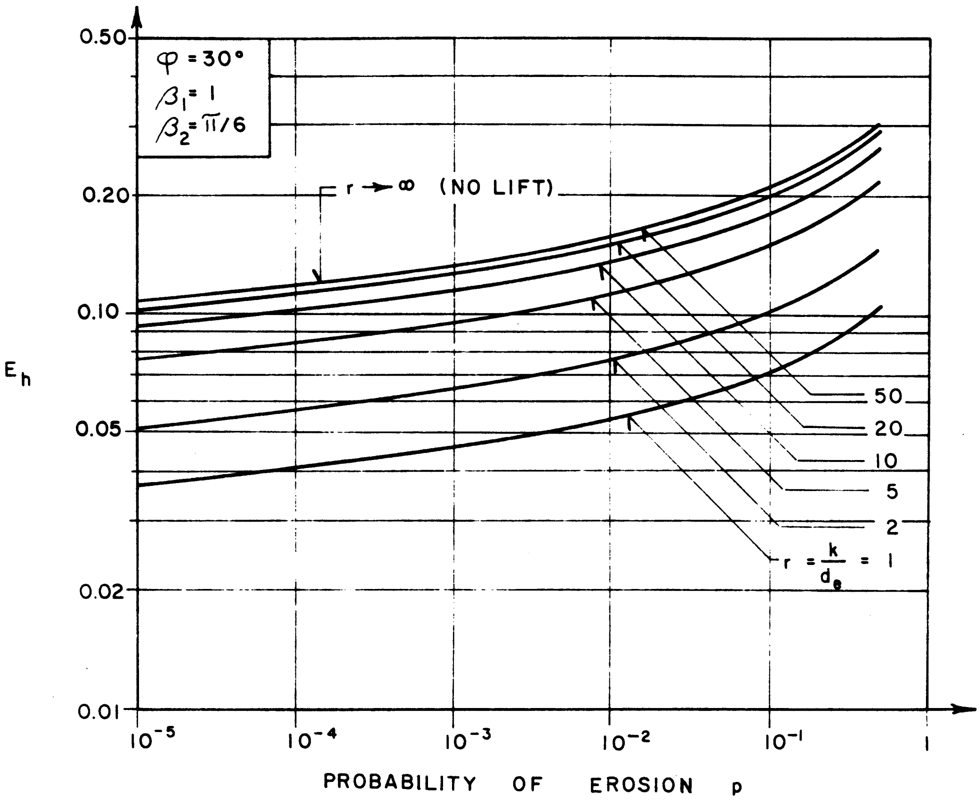

Figure 3. Critical value of entrainment function as function of probability of erosion p and roughness/grain-size ratio r for typical Florida coastal sediments. Horizontal bed. Christensen (1983b).

For a *transverse* slope i.e. a bank slope normal to the direction of flow a correction factor, E_b/E_h, may be developed. This factor is plotted for ϕ = 30° in Figure 4 as a function of the roughness/grain-size ratio r and the bank slope s here defined as the cotangent of the inclination of the embankment with horizontal (Christensen, 1972).

EROSION OF TRANSPLANTED RUNNERS

Individual runners and their anchoring devices such as steel staples are acted upon by hydrodynamic forces, i.e. drag and lift, caused by the flowing water, buoyant forces and gravity as indicated in the left part of Figure 5. While the hydrodynamic and buoyant forces are trying to break the bond between the runner and the bed the gravity force is stabilizing this bond.

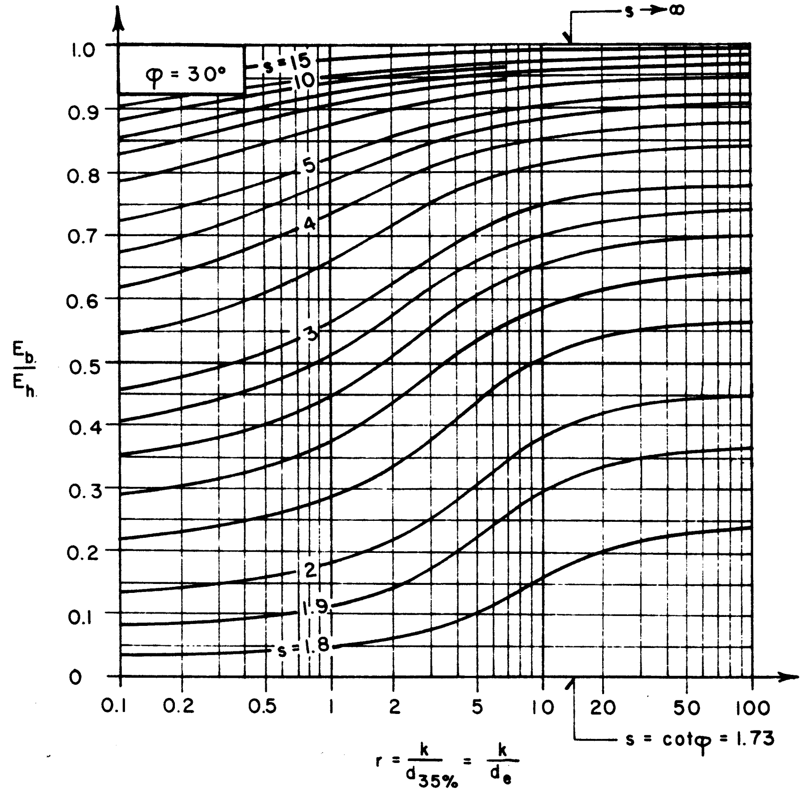

Figure 4 Correction factor for bank slope as function of roughness/grain-size ratio and bank slope for typical Florida coastal sediments. Christensen (1972).

The local mean velocity at which the bond between transplanted runners or rather their growing root systems and the sand bed is broken may be determined by model experiments in a hydraulic research flume in which live runners are planted in a sand with the same effective grain-size as the sand on the site where the vegetation is to be established.

Since plant material (and thereby sand) must have the same size in model and prototype and a natural depth of 1.5 m to 2.5 m usually cannot be established in research flumes, it is necessary to use a hydraulic model law that allows for reduction of the model depth. Such a model law must be based on the requirement that the velocity fields in model and prototype are identical near the runners so that all hydrodynamic forces that are induced by the flow fields are the same in the two systems. This is illustrated in the right-hand side of Figure 5

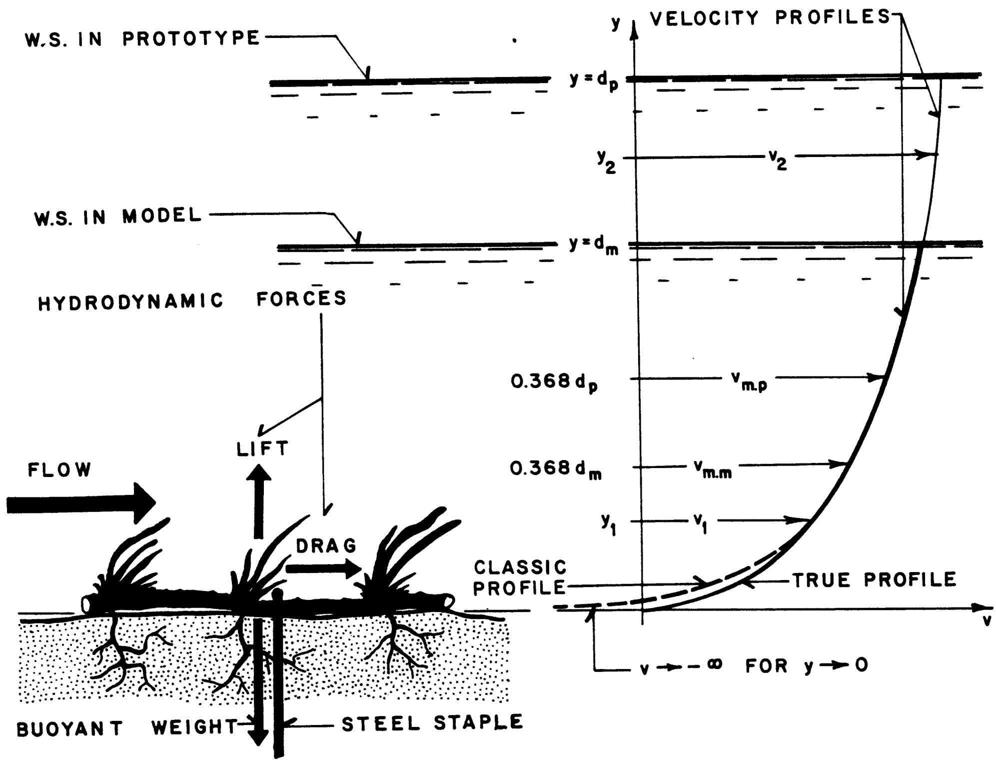

Figure 5 Velocity profiles and hydrodynamic forces acting on a newly transplanted seagrass runner in model (hydraulic flume) and in prototype (nature).

where the velocity profile below the model water surface corresponding to depth d_m is the same as the prototypes velocity profile.

The classic Prandtl-profile that usually is used to represent turbulent flow over coastal and riverine sand bed reads

$$\frac{v}{v_f} = 8.48 + 2.5 \ln \frac{y}{k} = 2.5 \ln \frac{29.7y}{k} \quad . \; . \; . \; . \; . \; . \; . \; . \; . \; . \; (5)$$

in which y = vertical distance from the bed to the point where the velocity is v. See for instance Streeter and Wylie (1975).

While equation (5) certainly represents the true velocity profile in most of the water column it fails near the bed where $v \to -\infty$ for $y \to 0$. Since the hydrodynamic forces acting on the runners depend on the velocities in the proximity of the bed it is necessary to use a modified version of equation (5). Such a profile was proposed by Christensen (1972). It may be written

$$\frac{v}{v_f} = 8.48 + 2.5 \ln \left(\frac{y}{k} + 0.338\right) = 2.5 \ln \left(\frac{29.7y}{k} + 1\right) \quad \ldots\ldots\ldots (6)$$

It represents the true velocity profile near the bed.

Recalling that the mean velocity in a vertical may be measured at a distance from the bed equal to 0.368 times the depth, applying equation (6) in model and prototype, and realizing that v_f must have the same value in model and prototype lead to the ratio

$$\frac{v_{m.p}}{v_{m.m}} = \frac{\ln\left(\frac{10.9d_p}{k} + 1\right)}{\ln\left(\frac{10.9d_m}{k} + 1\right)} \quad \ldots\ldots\ldots (7)$$

in which $v_{m.p}$ and $v_{m.m}$ are the mean velocities in prototype and model verticals, respectively, d_p = prototype depth, d_m = model depth and of course k = roughness which must be the same in prototype and model in this case.

If model tests show that the bond between transplanted runner and sandbed fails at mean velocity $v_{m.m}$ the mean velocity $v_{m.p}$ at which this will happen in the prototype (nature) may be found from Equation (7) when d_p, d_m and k are known.

The roughness k may be found by measuring the velocities v_2 and v_1 at distances y_2 and y_1 from the bed in the prototype as seen in Figure 5. Applying Equation (6) to these two points of the velocity profile and eliminating v_f yields the iterative formula for k

$$k = \frac{29.7 y_2}{\left(\frac{y_2}{y_1}\right)^{\frac{v_2}{v_2 - v_1}}} \cdot \frac{\left(1 + \frac{k}{29.7 y_1}\right)^{\frac{v_2}{v_2 - v_1}}}{\left(1 + \frac{k}{29.7 y_2}\right)^{\frac{v_1}{v_2 - v_1}}} \quad \ldots\ldots\ldots\ldots\ldots (8)$$

The above mentioned technique was applied to evaluate the impact of flow on an area to be restored in the Florida Keys as outlined in Tecnical Report No. 8301 from University of Florida's Hydraulic Laboratory. The model was established in the University of Florida's hydraulic research flume where a model bed consisting of 10 tons of sand with the same effective grain size ($d_e \cong 0.6$ mm) as the natural bed was constructed . Halodule runners 15 cm to 30 cm long were placed at 90 cm between centers and anchored by 15 cm steel staples made of 6 mm steel wire. Model velocities were measured with precision laboratory propeller meters and checked by discharge observations provided by the flume's main V-notch weir. A total of five tests were performed and the following observations made:

TEST NO. 1:	Model Depth:	$d_m = 0.49$ m
	Prototype Depth:	$d_p \cong 1.37$ m
	Roughness (apparent):	$k = 0.55$ m*
	Model Mean Velocity:	$v_{m.m} = 0.05\ ms^{-1}$
	Prototype Mean Velocity:	$v_{m.p} = 0.07\ ms^{-1}$

Observations: No changes to plants or sand.

* Roughnesses in excess of the depth have been observed by the authors on many occasions in Florida. These values are apparent roughnesses caused by the flexibility of the bed vegetation.

TEST NO. 2: Model Depth: $d_m = 0.48$ m

Prototype Depth: $d_p \cong 1.37$ m

Roughness (apparent): $k = 0.55$ m

Model Mean Velocity: $v_{m.m} = 0.10\ ms^{-1}$

Prototype Mean Velocity: $v_{m.p} = 0.15\ ms^{-1}$

Observations: No changes to plants or sand.

TEST NO. 3: Model Depth: $d_m = 0.48$ m

Prototype Depth: $d_p \cong 1.37$ m

Roughness (apparent): $k = 0.55$ m

Model Mean Velocity: $v_{m.m} = 0.18\ ms^{-1}$

Prototype Mean Velocity: $v_{m.p} = 0.27\ ms^{-1}$

Observations: No apparent change to sand but two plants have had one of their ends pulled out of the sand. The other end for each of these two plants has remained in its original position.

TEST NO. 4: Model Depth: $d_m = 0.47$ m

Prototype Depth: $d_p \cong 1.37$ m

Roughness (apparent): $k = 0.55$ m

Model Mean Velocity: $v_{m.m} = 0.26\ ms^{-1}$

Prototype Mean Velocity: $v_{m.p} = 0.37\ ms^{-1}$

Observations: Some transport of the sand is apparent; however, the ripples are very small.

Three plants have had one of their ends pulled free of the sand.

Some of the stem portions of the grass which were initially covered with sand have been uncovered.

Two staples are now visible (the top).

Most of the plants are surrounded by very small ripples.

TEST NO. 5: Model Depth: $d_m = 0.46$ m

Prototype Depth: $d_p \cong 1.37$ m

Roughness (apparent): $k = 0.55$ m

Mean Model Velocity: $v_{m.m} = 0.32\ ms^{-1}$

Mean Prototype Velocity $v_{m.p} = 0.46\ ms^{-1}$

Observations: There was a tendency for the plants to become uncovered initially; however, ripples then formed around each plant and tended to build up sand around the plant. Two of the plants were almost totally covered at the end of the test and each of the others had a dune built up around it.

These tests seem to indicate that the newly transplanted runners will be affected by the flow at lower mean velocities than the sand bed. Damage to the runners may be expected already at a mean velocity of $v_{m.p} = 0.27\ ms^{-1}$ in the vertical bed.

PROTECTION OF NEWLY RESTORED AREAS

If prototype mean velocities exceed about 0.25 ms^{-1} in water with a depth of around 1.35 m protection against erosion of a newly transplanted seagrass bed may be necessary. This may be accomplished by installing a small dike upstream of the restored bed and another downstream of the bed. These dikes should be slightly higher than the depth corresponding to high tide.

Using the experimental results of Jensen's (1954) research on shelter effect the length L of a dike needed to protect a bed area of length ℓ and width b may be found from the formula

$$L = b + \frac{\ell}{12.5} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \text{(9)}$$

Equation (9) takes into consideration the extend of the shelter zones in the main direction of flow.

CONCLUSIONS

Methods for quantification of the erosion risk of newly restored grassbeds are presented considering erosion of the sandy bed itself as well as flow induced damage to the bond between roots and bed.

A stochastic diagram for the bed shear stress at which erosion of the bed begins is given together with a correction factor to be applied in the case of transverse bed slope.

A special model to prototype transfer law is developed for transfer of observations made of the resistance of newly transplanted runners in the laboratory. Application of this law to observations made in the University of Florida's hydraulic research flume shows that the growth of Halodule runners may be disrupted when the mean velocity exceeds about 0.25 ms^{-1} corresponding to a depth of around 1.40 m. Damage to the bond between root and sand occurs before the sand bed itself is beginning to erode.

Based on earlier experiments with shelter effect a simple formula for the length of dikes used to protect newly restored areas has been developed.

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LITERATURE CITED

Christensen, B.A. Effective grain-size in sediment transport. International Assoc. for Hydraulic Research, 13th Congress, Kyoto, Japan; 1969.

Christensen, B.A. Incipient motion on cohesionless channel banks. Sedimentation, (H.W. Shen editor), Fort Collins; 1972.

Christensen, B.A. Hydraulics, Class notes ECI 4214, University of Florida, Gainesville; 1983a.

Christensen, B.A. The Mechanism of sea floor erosion. Proceedings of the World Dredging Congress, BHRA, Singapore; 1983b.

Forchheimer, P. Hydraulik. Tuebner Verlag, Leipzig Berlin, Germany; 1914.

Fortier, S. and Scobey, F.C. Permissible canal velocities. Trans. American Society of Civil Engineers, Vol. 89; 1926.

Graf, W.H. Hydraulics of sediment transport. McGraw-Hill Book Company, New York; 1971.

Hjulström, F. The morphological activity of rivers as illustrated by river Fyris. Bull. Geol. Inst. Upsala, Vol. 25, Upsala, Sweden; 1935.

Jensen, M. Shelter effect. The Danish Technical Press, Copenhagen; 1954.

Lane, E.W. Stable channels in erodible material. Trans. American Society of Civil Engineers, Vol. 102; 1937.

Lane, E.W. Progress reports on studies on the design of stable channels of the bureau of reclamation. Trans. American Society of Civil Engineers, Vol. 79; 1953.

Leliavsky, S. An introduction to fluvial hydraulics. Constable, London, England; 1955.

Schocklitsch, A. Über Schleppkraft and Geschiebebewegung. Englemann, Leipzig, Germany; 1914.

Schocklitsch, A. Handbuck des Wasserbaues. 2nd edition, Springer, Vienna, Austira; 1950.

Shields, A. Änwendung der Ähnlichkeitmechanik and Turbulenzforschung auf die geschiebebewegung. Mitteil. Preuss. Versuchanst. Wasser, Erd, Schiffsbau, Berlin, Germany, No. 26; 1936.

Sternberg, H. Untersuchungen über das Längen and Querprofil Geschiebeführender Flüsse. Z. Bauwesen, Vol. 25; 1875.

Streeter, V.L. and Wylie, E.B. Fluid mechanics. 6th edition, McGraw-Hill Book Company, New York; 1975.

Technical Report No. 8301. Niles channel restoration project. Hydraulic Laboratory, University of Florida, Gainesville, Florida; 1983.

NUMERICAL-HYDRODYNAMIC MODEL USE IN EVALUATION OF MARSH ISLAND CREATION

Charles K. Eleuterius and Lionel N. Eleuterius
Gulf Coast Research Laboratory
Ocean Springs, Mississippi

ABSTRACT

A computer-based hydrodynamic assessment system was developed to facilitate locating sites for spoil deposition that would favor marsh establishment in estuaries. The system incorporates a numerical model that is suitable for simulating circulation in vertically well-mixed estuaries. The system operates in an interactive mode and possesses a tutorial that explains the system and its proper use. Other components of the system compute statistics, manipulate data files, and generate charts. Computer outputs include tabular information on water elevations, water velocity, and descriptive statistics on these parameters. This capability allows detailed examination of current velocities and water levels around the edge of existing or proposed spoil islands. Very useful graphic outputs are contour charts of water elevations, vector diagrams of the currents, and hydrographs for user-selected sites within the estuary. Substantial hydrological data obtained directly from St. Louis Bay, Mississippi, were used to test and demonstrate the use of the newly developed system. The simulated effect of marsh islands on water circulation and vice versa in different locations in the bay are easily determined. Comparison of graphic and statistical results allow rapid assessment and evaluation. For example, computer simulations showed that a spoil island placed in certain parts of the bay had great effect on bay hydrodynamics. Strong currents around this island location indicated that marsh establishment here would not be favored. When the simulated island was placed in another location, little or no effect on the bay circulation was noted. Consequently, the absence of erosive currents around the latter location would certainly favor marsh establishment.

INTRODUCTION

The successful use of islands, created from the deposition of dredged materials, for the establishment of salt marsh plants depends basically on the judicious selection of the site with regard to the hydrodynamics within the estuary. Improper location of dredge spoil sites and the resulting erosion of man-made islands by strong currents are the primary reasons for failure of costly marsh transplanting projects (Eleuterius 1974). The use of a newly developed hydrodynamic analysis system, which provides a rapid, economical, and comprehensive means of assessing sites and configurations of potential marsh islands, is the topic of this paper. Information that we wanted to obtain from an assessment of the hydrodynamics dictated the required system components. We decided that the computerized system should offer flexibility, allow user-system interaction, provide sufficient information on the water circulation of the study area, convey the information in graphic form wherever possible, and permit quick simulated alterations to the physiography of the study area. These physiographic alterations would include changes in location and configuration of spoil islands. Rapid computer output in the form of graphics would allow numerous assessments upon which planning decisions could be based. Furthermore, to make the system easy to use, we designed a tutorial component that explains the structure and use of the system and how it can be applied to most estuaries. A list of options or "menu" guides the user in implementation of the system.

Price (1947) recognized that within tidal basins there exists a balance between landforms and the forces that act upon them. There have been many contributions to the literature concerning shoreline erosion and sediment transport. Among those of particular importance to the creation of marsh habitats are White (1966), Murray (1970), and Johnson (1974). The design of structures to contain dredged materials for the development of marsh habitats was the subject of a study by Eckert et al. (1978). An investigation of the relationship between the growth of certain marsh plants and the physicochemical character of dredged sediments was conducted by Barko et al. (1977). Although the study reported success in marsh creation within the controlled "green house" environment, it did not identify locations in estuaries where the dredged material should be placed to obtain comparable success. Numerical models of storm surges in estuaries have been developed by Reid and Bodine (1968), Reid and Whitaker (1976), and Reid et al. (1977). Simulation of the general estuarine circulation has been the objective of models developed by Jeglic (1966), Leendertse (1967), Dronkers (1969), Masch (1969), and Mungall and Matthews (1970). The purposes of other numerical models have been the prediction of salinity intrusion in estuaries (Thatcher and Harleman 1972), and the contribution of salinity to estuarine circulation (Elliot and Reid 1976). These references serve here only as a sampling of the major numerical models developed for estuarine hydrodynamics. An assessment of estuarine modeling prepared for the Environmental Protection Agency (Tracor, Inc. 1971) reviewed the developments in mathematical modeling of hydrodynamics, water quality, and various biological processes. The use of mathematical models to delineate and resolve problems within the estuaries was the fundamental theme of a symposium on mathematical modeling techniques in water resources (Orlob 1971). However, the subject of spoil island location was not addressed, although it is of fundamental importance in the successful management of estuaries.

Accounts of hazards to marsh habitat creation on dredged material in estuaries found along the Mississippi Gulf Coast were reported by Eleuterius (1974). After experiencing erosional loss of 12,000 transplants over a 3-year period on a spoil island in Horn Island Pass, Mississippi, he advised the need for more judicious placement of dredge spoil for the successful creation of habitats for marine plants. Further testimony to the need for prudent selection of sites for creation of marsh habitats was provided by Whitehurst (1977) who reported that an island on the James River constructed specifically for the development of marsh habitat was ultimately destroyed by hydraulic forces.

DESCRIPTION OF THE STUDY AREA

St. Louis Bay (Figure 1), located on the Misssissippi Gulf coast, is a shallow, mushroom-shaped basin with a surface area at mean low water (MLW) of about 4,000 hectares with a corresponding average depth of 1.34 m (Eleuterius 1980). The land adjoining the bay, except for the headlands, consists of salt marshes. The mouths of Jordan and Wolf rivers are situated almost diametrically opposite in the upper bay. The bay is tidal with water entering from Mississippi Sound through a relatively narrow passage approximately 3 km in width. St. Louis Bay is approximately 7.5 km wide. The diurnal tides of St. Louis Bay are similar to those of the nearby Gulf of Mexico. The average tidal range within the bay is 0.49 m (Eleuterius 1980). The locations of channels and areas presently used for disposal of dredged material are shown in Figure 2.

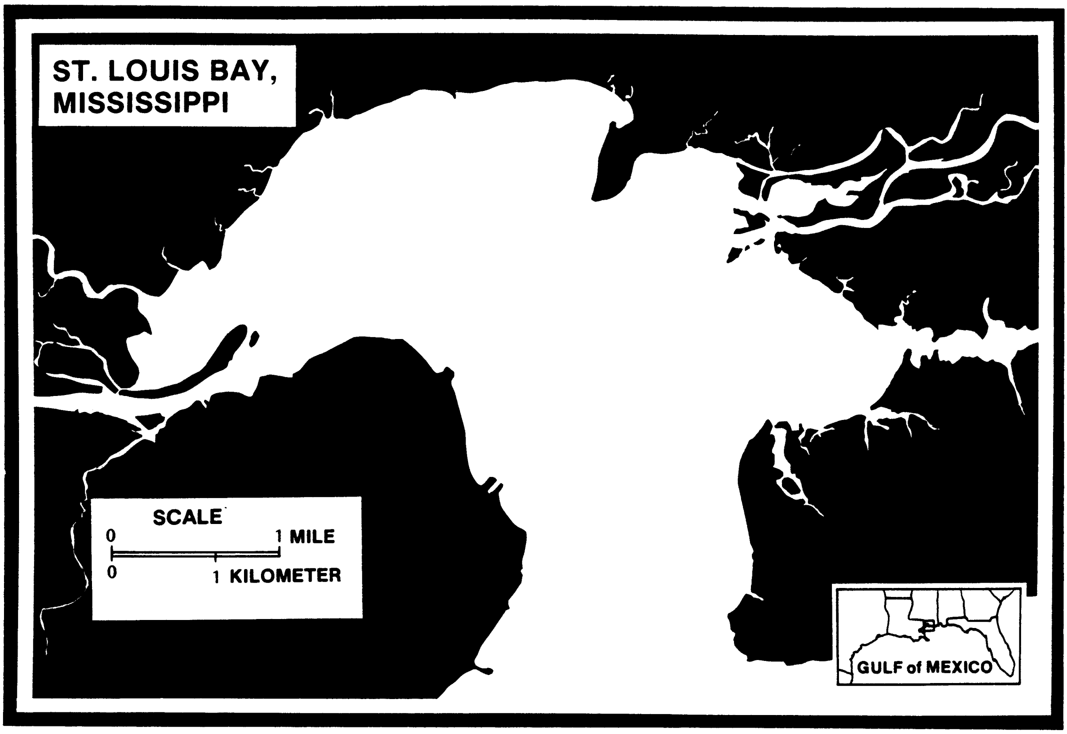

Figure 1. St. Louis Bay, Mississippi.

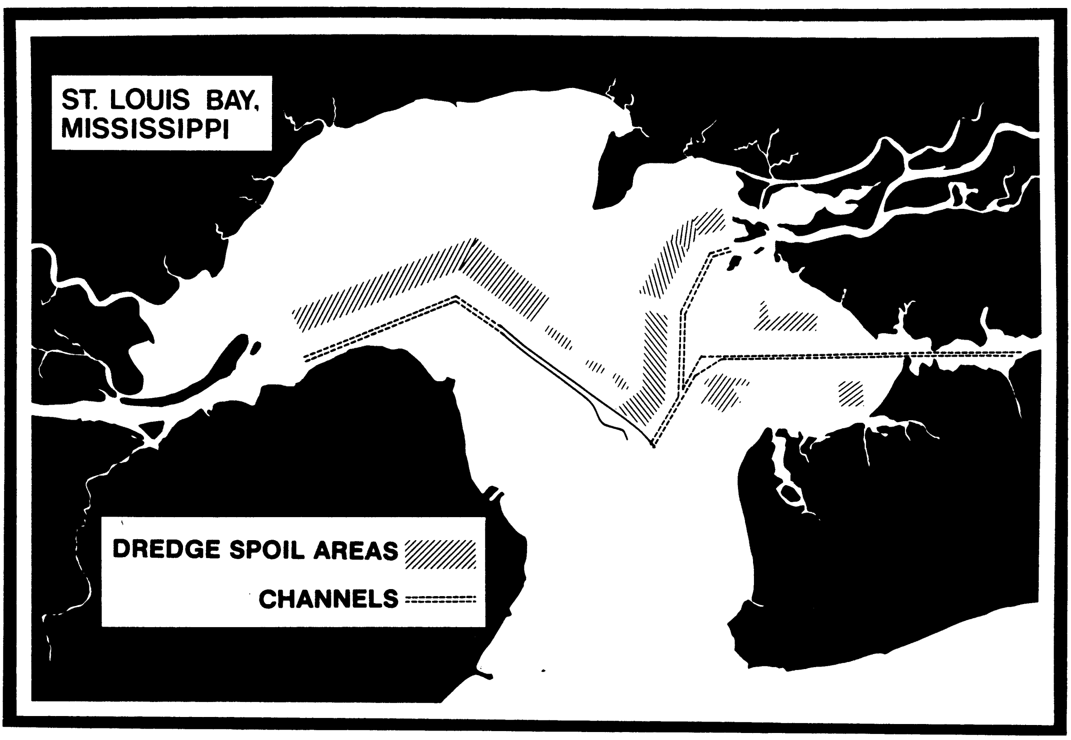

Figure 2. Location of channels and dredge spoil disposal areas, St. Louis Bay, Mississippi.

METHODS

After a review and evaluation of existing hydrodynamic models, we found none were designed to meet our specific needs. However, our search and study indicated that, with modifications, the numerical model of Reid and Bodine (1968) would satisfy our requirements. For example, simulation of the inundation and recession of waters from low-lying lands by the tidal flood and ebb had to be an essential feature of the numerical model. The model also had to be applicable to shallow, well-mixed estuaries. Furthermore, the locations at which water velocities and elevations are computed have to be arranged according to a spatial pattern that would facilitate the rapid graphic display of the results. The finite-difference model of Reid and Bodine (1968), based on the vertically integrated equations of motion and continuity, was appropriate for vertically well-mixed estuaries. Tests, with and without a mathematical expression for the Coriolis force, showed that its effect is negligible in small, shallow estuaries like St. Louis Bay. We modified the algorithm for bottom friction proposed by Reid and Bodine (1968) so that the user could assign various frictional coefficients over the study area. Because only a first-order approximation to the actual circulation is necessary for the initial evaluations, the advective, or field acceleration, terms were omitted. Exclusion of the advective terms substantially reduced computer usage and associated costs. When additional accuracy is required, the advective terms should be included. The subroutines that comprise the hydrodynamic analysis system, i.e. the numerical model, graphic generators, system driver, and tutorial, were written in the FORTRAN computer language. The system is modular in design to facilitate future changes and additions and thus avoid otherwise extensive and costly reprogramming.

The schematized bathymetry from an earlier study on St. Louis Bay (Eleuterius, In Preparation) was used to test the numerical model. The length of a side of the grid cell was 300 m, which was determined as a compromise between spatial resolution and computer core memory capacity. This latter point should be emphasized here. Had a computer with a larger storage capacity been used, a smaller grid size could have been specified that would have resulted in a finer spatial resolution. Data on tidal currents, tidal elevations, and winds, obtained during an extensive and detailed field study on St. Louis Bay, were used to calibrate the numerical model (Eleuterius 1980). A sine function, scaled to generate one high-water and one low-water stage per 24 hours with a range of 0.5 m, was used to simulate tidal forcing in this study because it is a good approximation to the actual tides of St. Louis Bay. The tidal circulation patterns of the bay with and without the presence of simulated spoil islands were computed and both tabular and graphic output obtained via the hydrodynamic assessment system. The graphics were used to show the effect of the configuration and location of spoil islands on the tidal dynamics of the bay and vice versa.

RESULTS

Concept of the hydrodynamic analysis system. The conceptual organization of the computer-based hydrodynamic assessment system is shown in Figure 3. The "driver" which includes the menu of options is that portion of the system that allows communication with the user in an interactive mode. Depending upon the user's responses and instructions, the system performs various tasks and provides requested outputs. Among

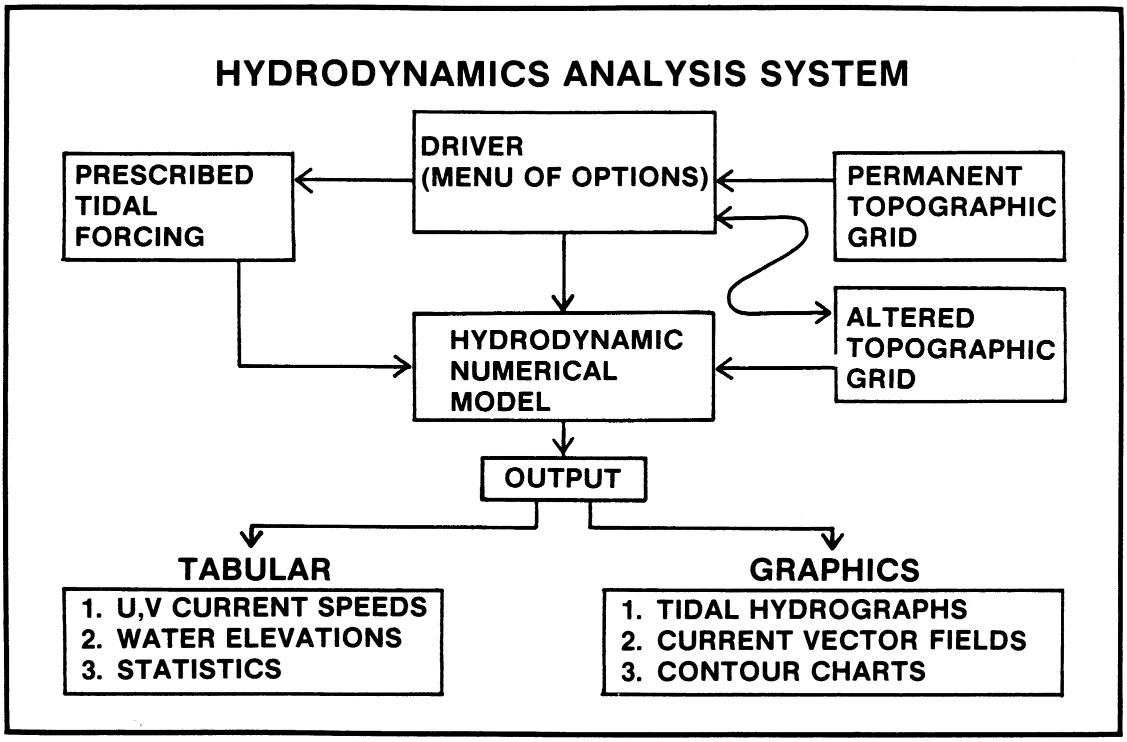

Figure 3. Conceptual organization of computer-based hydrodynamic assessment system.

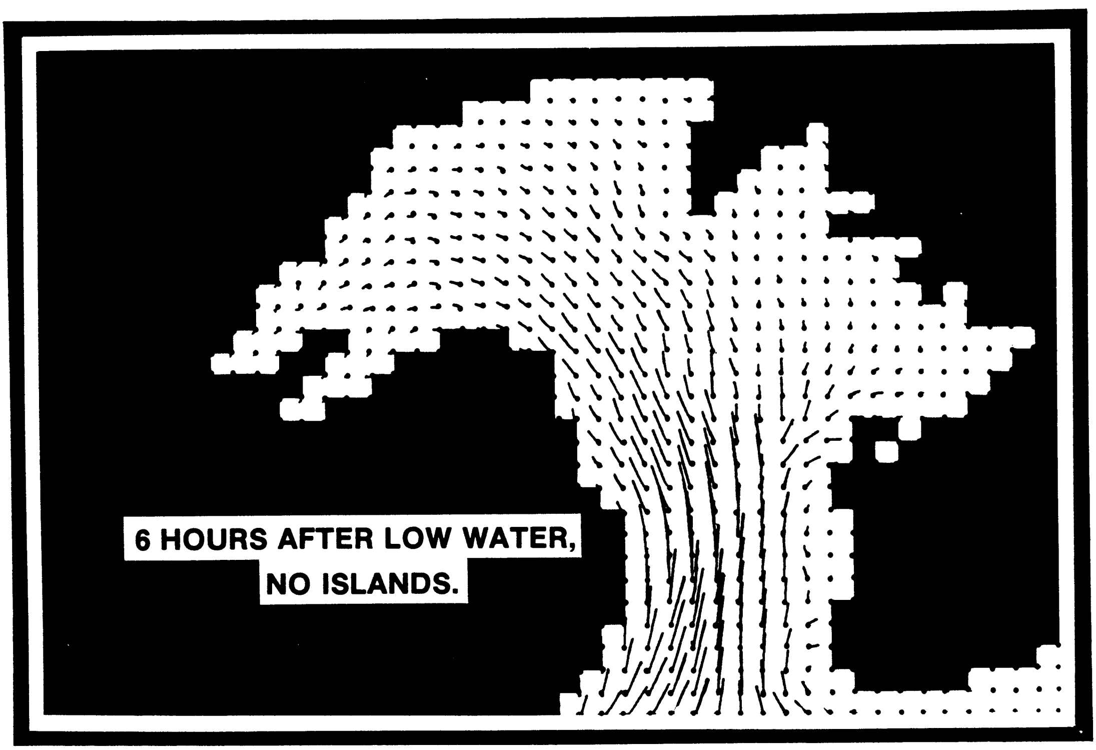

Figure 4. St. Louis Bay without islands, computed currents 6 hours after low water.

such tasks performed is the generation of an altered topographic grid which would incorporate user-specified simulated spoil islands. The numerical hydrodynamic model is executed at finite increments of prototype time. At each of these time steps, the model computes water elevations and X and Y components of current velocity over the modeled area. Additonally, the model allows for recording, in accordance with the number of time steps executed, a time series of water elevations at user-specified locations. A means of conveying this massive amount of information in a form that would facilitate rapid evaluation by the user was an essential part of the original system concept.

Graphic displays via cathode ray tube (CRT) computer terminals present these large amounts of data in a fast and condensed form. Circulation within the estuary is depicted by current vectors over the study area. Hydrographs, plots of water elevations versus time, can be produced for user-specified sites within the study area. The spatial characteristics of the estuary as to current speeds and water elevations may be presented in the form of contour charts. All the computer-generated graphics are optional.

Corresponding tabular information on current strength and water elevation for the entire study area is output. Statistics on the mean and extremes of current speed and water elevation would be especially useful in assessing the effect of the currents on proposed spoil islands. More detailed information can be obtained by subdivision of the 300-m grid used in this study. However, to double the resolution would more than double the required computer storage and computer time. Presentation of the computer programs or the algorithms was not practical considering their length and the limitation of space in the present paper. However, a copy of the computer program and instructional information can be obtained from the senior author of this paper.

Calibration of the numerical-hydrodynamic model. The accuracy of the model was ascertained by comparison of computed water levels and current velocities with observed water levels and current velocities at several different places in the St Louis Bay. Results obtained showed that computed and observed water levels and current velocities were in close agreement for several different sampling stations in the bay.

Use of the hydrodynamic analysis system. The current vector diagram depicting the vertically averaged tidal currents by use of the sine function as a tidal forcing function is shown in Figure 4. The direction of the line extending from the grid points indicates the direction of the current at that point. The length of each line corresponds to the magnitude of the current at the grid point. The longer the line, the greater the speed. Although the vector diagrams normally include scales for interpretive purposes, they do not appear on the charts included here. The current patterns in the bay are vividly illustrated. The greatest current speeds occur in the constricted entrance, becoming reduced in the upper portion of the bay with deflections to the east and west. The tidal current in the northwest portion of the bay is eventually deflected to the southwest. The current patterns during an ebbing tide are shown in Figure 5. The current patterns for the rising and falling phases of the tide shown in Figures 4 and 5 are influenced by the natural configuration and bathymetry of the bay as well as by the submerged spoil area.

The effect of locating a large spoil island in the lower portion of St. Louis Bay on tidal currents during flood stage is shown in Figure 6.

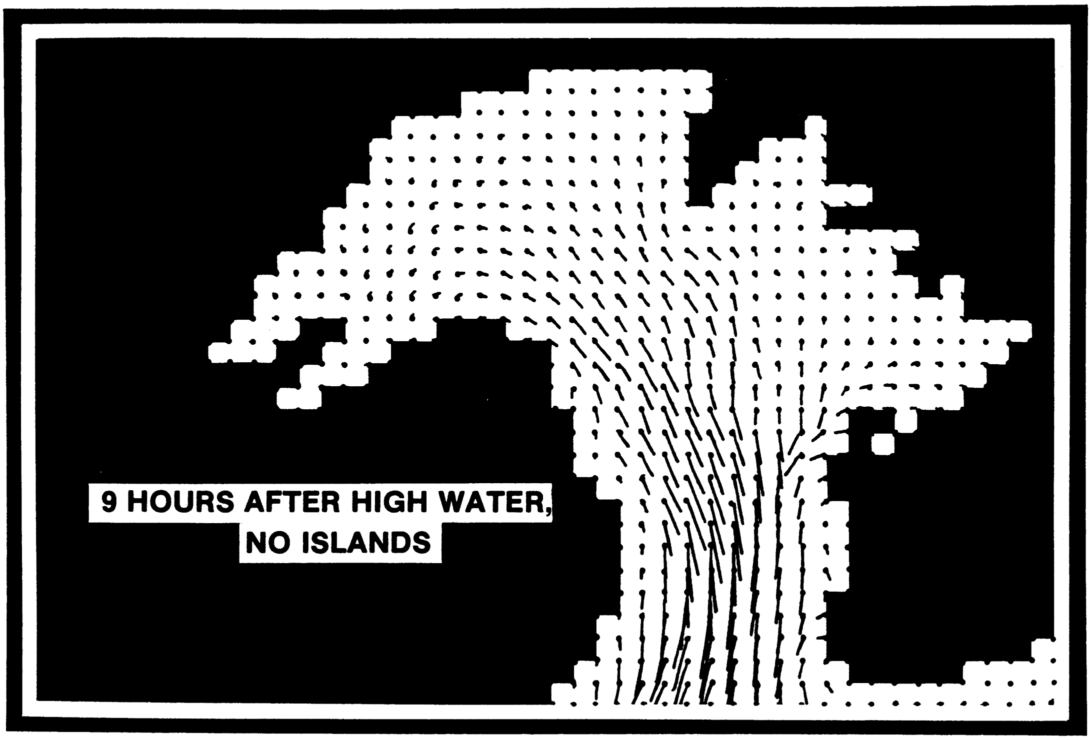

Figure 5. St. Louis Bay without islands, computed currents 9 hours after high water.

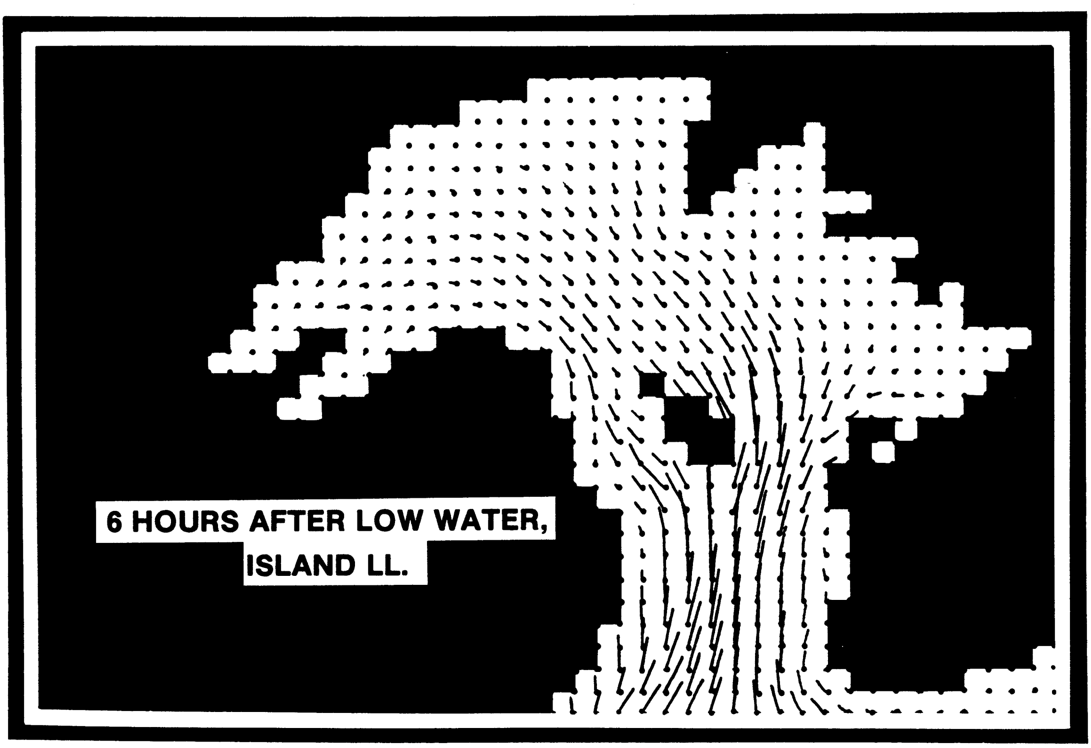

Figure 6. St. Louis Bay with island, computed currents 6 hours after low water.

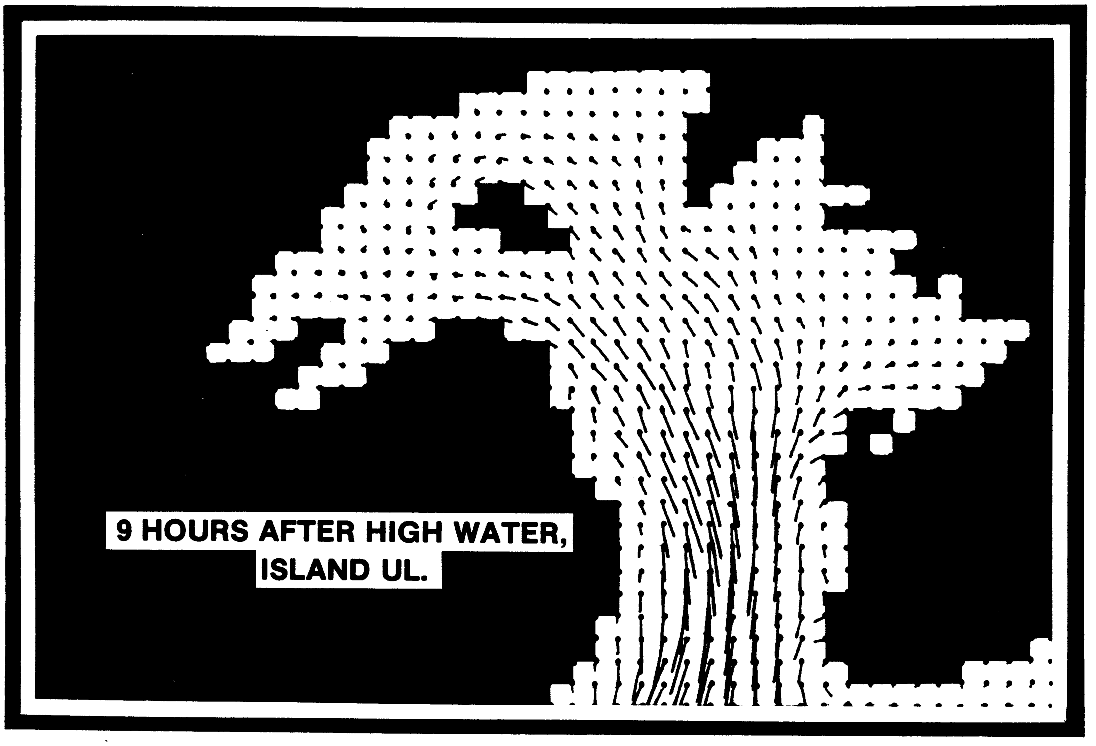

Figure 7. St. Louis Bay with island, computed currents 9 hours after high water.

Locating a spoil island in this area would cause a severe deflection of the tidal currents, with stronger currents being displaced closer to the mainland shores both east and west of the island. The point at which the normal deflection of the current to the northwest occurs in the upper bay has shifted farther north.

Another test of the hydrodynamic analysis system produced the current vector diagram shown in Figure 7. In this test the simulated spoil island is located in the northwest portion of St. Louis Bay and the vector diagram shown depicts the current patterns during the ebb stage. Although some deflection of tidal currents is apparent, the location of the spoil island in this upper portion of St. Louis Bay seems to have little effect on the ebb tide circulation pattern. No appreciable changes were noted on the current patterns during the flood stage. Graphic presentations were limited in the present paper again because of restrictions on length.

DISCUSSION

The survival of angiosperm transplants from tidal marshes to dredged spoil islands depends upon many factors. Each location within and between geographical areas must be considered on an individual basis. Predictive information obtained through the use of numerical models on the hydrodynamics of coastal waters would be basic in the evaluation and assessment process of site selection for open water spoil deposition. If a spoil island is to be created for the establishment of plants which will lead to productive tidal marsh, we realize the wind-driven waves, sediment type, water depth, and a host of other factors are also important. However, experience of the junior author has shown that spoil islands will rapidly erode in areas of strong tidal currents. Furthermore, many transplants placed low on the tidal plane are lost because of changes in the island configuration. Experimentation with various island configurations in relation to the tidal regimes of St. Louis Bay would obviously result in the selection of a shape compatible with the particular tidal hydraulics of the bay. We feel that the use of the numerical model presented herein would be beneficial in two ways where open water spoil deposition or marsh island creation was part of a biological-engineering project. The "least effect" on the tidal hydraulics of the bay or water body could be rapidly evaluated and the "best" location of a spoil island intended for marsh establishment could be obtained, simultaneoulsy.

In the past, considerable knowledge of the numerical model algorithm and a definite proficiency with computers were required. These prerequisites discouraged, if not prevented, the use of numerical models for site evaluation.

We recommend the use of the computer-based, hydrodynamic analysis system proposed here because our system allows a rapid preview of the consequences of using any number of possible disposal sites and their configurations. Furthermore, it eliminates the need for detailed knowledge of computers or the computer programs. One obvious benefit is that a larger area may be considered in site evaluation than is economically feasible by conventional studies because of the extensive data aquisition that would be required.

The numerical-hydrodynamic model presented in this paper used approximately 550 grid points to represent the normal water area. To obtain the same resolution by conventional techniques would require that 550 stations be sampled simultaneously with a high frequency. This would be an extremely costly, if not impossible, task. The numerical model can also be used to simulate the hydrodynamics during severe weather conditions when it would be impossible to conduct conventional sampling. Our hydrodynamic analysis system is flexible and can be applied to estuaries throughout the world.

Estuarine modification will continue and perhaps accelerate in the future in a direct relationship to the population and industrial growth projected for the coastal areas of the world. Therefore, the most important decisions in estuarine engineering projects, where dredging and spoil deposition must be considered, are whether or not marsh creation projects should be attempted; and if so, where they should be carried out. The use of the hydrodynamic analysis system presented in this paper should play a major role in this decision-making process before costly mitigating programs, such as marsh creation, are undertaken.

ACKNOWLEDGMENTS

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REFERENCES CITED

Barko, J. W., R. M. Smart, C. R. Lee, M. C. Landin, T. C. Sturgis, and R. N. Gordon. Establishment and growth of selected freshwater and coastal marsh plants in relation to characteristics of dredged sediments. Technical Report TRD-72-2:1-41, U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS. 1977.

Dronkers, J. J. Tidal computation for rivers, coastal areas, and seas. J. Hydraul. Div., Proc. Am. Soc. Civ. Eng., 95(HY1):29-77; 1969.

Eckert, J. W., M. L. Giles, G. M. Smith. Design concepts for in-water containment structures for marsh habitat development. Technical Report D-78-31, U. S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS. 1978.

Eleuterius, C. K. Environmental Baseline Survey, St. Louis Bay, 1978. Vol. 1, Physical Oceanography; Sec. 2, Hydrodynamical-Numerical Model. Final report of Gulf Coast Research Laboratory, Ocean Springs, MS to E. I. du Pont de Nemours and Co., Wilmington, DE. 1980.

Eleuterius, C. K. Application of a hydrodynamic-numerical model to the tides of St. Louis Bay, Mississippi. (In preparation)

Eleuterius, L. N. A study of plant establishment on dredge spoil in Mississippi Sound and adjacent waters. Final report to U. S. Army Corps of Engineers, Mobile District, Contract No. DACW01-72-C-0001. 1974.

Elliot, B. A. and R. O. Reid. Salinity induced horizontal estuarine circulation. J. Waterw. Harbors Div., Proc. Am. Soc. Civ. Eng., 102(WW4):425-442; 1976.

Jeglic, J. M. DECSIII, Mathematical simulation of the estuarine behavior. Report No. 1032, General Electric ReEntry Systems Department. 1966.

Johnson, B. H. Investigation of mathematical models for the physical fate and prediction of dredged material. Technical Report D-74-1: 1-54, U. S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS. 1974.

Leendertse, J. J. Aspects of a computational model for long-period water-wave propagation. Thesis, Technische Hogeschool, Delft. 1967.

Masch, F. D. A numerical model for the simulation of tidal hydrodynamics in shallow irregular estuaries. Technical Report HYD-12-6901, Hydraulic Engineering Laboratory, Department of Civil Engineering, The University of Texas at Austin. 1969.

Mungall, J. C. H. and J. B. Matthews. A variable-boundary numerical tidal model. Report R70-4, Institute of Marine Science, University of Alaska. 1970.

Murray, S. P. Settling velocities and vertical diffusion of particles in turbulent waters. J. Geophys. Res., 75(9):1647-1654; 1970.

Orlob, G. T. Mathematical modeling of estuarial systems. Proceedings, International Symposium on Mathematical Modeling Techniques in Water Resources Systems. 1971.

Price, W. A. Equilibrium of form and forces in tidal basins of the coast of Texas and Louisiana. Bull. Am. Assoc. Pet. Geol., 31: 1619-1663; 1947.

Reid, R. O. and B. R. Bodine. Numerical model for storm surges in Galveston Bay. J. of the Waterw. Harbors, Proc. Am. Soc. Civ. Eng., 94(WW1):33-57; 1968.

Reid, R. O., A. C. Vastano, R. E. Whitaker, and J. J. Wanstrath. Experiments in storm surge simulation. Pages 145-168 in E. D. Goldberg, Ed., The Sea: Ideas and Observations on Progress in the Study of the Seas. John Wiley and Sons, Inc., New York, NY. 1977.

Reid, R. O. and R. E. Whitaker. Wind-driven flow of water influenced by a canopy. J. of the Waterw., Harbors and Coastal Eng. Div., Proc. Am. Soc. Civ. Eng., 102(WW1):61-78; 1976

Thatcher, M. L. and D. R. F. Harleman. A mathematical model for the prediction of unsteady salinity intrusion in estuaries. Sea Grant publication MITSG-72-7, Massachusetts Institute of Technology. 1972.

Tracor, Inc. Estuarine modeling: An assessment. Report to National Coastal Pollution Research Program, Water Quality Office, Environmental Protection Agency. 1971.

White, W. A. Controlled investigation of the movement of dredge sediments as a density current. M.S. Thesis, University of Texas, Austin, TX. 1966.

Whitehurst, E. E. Chronology of island construction, James River habitat development site. Internal working document, March 1977, U. S. Army Corps of Engineers, Norfolk District.

CONVERSION OF AN IMPACTED FRESHWATER WET PRAIRIE INTO A FUNCTIONAL AESTHETIC MARSHLAND

JOSEPH L. GILIO

Wetlands Management, Inc.
P.O. Box 1122
Jensen Beach, Florida 33457

ABSTRACT

An isolated wet prairie intentionally preserved in land clearing operations developed an increase in hydroperiod. Hypericum fasciculatum originally dominated in this palustrine, emergent, seasonally flooded wet prairie, but completely died out during more than two years of an increased water regime. In the normal time span for succession, invasive emergents, both native and exotic, did little to improve the aesthetic and biological functions of this wetland. Intervention was selected to shorten this time period, and select the preferred new emergent marsh dominants. Plant selection was dictated by consideration of aesthetics, hardiness, growth-spread, and multi-species associations as revealed in local marshes and the National Wetlands Inventory. Manual removal of dead and selected living macrophytes was followed by transplanting of Pontederia cordata Sagittaria latifolia, and S. graminea, Cladium jamaicense and Nymphaea odorata among others. Cost effective maintenance has been enhanced through introduction of mallard ducks. The use of chemical controls has been reduced to an absolute minimum. Aesthetic and biological values of wetlands have increased, and the marsh has attracted many wading birds of special concern to the State of Florida.

INTRODUCTION

With the increase in awareness of wetlands values for the benefit of man has come an effort to incorporate wetlands into residential developments. The preservation of wet prairies in development has been a recent undertaking by planners/engineers, and it has been of mixed success.

Piper's Landing, a residential development of 234 ha in Palm City, Martin County, Florida, was intentionally planned in the late 1970's to incorporate existing wet prairies into the drainage and landscape features of the site. This paper reports the successes and limitations to date in converting a drowned wet prairie into a marsh of good aesthetics and biological diversity within an urban development.

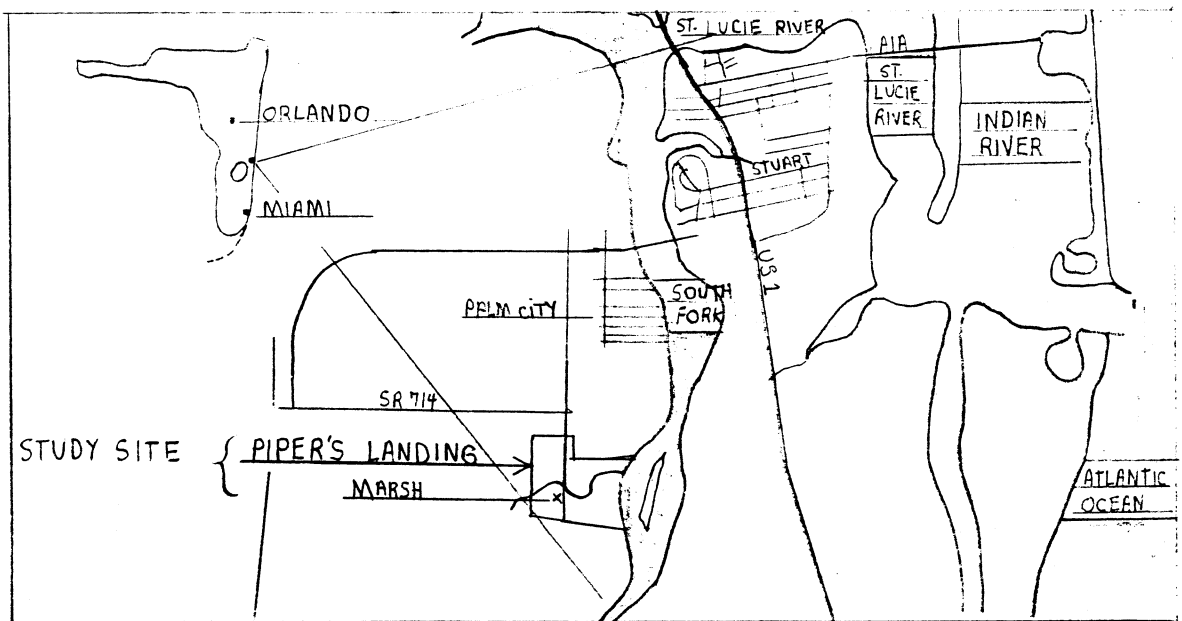

FIGURE 1

Area Description

The location of Piper's Landing Development is shown in Figure 1. The original site consisted of a slash pine flatwood interspersed with several isolated wet prairies dominated by grasses, rushes, and sedges and bisected by a bottomland wet hammock bordering a stream. This stream drains into the South Fork of the St. Lucie River where brackish water conditions exist in a red mangrove dominated shoreline. After development, the wetland habitats were preserved but much of the flatwoods was substantially altered. This alteration involved additions of fill, dredging of lakes and road construction. Runoff on the low profile development is directed from imprevious surfaces through culverts into lakes with spreader berms. Overflow from lakes via wiers is directed into the stream. Wet prairies receive water from greater areas than their original drainage. Culverts and grassed swales convey precipitation from impervious surfaces to the receiving wet prairies in the finished development. Fill added to surrounding uplands may also increase groundwater storage and hence groundwater input into the wet prairie.

Preserved Wet Prairies

Twelve ha of wet prairie originally existed on site prior to development with 60% preserved. The preserved 7.2 ha are distributed among fifteen wet prairies ranging in quality from good to poor. Judgement was made on the basis of aesthetic value, species diversity, plant health and lack of exotics. This paper deals with two of these preserved wet prairies. The first wet prairie of 2 ha is shown in Figure 2. During the wet season, this prairie was continuously inundated to an average depth of 15 cm, which periodically increased to a maximum of 40 cm. It is dominated by a grass tentatively identified as _Andropogon sp_. with some St. John's wort (_Hypericum fasciculatum_), and understoried with _Lycopodium sp_. with minor representations of _Eleocharis flauescens_, _E obtusa_ and _Eriocaulon compressum_. The upland surrounding this wet prairie was cleared the beginning of 1981. A 3 to 15 m wide fringe of upland consisting of slash pine and saw palmetto scrub was left intact around 60% of the perimeter of the wetland. Surrounding

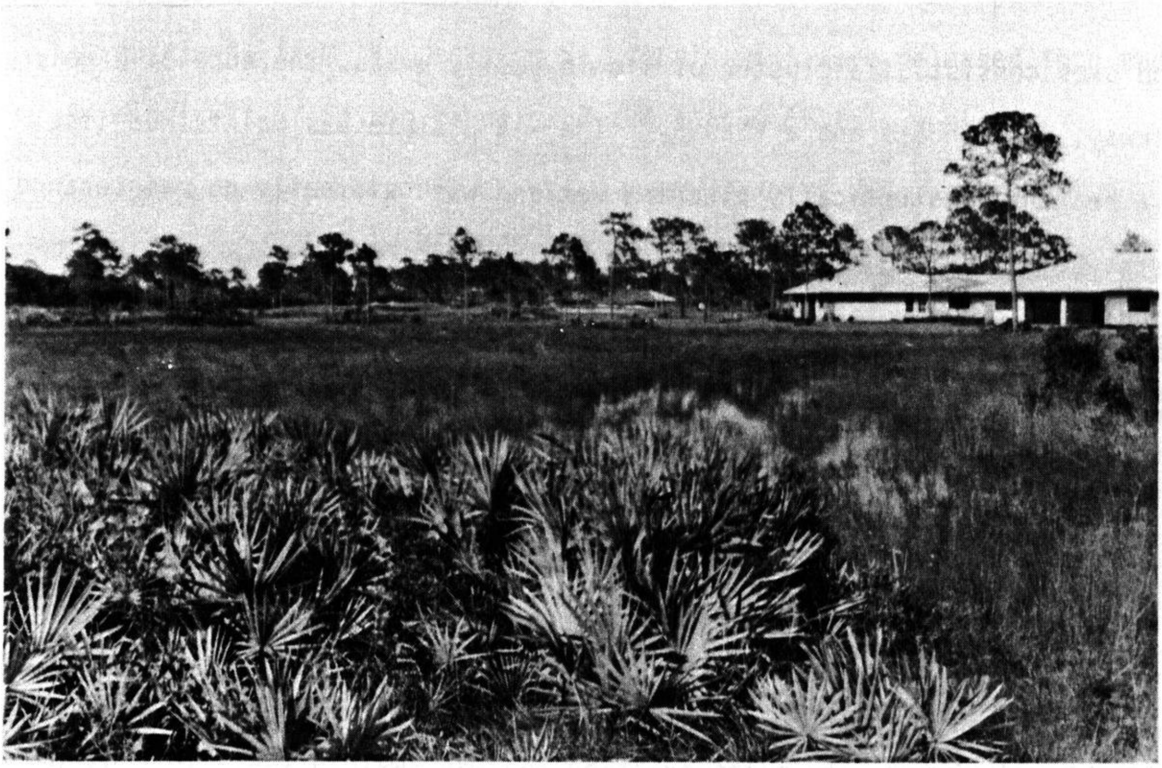

Figure 2. A successfully preserved wet prairie incorporating fringe of upland ecotone.

Figure 3. The drowned wet prairie which was restored into a marsh.

land uses consist of a cluster of single family units, the edge of a golf fairway, a clubhouse and a roadway. The wet prairie has maintained itself as a healthy, aesthetically pleasing wetland with virtually no maintenance costs.

The location of the second and smaller St. John's wort (*Hypericum fasciculatum*) dominated wet prairie is at the junction of the development entrance and the 18th green. Thirty percent of the original 0.2 ha circular depression was filled along the edge closest to the golf green. The opposite edge was just infringed upon with sand fill in construction of the entrance roadway, so that 70% of the original wet prairie was left intact. The original overflow was a low spot in the surrounding perimeter berm draining into a pre-existing roadway swale/ditch. This outlet was retained but may have been raised by about 10 cm. The overall result was an alteration in hydroperiod from a typical semi-permanently flooded wetland to a permanent water covered one between 30-60 cm deep. The increased hydroperiod appears to be due to multiple causes. The damming of the outlet accounts for part of the increase of water level but not its supply. The increased water supply apparently comes from irrigation, and/or a raised water table beneath the artificially created golf green hill. This flooding of the wet prairie, which began in February 1981, occurred during a prolonged period with little rainfall. For the next 20 months, a permanent water cover existed at various depths so that by October 1982, the *Hypericum fasciculatum* was essentially dead. This can be seen in Figure 3. Also, the toe of the slope of the 18th green had been invaded by cattail (*Typha sp.*), water primrose (*Ludwigia octovalis*), and willows (*Salix*). Furthermore, the mucky nature of this shoreline prevented mechanical maintenance. In addition, the floating algae-fern symbiont, *Azolla caroliniana* occupied about 20% of the surface and apparently was increasing as nutrients from the decaying St. John's wort were released into the water. An algal mat of Spirogyra covered the bottom of this wetland.

METHODS AND MATERIALS

Prior to restoration, no herbicidal or algacidal treatment had ever been applied directly to this wetland, nor was spray drift a real possibility to account for the severely stressed nature of the H. fasciculatum. A chemical analysis was made on the water sample of this drowned wet prairie using the Hack FF-2 kit. The results of the analysis are given in Table 1 and indicate a high surface dissolved oxygen accountable both from atmospheric diffusion and photosynthesis by the Spirogyra. The shallow water was highly dystrophic, of moderate hardness, low in nitrate and nitrite but high in ammonia. This is consistent with the moderate Azolla and Spirogyra growth and ammonia production in anaerobic muds. The low chloride indicates virtually no salt water intrusion even though the site is near an estuarine river. The chlorinity is explainable from rainfall input and chloride application in fertilization. The most unusual chemical feature is the pH of 8.5. Lake systems in the vicinity of this wet prairie are in the 7-7.5 range. Apparently the areal ratio of filamentous algae to total water area is much higher in this wetland than in the lakes of similar water chemistries, so that the photosynthetic increase of pH by CO_2 removal is very pronounced in this wetland.

Plan of Attack

Four options were considered to bring this wet prairie into a functionally autotrophic (P/R > 1) wetland of high aesthetic value. Conversion of the drowned wet prairie into a pond was not cost and time effective in terms of permitting requirements and real estate sales considerations. To reconvert the hydroperiod back to the original wet prairie involved transplanting and survival of those species for which little was known. To let nature develop its own species assemblage would have resulted in a marsh of the exotic torpedo grass (*Panicium repens*), and invasive natives water primrose (*Ludwigia octovalis*) with dog fennel (*Eupatorium capillifolium*) along the shoreline, with a cattail-dominated interior with little opportunity for maintenance and of poor aesthetics. In addition, the time scale for natural transition would have taken several to many years during which the dead branches of the St. John's wort would have slowly decayed presenting an unsightly view. The last option and the one selected was to intervene in natural succession and replant this marsh with suitable plant material.

Table 1. WATER CHEMISTRY 10/1/82 of the Transitional Wet Prairie-Marsh at Piper's Landing.

Parameter	Value
Conductivity	130 umhos/cm
pH	8.5
NH_3	0.72 mg/l
NO_2^-	BDL
NO_3	BDL
CL	30 mg/l
Alkalinity	24 as mg $CaCO_3$/l
free CO_2	0 mg/l
O_2	10.24 mg/l
Hardness	100 as mg $CaCO_3$/l
Temperature	30^oC

The beginning of restoration involved hand cutting of the cattail, willow and primrose, and raking out all the dead St. John's wort. Two monotypic cattails clumps were retained. Clean sand fill was added to the upland portion of the slope of the 18th green to give a firmer and more manageable slope for riding mowers and to allow most precipitation to infiltrate prior to entering the wetland.

The planting scheme involved converting a palustrine, non-persistent wetland into a rooted vascular, aquatic bed, and narrow leaved persistent wetland.[1] At overflow level, the deep water (30-60 cm) areas overlaying mucky hydrosoil were planted with clumps of fragrant white water lily (Nymphaea odorata).

The shallow water areas, whether mucky or sandy, were planted with a variety of marsh plants. Pickerelweed (Pontederia cordata) was planted in clumps throughout the marsh. The shoreline areas had some naturally invasive arrowhead (Sagittaria graminea) some of which were transplanted to other areas for visual appeal. Sagittaria lancifolia was added for species diversity at the water's edge. A major effort to introduce sawgrass (Cladium jamaicense) in clumps was made. Along the shoreline, transplanted baker's cordgrass (Spartina bakerii) grew to 1.5 m in height and flowered within one year. Several swamp maple trees (Acer rubrum) were planted in the moist upland and have thrived. Bacopa monnieri was introduced in the very shallow water covered muck area. Clumps of Eleocharis flavenscens and Fimbristylis sp. were transplanted successfully along the wetland/upland border in the exposed sandy fill with the intent of forming a small ecotone between the marsh area and the golf green.

Landward of the two preserved cattail clumps, yellow-eyed grass (Xyris iridofolia) and wax myrtle (Myrica cerifera) were intentionally cultivated. At the outlet, existing clumps of soft rush (Juncus effusus) and wax myrtle

(*Myrica cerifera*) were landscaped along the narrow outflow swale. A concerted effort was made to preserve these and other wetland plants into the landscaped program.

RESULTS

With the exception of most of the sawgrass, all of the introduced wetland plants have thrived within the past 18 months. Especially good growth was noted in the *Pontederia cordata* which developed into purple flowered clumps 1-2 m thick. Water lily growth resulted in petiole extension up to 1 m and flowering throughout the first mild winter. The swamp maples put on 30 cm of new growth in the first spring. Initially, *Bacopa* had spread rapidly over a thick muck (ecotone) area to the exclusion of any other plants. During the first summer willow primrose and cattails grew through the mat and presented a maintenance problem area.

Cladium jamaicense had a 20% transplant success after 18 months. Wetlands Management, Inc. has had a greater than 50% transplant success and 10 to 20 fold increase in aerial extent after 18 months in other restoration projects. The low success rate here can only be attributed to the physico-chemical nature of the hydrosoil. It was highly anaerobic, thick and odoriferous, being neither sapropel nor dy but having some properties in common with both these lacustrine sediment types. In the more sandy bottom areas of the marsh and in littoral zones of retention ponds, sawgrass transplanting and subsequent growth to flowering has been excellent.

The preserved vegetation thrived well also. The major maintenance problem has been to keep the cattail growth within the two preserved areas. This has been accomplished in part by placing cinder blocks in a border around the waterward extension of the cattail. The top of the blocks are below the control water elevation. Routine maintenance by cutting or pulling extraneous cattails which occasionally sprout beyond the cinder-block containments or spread under the blocks by rhizome growth has been carried out with a few hours of manual labor every half year.

The filamentous alga, Spirogyra, has not grown excessively as anticipated. Part of the reason has been the high degree of biological control afforded by intentional introduction of seven male mallard ducks. They have made the marsh their home base and have provided an unexpected public relations benefit to the Piper's Landing real estate staff. The ducks have also effectively controlled outbreaks of duckweed (Lemna minor).

Azolla was present prior to restoration and became an aesthetics problem within three months of the marsh replanting. A minimum amount of diquat was selectively applied twice within the first half year to reduce its concentration to acceptable levels. Care was taken to do this when the ducks were feeding in an adjacent lake. With stabilization of the habitat, Azolla reached an acceptable aesthetic level without further chemical control till the present time. This Azolla treatment is the only chemical control which has been applied to this restoration project. All other maintenance has been through biological, mechanical or manual control.

Routine management of the marsh involves mechanical and manual maintenance, minor water level adjustments and limited chemical control. Monthly mechanical and manual maintenance occurs along the fringe in two areas. One area is steep sloped turf to the water control level with a border of arrowhead growing in the water. It requires 0.4 man hours per month of grass cutting, raking and some hand pulling of undesireable emergents. The water-spider orchid (Habenaria repens) has naturally introduced along this fringed area. The other fringe area is shallow sloped and has developed an extensive growth of water primrose and cattail. It has required 4 man hours per month of maintenance on a shoreline length similar to the steep sided area. This second area will be converted to a steep sided fringe to reduce maintenance and improve both aesthetic and view sites.

The open water areas require a once yearly manual clean-out of 20 man hours removing extraneous cattails and thinning of pickerelweeds. Water level adjustments involved a one-time addition of 300m^3 of irrigated water in May of 1983 near the end of a prolonged seasonal dry period. At all other times, precipitation and groundwater flow has maintained the water level at or near the control elevations. Chemical control involved selective application of diquat on Azolla twice within three months of replanting.

An unexpected benefit of this restoration project has been a definite increase in the number, frequency and species of aquatic birds which visit in the marsh. These are listed in Table 2, which by no means contains a complete tabulation since no special effort was made to collect this data.

Some of these species are of special concern to the State of Florida mainly because of loss of habitat. This restored marsh serves as a more preferred habitat for these birds than the pre-existing flooded wet prairie. The ducks and herons were also observed feeding in the marsh. The marsh has a high sustained population of mosquito fish (Gambusia affinis), tadpoles and odonate larvae.

Table 2. Aquatic birds observed frequently at the restored marsh at Piper's Landing. The mallard ducks were introduced as biological controls for algae. SC= special concern to the State of Florida; E = endangered to the State of Florida.[2]

American coot	*Fulica americana*	
green heron	*Butorides striatus*	
little blue heron	*Florida caerulea*	SC
snowy egert	*Leucophoyx thula*	SC
cattle egert	*Bubulcus ibis*	
wood stork	*Mycteria americana*	E
mallard duck	*Anas platyrhynches*	

Reference to Figure 4 indicates part of the restored marsh. Clumps of the planted and preserved vegetation can be seen starting to thrive during the first spring after a previous fall replanting.

DISCUSSION

The conversion of the wet prairie into a marsh was dictated in species selection by selecting those plants which dominate in natural marshes. Pickerelweed (*Pontederia cordata*) and the arrowheads (*Sagittaria latifolia* and *S. graminea*) proved to be excellent marsh construction material for their aesthetics, limited laterial spread but vigorous growth with flowering. The white water lily (*Nymphaea odorata*) was instrumental in creating a more open water habitat. Its blossoms added color to the marsh throughout the first mild winter of 1982-83. Sawgrass (*Cadium jamaicense*) will thrive in constructed marshes where the hydrosoil is sandy, marly or of moderately organic content.

The maintenance of this wet prairie turned marsh had to consider the dynamic nature of the planted and invasive plants. In the open water situation, chemical control of *Azolla* proved so successful that this species,

Figure 4. Part of the restored wet prairie planted with marsh vegetation. Photo taken 6 months after the fall replanting.

although always detectable, has never reestablished itself as a dominant. Biological control of Spirogyra by ducks and tadpoles has eliminated any other algae control. Control of cattail spread outside the preserved areas could have been obtained through chemical control but was not selected based upon both personal choice and visual undesirability of massive cattail die off. The cinderblock border was only partially successful in stopping rhizome growth beyond the preserved areas. A border area of more expensive cypress planks buried 30 cm deep into the muck with the top below water level could have proven more effective in stopping rhizome growth. Germination of cattail seeds occurred in open water areas, but removal by cutting or pulling has been effective.

The steep shoreline curl has been and remains virtually maintenance free. The shallow sloped shoreline and the small ecotone area have proven to be the most difficult of the maintenance problems. These two troublesome areas will be modified to a steep sloped area for the shallow sloped shoreline and a wetland tree with fern understory for the ecotone area.

During the end of a prolonged recent dry spell period with extensive loss of water cover, reseeding of pickerelweed occurred so that less open water is now available. Piper's Landing has opted for an open water vista as more appealing. Water input would be needed during any periods of dryness to prevent reseeding and maintain the open water feature on a cost-effective manner.

The increase of bird life associated with the newly planted marsh may be attributed to the overall increase in species diversity and productivity of the ecosystem. Also, many existing wet prairies have a grass/herb/sedge cover of about 100%. A plant cover of about one half the areal extent of wetland correlated with high aquatic plant use.[3] The open water to total marsh area ratio is high in this marsh and apparently is another reason for the increased bird wildlife use.

The increase of real estate development in the Florida peninsula can co-exist with preservation of wet prairies. However, although wet prairies areas are being spared land clearing operations, their long-term preservation is now starting to be considered. The few relevant research results which are available for long-term preservation management of wet prairies[4] have yet to be fully utilized for the information they contain. From observations of wet prairie survival at Piper's Landing, several suggestions can be made to enhance long-term stability in wet prairie plant species associations.

The preservation of upland fringe[5] around the desired wet prairie can be a very effective approach to preventing undesirable plants which quickly establish in disturbed areas. The elimination of disturbed edges around wet prairies would prevent much willow primrose and cattail establishment. Upland fringe preservation is, however, only one factor in successful preservation.

The maintenance of the historical hydroperiod in the wet prairie is probably the most critical factor in maintaining the wet prairie plant species assemblage from year to year. Permitted water table changes from pre-development levels may either be raised or lowered after development. In those instances where the water table is permanently raised, prairies on site may lose their previous long term plant species assemblage by die-out and replacement with cattail and torpedo grass. This was the case of the entrance wet prairie at Piper's Landing.

Many wet prairies are utilized as part of the surfaces water management systems. With no changes in the pre-existing water table elevation, a preserved wet prairie can be put into a higher than normal hydroperiod by incorrectly setting a control elevation. Currently, control elevation for wet prairie preservation is set correctly at the middle of the _Hypericum fasciculatum_ elevation.[6] Assuming a 0.5-1.0 m drop in water table during the normal dry season, the hydroperiod would mimic nature to a high degree. Unfortunately, the hydroperiods of such wet prairies are greater than optional through a combination of changed water table, greater post development water volume inputs, locally variable precipitation and unseasonal precipitation patterns. These factors combined with a set control elevation outlet can result in flooded wet prairie during the typical dry periods so essential for seed germination and perennial survival. It is recommended that the appropriate authorities consider the option of lowering the control elevation during the normally dry season sufficient to mimic existing wet prairies. This option would only be exercised when the wet prairie is flooded throughout the normal dry season and discharge is within permitted limits. This one change in surface water management would be a major factor in long-term perservation of wet prairies with their native plant assemblage.

In conclusion, wet prairies can be integrated into developments as part of the natural landscape but planning and engineering must design the water levels and duration to more closely mimic nature. Drowned wet prairies can be rejuvenated into healthy, aesthetically pleasing and cost-effective marshes with little to no use of herbicides or algacides in maintenance.

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LITERATURE CITED

1. Cowardin, L.M., et. al.; Classification of Wetland and Deep Water Habitats of the United States. Department of Interior. 1979. Publication No. FWS/OBS-79131

2. Kale, H.W. II; Birds: Volume 2, In Rare and Endangered Biota of Florida. Peter H. Pritchard, ed., University Presses of Florida, 1978.

3. Wetzel, R.G., Limnology, 1st Ed. Sauders College Publishing, 1975.

4. Goodrick, Robert L., The Wet Prairies of the Northern Everglades, in "Environments of South Florida: Present and Past". Patrick Gleason, ed. Memoir #2, Miami's Geological Society. 1974, p. 47-52.

5. Shannon, S., et. al., Staff Assessment of Harbour Ridge, Ltd., DRI, Treasure Coast Regional Planning Council, Stuart, Florida, 1982.

6. Mr. Bill Cary, P.E. and with South Florida Water Management Council Staff, personal communication.

CREATION OF WETLANDS
IN A XERIC COMMUNITY

J. Steve Godley
Richard J. Callahan, Jr.
Biological Research Associates, Inc.
4010 West Cayuga
Post Office Box 15386
Tampa, Florida 33684

ABSTRACT

A wetlands greenbelt suitable for stormwater retention was created in the Beacon Woods development near Hudson, in Pasco County, Florida. The project was unique because the site was located in exceedingly well-drained sandhill habitats. Landscape design included spreading a clay liner over the bottom of the designated wetland depressions to insure water retention capabilities, and covering this confining layer with organic soils to provide a substratum suitable for wetland species. Both wetland and xeric-adapted, upland species were planted in beds simulating natural community composition. Only native species were used, many of which had never been planted on a commercial scale.

INTRODUCTION

In 1982 Beacon Homes of Florida, Inc. contracted Post, Buckley, Schuh and Jernigan, Inc. (PBSJ) to design a greenbelt suitable for stormwater retention within their Beacon Woods development near Hudson, in Pasco County, Florida. PBSJ in consultation with Central Florida Native Flora, Inc. (CFNF) and Biological Research Associates, Inc. (BRA) developed a plan for creating an urban landscape composed entirely of native flora within this water retention area.

THE PROBLEM

The Beacon Woods site is located in the most xeric habitats of Florida: sand pine scrub and pine-turkey oak communities. Soils consist of nutrient poor, exceedingly well-drained Paola and Candler fine sands. Porous, karst limestone outcroppings are numerous beneath the deep sands and water levels currently average 3.1 m below the surface.

To meet 100-year flood protection capabilities PBSJ designed a series of five, interconnected drainage basins that meandered through the Beacon Woods development. These basins were staged to prevent localized flooding and each received stormwater runoff from 3 to 5 culverts. The drainage system totaled 1.5 kilometers in length.

Drainage basins such as designed for Beacon Woods typically are mulched and seeded to produce a permanent stand of grass. Although inexpensive to install, these monotonous grasslands are costly over the long-term, requiring continual mowing, fertilization, and maintenance.

To produce a cost-effective yet aesthetically pleasing and biologically functional greenbelt, the developer agreed to plant this open space using only native flora, which is naturally adapted to the climate, soils, and pests of Florida.

The location and design of the Beacon Woods drainage system severely constrained the types of plants that could be installed successfully. Removal of topsoil during excavation of the basin exposed soils of extremely poor quality in a habitat where nutrients are limiting. In addition, the narrow, often steep slopes of the drainageway required the selection and placement of species tolerant of the predicted moisture gradient. Finally, many of the species, especially xeric upland forms, had never been grown or transplanted on a commercial scale and the nursery technology for these species practically was non-existant.

GREENBELT DESIGN

The initial section to be planted was 409 m in length, averaged 45.5 m in width, and totaled 2.31 ha (Figure 1). To produce an aesthetically pleasing landscape the drainage basin meandered through the open space corridor. Patches of existing sandhill vegetation were left intact in three locations. Slopes of the drainage basins were of variable pitch with intervening plateaus in several areas. Limestone boulders uncovered during earthwork were left exposed or moved to form distinct outcroppings. Both wetland and xeric-adapted, upland species were to be planted in beds which simulated natural community composition.

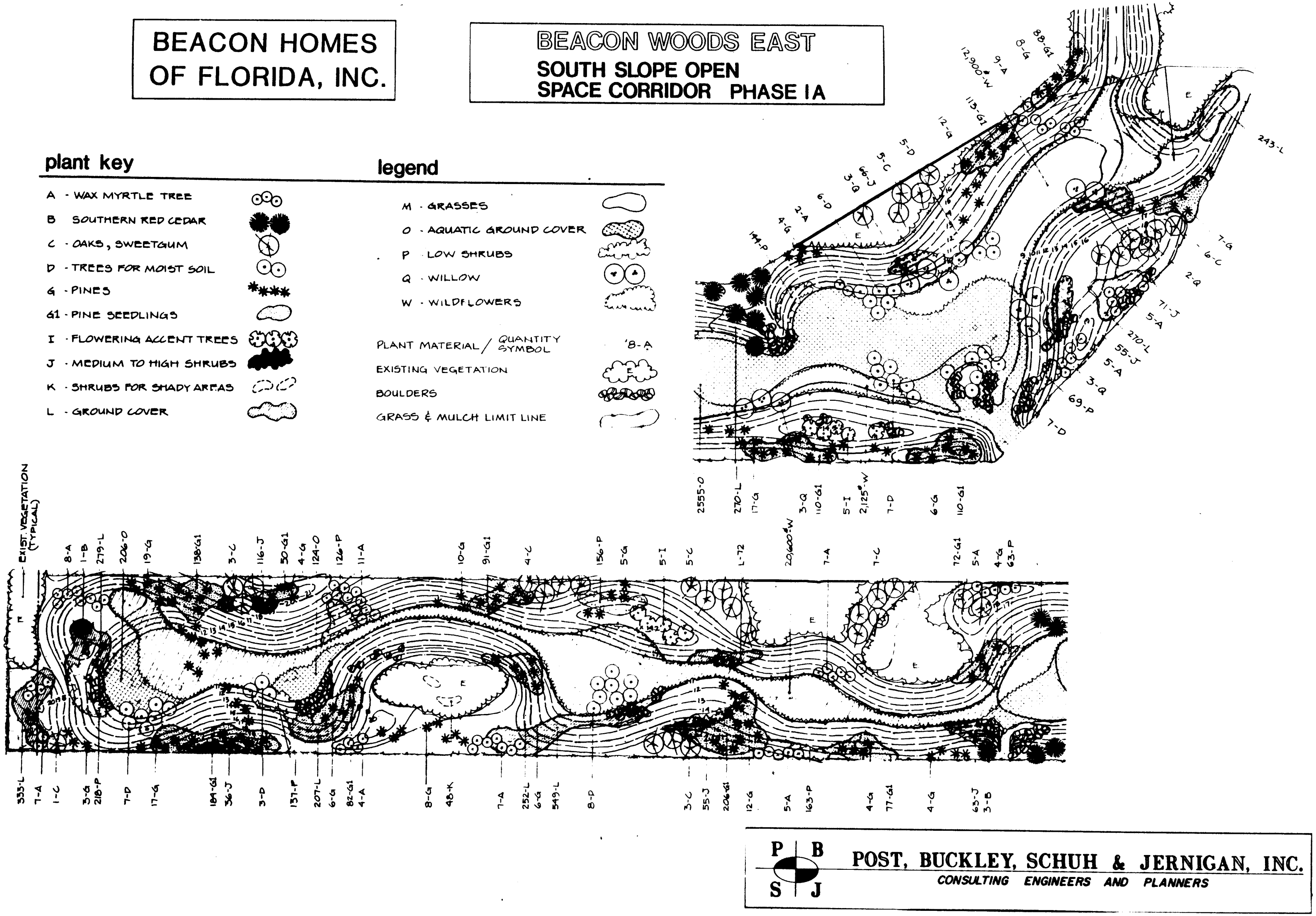

Figure 1. Greenbelt design of the Beacon Woods site.

Three distinct wetlands of 0.02 ha, 0.02 ha, and 0.21 ha were created in this basin. Each was strategically located near culvert outfalls to insure an adequate water supply. The basins were excavated 2.1 m below existing ground level, then undercut an additional 30 cm to form the wetlands. Ground water levels were approximately 1 m below this depth. To insure water retention capabilities in the porous sands of the site, a clay liner was spread over the designated wetland depressions (Figure 2). This confining layer (American Association of State Highway Officials Type A-1 clay) was compacted with heavy machinery to a minimum depth of 15 cm and also spread on all sides of the depressions to seal the edges against leakage of water. To provide a suitable substratum and an additional propagule source for wetland plants, a 15-20 cm layer of peat/muck mixture was then spread over the clay liner.

Approximately 8,610 individuals of 48 species were planted at the Beacon Woods site between January and April 1983. Many plants, especially upland species, were transplanted from the Beacon Woods tract which contains 1200 acres, most of which is scheduled for development. Other plants were nursery grown or taken from powerline right-of-ways subject to clearing. Thus, the effect of the project on native plant populations was minimal.

Table 1 presents the approximate number of individuals of each species planted at the Beacon Woods site and mean survivorship estimates of these plants six months after planting. Terrestrial species are included in this table because most consulting ecologists have little experience planting these forms and because the integration of upland and wetland plant communities is expected to become a more common procedure in the future.

FIGURE 2

Installation of the clay liner for creating a marsh at Beacon Woods. Note the barren slopes and sandy soils. Photo taken March, 1983.

FIGURE 3

Same view as Figure 2; photo taken 5 August, 1983.

Table 1. Species composition and survivorship estimates for plants installed at the Beacon Woods site, Pasco County, Florida (t = transplanted, c = container grown)

Species	N	% Survivorship	Method of Procurement
UPLANDS			
Upland Trees			
Sand pine (*Pinus clausa*)	15	73	t
Slash pine (*Pinus elliottii*)	126	88	c/t
Longleaf pine (*Pinus palustris*)	15	47	t
Blue-jack oak (*Quercus incana*)	6	100	t
Turkey oak (*Quercus laevis*)	8	100	t
Live oak (*Quercus virginiana*)	16	75	t
Wax myrtle (*Myrica cerifera*)	75	83	t
Red cedar (*Juniperus silicicola*)	13	100	t
Parkinsonia (*Parkinsonia aculeata*)	5	100	c
Chickasaw plum (*Prunus angustifolia*)	5	100	t
Pine Seedlings			
Slash pine (*Pinus elliottii*)	1321	60	t
Medium to High Shrubs			
Rusty lyonia (*Lyonia ferrugenia*)	72	94	t
Gallberry (*Ilex glabra*)	44	80	t
Saltbush (*Baccharis halimifolia*)	101	91	t
Wax myrtle (*Myrica cerifera*)	123	93	t
Firebush (*Hamelia patens*)	64	67	c
Garberia (*Garberia heterophylla*)	58	24	t
Low Shrubs			
Rosemary (*Ceratiola ericoides*)	190	13	t
Saw palmetto (*Serenoa repens*)	185	91	t
Adams needle (*Yucca filamentosa*)	250	93	t
Coral bean (*Erythrina herbacea*)	65	75	c
Shrub verbena (*Lantana camara*)	75	78	t
Shady Area Shrubs			
Beautybush (*Callicarpa americana*)	24	88	c
Small viburnum (*Viburnum obovatum*)	6	50	c
Coontie (*Zamia pumila*)	18	78	c
Ground Cover			
Beach sunflower (*Helianthus debilis*)	333	85	c
Blackberry (Native) (*Rubus sp.*)	212	85	t
Prickly pear cactus (*Opuntia humifusa*)	468	95	t

Species	N	% Survivorship	Method of Procurement
Shiny Blueberry (*Vaccinium myrsinites*)	250	15	t
Elephant's Foot (*Elephantopus tomentosus*)	260	85	t
Coontie (*Zamia pumila*)	74	82	t
Passion Flower (*Passiflora incarnata*)	250	80	t
Pennyroyal (*Piloblephis rigida*)	75	40	t
Phoebanthus (*Phoebanthus grandiflorus*)	68	85	t
Tephrosia (*Tephrosia spicata*)	114	70	t
Greeneyes (*Berlandiera subacaulis*)	315	95	t
WETLANDS			
Trees			
Pond cypress (*Taxodium ascendens*)	13	100	c
Red maple (*Acer rubrum*)	13	100	c
Cabbage palm (*Sabal palmetto*)	7	71	t
Dahoon holly (*Ilex cassine*)	5	80	c/t
Sweet bay (*Magnolia virginiana*)	9	100	
Carolina willow (*Salix caroliniana*)	7	100	t
Aquatic Plants			
Pickerelweed (*Pontederia cordata*)	600	95	t
Soft rush (*Juncus effusus*)	600	95	t
Cordgrass (*Spartina bakeri*)	400	90	t
Spatterdock (*Nuphar luteum*)	75	95	t
Prairie iris (*Iris hexagona*)	275	90	t
Arrowhead (*Sagittaria lanceolata*)	450	90	t
Sawgrass (*Cladium jamaicense*)	20	70	t
Fragrant water lily (*Nymphaea odorata*)	75	95	t
Spiderwort (*Tradescantia ohiensis*)	150	85	t

Upland Species

The terrestrial plants that were installed can be divided into eight broad categories: trees, medium to high shrubs, low shrubs, shrubs for shady areas, pine seedlings, ground cover, wildflowers, and grasses. In general, mean survivorship of upland forms (76.6%) was lower than that of wetland species (90.4%). Considerable interspecific variation in survivorship also existed among upland species, and between planting methods for these species.

To insure an adequate moisture regime for upland species, a 2.5 cm PVC pipe watering system was installed. Plants were watered about twice a week until the initiation of summer rains in June. The holes excavated for most species were backfilled with a high-grade organic top soil supplemented with 6-6-6 fertilizer. All planted beds were mulched with cypress chips to retain soil moisture and to impede the growth of competing weeds.

With the exception of the parkinsonia and some slash pine, which were container grown, all trees (1.2-4.2 m) were dug with a 61 cm tree spade, then wire balled and burlapped before installation. Survivorship among the oak species was lowest for live oak; most individuals of this species were taken from rather mesic habitats and showed greater stress than the turkey oak or blue-jack oak which are adapted to xeric situations. Also, many of the live oaks were transplanted while in leaf, the other oak species during dormancy. Among wax myrtles (2.0 to 3.5 m in height) survivorship was greatest for smaller individuals and for those planted on lower slopes nearest the wetland depressions. This latter effect was expected since this shrub often is common in wetland transitional zones.

Among pine tree species, survivorship was highest in the slash pines and sand pines; the long taproots of longleaf pine, which often had to be severed during tree-spading, made it more difficult to transplant. About 60% of the bare root pine seedlings survived.

The high to medium shrubs (0.6 - 1.4 m) included six species, two of which (garberia and rusty lyonia) are characteristic of xeric habitats. With the exception of the container-grown firebush, all other shrubs were dug with shovels, burlap bagged and planted with burlap in place to avoid damage to root systems. Saltbush, wax myrtle, and firebush proved to be the easiest to transplant successfully (Table 1). All are characteristic of moderately fertile soils and responded well to the addition of topsoil and fertilizer. The other three species were planted in replicate beds; one bed of each was backfilled with topsoil and fertilizer, the other beds received only a fertilizer supplement. The topsoil treatment apparently enhanced the survivorship of gallberry (86% vs 74%) but not that of garberia (16% vs 32%) or rusty lyonia (58% vs 84%). We suspect that the two xeric-adapted species could not tolerate the changes in microbial root populations brought about by the addition of topsoil.

Six species of low shrubs (0.3 - 1.0 m) were installed (Table 1). All were transplanted except eastern coral bean which was nursery grown. Rosemary proved the most difficult to transplant with slightly higher success in beds planted in January (16%) than in early (11%) or late (8%) February. Saw palmetto was obtained by two methods: cutting off and digging up 0.7 to 1.3 m apical sections of mature plants, and digging up entire young individuals whose stems had not yet branched.

Survivorship was higher among younger plants (90%) than among mature individuals (60%). Some young specimens also set fruit during the first summer. Survivorship among all other low scrubs was high, although most Adams needle initially dropped their leaves.

Shady area shrubs were planted in beds in existing sandhill vegetation. Survivorship was lowest for the viburnum even though this species (and beautybush) were container grown. Apparently the viburnum requires a more mesic setting.

Eleven perennials adapted to well-drained upland habitats were transplanted from the wild to provide ground cover. All showed high to moderate survivorship except shiny blueberry and pennyroyal. Excavation of blueberry revealed that many beds are composed of a single individual vegetatively connected by very brittle rhizomes with few roots. We suspect that most pennyroyal died because they were not adequately watered after transplanting.

Slopes not planted in beds or with trees were seeded initially in January with winter rye to stabilize the soils and prevent erosion. Later, in March, dry mulch and a mixture of 20 parts Bermuda seed and 80 parts Pensacola Bahia were cut-in to produce a permanent stand of grass on these slopes.

A commercially prepared wild flower mix (Pinto Brand) of 11 species advertised as adapted to Florida climates was seeded on 0.8 acres of the drainage basin in March. Although the seed mix produced a colorful variety of flowers during the first growing season, about five of the species did not germinate/mature and it remains to be seen if the other species will become established on the site.

Wetlands

A total of 15 wetland species (2,700 individuals) was planted at the Beacon Woods site (Table 1). Most plant material was installed in the created wetland depressions, but some trees (red maples and dahoon holly) were planted in groupings in other seasonally flooded sections of the drainage basin. Figure 3 shows a view of the largest wetland depression taken from the same angle as Figure 2, but 3.2 months after plant installation was completed. Figures 4-6 show close-ups of this same marsh at various stages of completion.

Wetland species were planted in a zone configuration to mimic natural marshes and to maximize survivorship. Along the marsh edges in areas expected to be flooded only seasonally, spiderwort and cordgrass were planted along with some red maple and dahoon holly. These species gave way to soft rush, prairie iris, and sawgrass, with arrowhead and pickerelweed and pond cypress planted in slightly deeper water. Fragrant water lily and spatterdock occupied the deepest, open water sections of the marsh.

Survivorship among wetland species was uniformly high. High survivorship was attributed to five factors: (1) adequate water-retention capabilities of the clay liner, (2) the addition of the muck/peat layer which provided an excellent growth medium (3) all herbaceous plants were transplanted the same day as they were dug, thus avoiding desiccation and severe stress, (4) entire plants rather than plugs were transplanted, and (5) most trees were container grown.

In addition to the 15 planted species, approximately 25 other wetland species germinated or resprouted from the organic layer that was spread over the clay liner. Among these volunteers the most notable were

FIGURE 4

Close-up of wetland depression at Beacon Woods on 24 January, 1983, before installation of the clay liner. The rainwater disappeared within two days.

FIGURE 5

Same view as Figure 4 taken on 6 April, 1983, after the clay liner, organics and approximately one-third of plants had been installed.

FIGURE 6

Final product. Photo taken 5 August, 1983 smartweed *(Polygonum hydropiperoides)*, flat sedge *(Cyperus odoratus)*, coastal arrowhead *(Sagittaria graminea)*, primrose willow *(Ludwigia peruviana)*, bacopa *(Bacopa monnieri)* and pennywort *(Hydrocotyle umbellata)*. Cattails *(Typha sp.)* were not among the colonizing species, presumably because (1) cattails did not occur at the site where the organics were mined, (2) this organic layer was covered with water shortly after installation and air-borne seed had little chance to germinate, and (3) the dense stand of planted and colonizing species that developed precluded the establishment of any cattail seed that germinated.

DISCUSSION

Stormwater treatment facilities and flood detention areas are becoming a conspicuous component of Florida's urbanized landscape. Typically, these necessary structures are a public eyesore and a developers' nightmare, requiring continual expensive maintenance to control noxious mats of filamentous algae or monocultures of cattails.

The Beacon Woods project has shown that stormwater retention basins can be an amenity to the community, providing an attractive, functional greenbelt within a sea of concrete. The integrated use of terrestrial and wetland plants in a naturalistic setting serves to enhance the public's appreciation of our native flora and to provide critical habitat for wildlife in urban areas. For example, at Beacon Woods visitors often are given "tours" of the greenbelt by local residents, and several species of frogs *(Bufo terrestris, Hyla femoralis, Hyla gratiosa)* bred in the newly created wetlands the first summer.

The creation of wetlands in the sand pine scrub and turkey oak communities of the Beacon Woods site initially appears to have been a successful venture. The clay liner served to retain most of the water entering the catchment basin from street culverts. However, once water levels rose above the sides of this confining layer lateral seepage into the surrounding sandy soils occurred rapidly, preventing water depth in the center of the marshes from exceeding 0.5 m. As a result these marshes remain relatively shallow and experience comparatively large fluctuations in water level. Also, rates of evapotranspiration are expected to be high. All these factors will tend to destabilize the marsh systems and promote successional change.

This system of water retention and loss is radically different from natural marshes. Most small, natural marshes in Florida and those near Beacon Woods are surrounded by pine flatwoods. This habitat typically contains a shallow organic hardpan which impedes the vertical movement of water through the soil (Laessle 1942, Edmisten 1962). Thus, natural marshes receive water from two major sources: a rapid pulse of surface runoff, and a more sustained flow of lateral subsurface seepage which tends to reduce oscillations in water level. A major challange for the future will be creating more stable wetland habitats, possibly by mimicking these natural water input conditions.

ACKNOWLEDGEMENTS

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LITERATURE CITED

Edmisten, J.A. The ecology of the Florida pine flatwoods. Ph. D. Dissertation, University of Florida, Gainesville. 1963.

Laessle, M.A. The plant communities of the Welaka area. Gainesville: University of Florida Press, Biological Science Series IV (1):1-143; 1942.

SURVIVAL AND GROWTH OF RED MANGROVES
(*Rhizophora mangle* L.)
PLANTED UPON MARL SHORELINES IN THE FLORIDA KEYS
(A FIVE YEAR STUDY)

by

Harold W. Goforth, Jr.
Bioengineering Branch
Biosciences Department
Naval Ocean Systems Center
San Diego, California 92152

and

Mark Williams
Florida Keys Community Hospital
Key West, Florida 33040

ABSTRACT

Planting mangroves for stabilizing man-made marl shorelines in the Florida Keys is an environmentally preferred alternative to seawall and riprap construction. In June 1977, 126 red mangroves were planted along three marl shorelines to determine the relative significance of: (a) tidal height (+0.3 m and 0.0 m), (b) degree of exposure to erosion forces (protected, exposed and partially protected), and (c) plant size-- propagule (type A), 1 yr seedling (type B) and 2 to 3 yr old small trees (type C)-- upon growth and survival. Sphagnum peat moss and seagrass detritus ("wrack") were tested as organic amendments to the marl substrate. Planting procedures have been previously reported (Goforth and Thomas, 1979). (1) After 5 years, plant survival for type A, B and C plants on the exposed shore was 0, 0 and 79%; the protected shore 64, 43 and 75%; the partially protected shore 43, 64 and 93%, respectively. (2) Height of type B plants was greater on the partially protected shore (66 cm) than on the protected shore (53 cm). (3) Height of type C plants at the protected, exposed and partially protected shores was 73, 80 and 74 cm, respectively. (4) Combined plant survival was 68% at +0.3 m compared to 29% at 0.0 m tidal height, only 12% type A and B survived at 0.0 m. (5) Survival rates for the three plant types (A, B and C) were 36, 36 and 80% with an average vertical growth of 33, 28 and 20cm, respectively. (6) Representatives of all three plant types had matured and were fruiting, type C plants having the greatest at 19%. (7) The two organic amendments tested showed similar effects on survival, 54% for peat and 43% for seagrass "wrack".

INTRODUCTION

The environmental role of mangroves as a natural shoreline stabilizer has been an accepted principle of wetlands ecology for many years. In the Florida Keys a majority of the shoreline is composed of an organically depauperate, compacted calcitic marl (Wanless, 1974). Under the proper conditions nature has been successful in gradually establishing a protective fringe of mangroves and associated vegetation along these shores (Davis, 1940; Thom, 1967; Savage, 1972; Carlton, 1974; Teas et al., 1976). Man has been relatively successful in developing engineering alternatives for protecting shorelines from erosion forces. Engineering alternatives however, have failed to replace the biological role played by mangroves and other shoreline vegetation. Along shorelines created with marl fill or shores where mangrove communities have been destroyed, only the typical slow-growing, stunted mangrove seedlings are found. A variety of techniques have been attempted under a wide range of conditions to plant/transplant 13 of the 72 species mangroves (for a review see, Lewis 1981). Few studies however, have employed an experimental design which included a number of controlled variables combined with long term monitoring. Frequently, transplants are performed without the inclusion of controls or comparisons with natural revegetation rates. The crucial criterion of success must always be the degree of long term survival and growth and restablishment/creation of the desired ecological condition. The goal of this study was to test the effect of tidal height, erosion forces, and organic amendments upon the relative survival and growth of three developmental stages of red mangrove transplant stock (propagules, seedlings and small trees).

AREA DESCRIPTION

The transplant and control sites for this study are located on the campus of Florida Keys Community College (FKCC) on Stock Island, Key West, Florida. The transplant stock came from a donor site in a mangrove "swamp" located nearby on Raccoon Key. Figure 1 shows the location of these sites and the average annual wind velocity and frequency (Boylan, 1974). Site A faced south and was the protected shore experiencing limited erosion. Site B faced east and was exposed to the greatest degree and had experienced significant erosion by 1977. Site C faced north and was "partially protected" by a small mangrove island located 25 m to the north. Experimental Site C and control Site E were the only two shores experiencing equivalent exposure to erosion forces. Both control shores, sites D and E were created at the same time (1967) as the experimental transplant sites from marl fill. All other sites were characterized by varying degrees of exposure and thus provided a basis for determining the relative survival and growth of the three types of transplant stock.

METHODS AND MATERIALS

Planting procedures involved the use of a hydraulic power auger to drill holes (0.41 m diameter and 0.45-0.61 m deep) in the packed marl shoreline. The loose marl removed from the augered holes was mixed 50:50 with one of two organic ammendment treatments, spahagnum peat or seagrass detritus ("wrack"), and placed in the hole with a mangrove. The details of the planting procedures used in this study have been previously reported (Goforth and Thomas, 1979). In June 1977, 126 red mangroves were planted along three marl shorelines (i.e., 42 on each shore) to test the

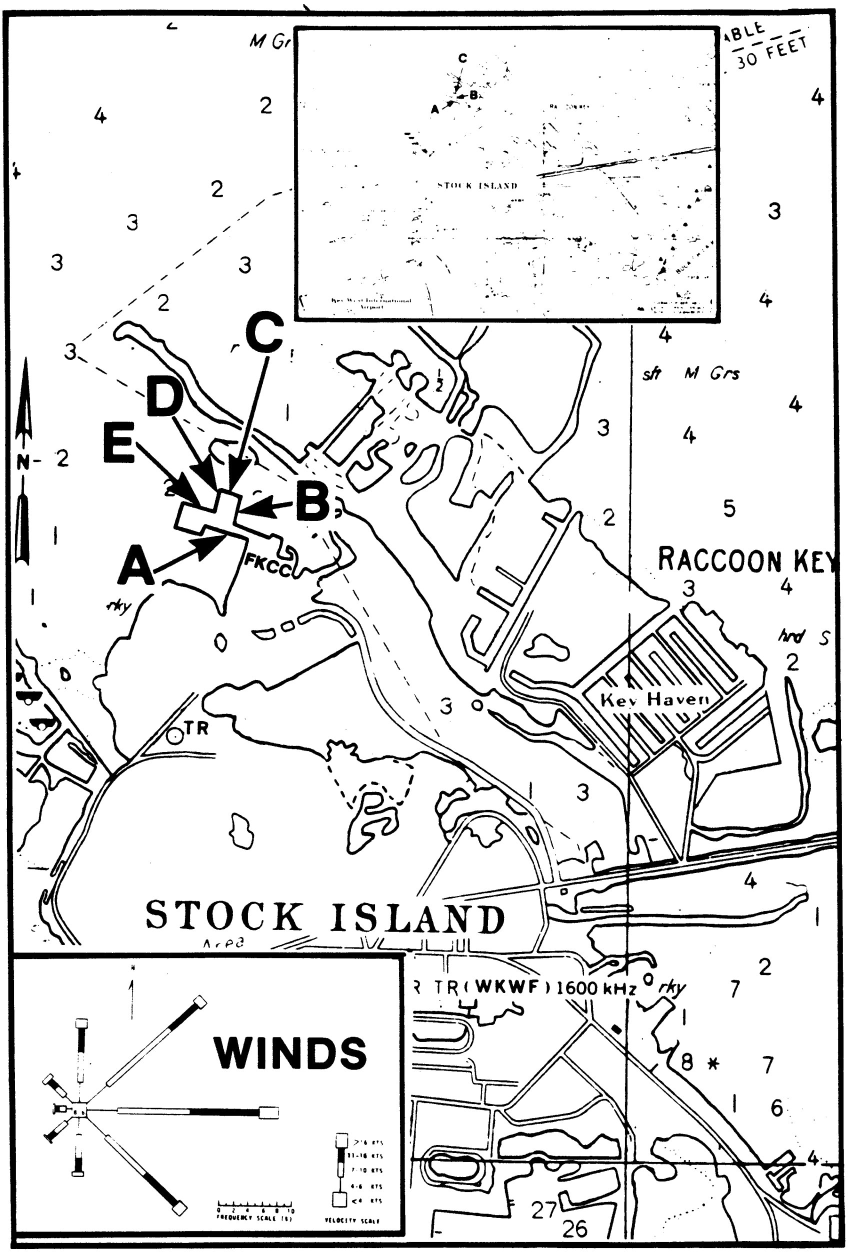


Figure 1. Location of transplant, control and donor sites at Key West, Fla.

effect of tidal height (+0.3 m and 0.0 m), degree of exposure (protected, exposed and partially protected), and plant size-- propagules (type A), 1- year-old seedlings (type B), and small 2 to 3-year-old trees (type C). This report is a result of a site survey conducted in October 1982. The purpose of this survey was to determine the long term (5-year) survival and growth of the experimental treatments for a comparison with adjacent control shores. Tree height measurements were made from the substrate to the tallest growth bud and recorded to nearest 0.5 cm. Prop root and leaf counts were also made but have not been included in the data analysis because of the difficulty obtaining accurate values and the large percentage of "too numerous to count" data points.

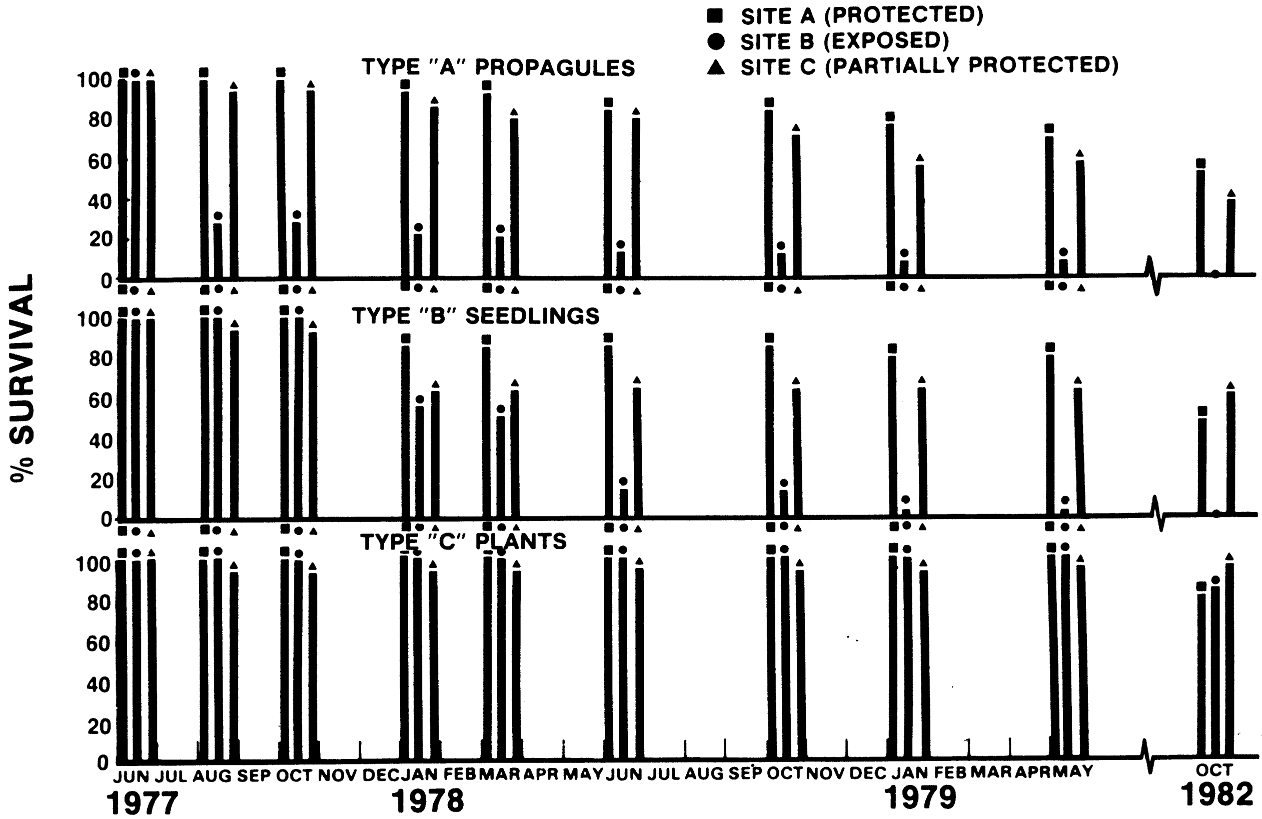


Figure 2. Survival of transplanted red mangroves (1977-1982).

RESULTS AND DISCUSSION

Propagules (type A)

Survival of propagules varied between sites depending on the degree of exposure (figure 2). The survival on the protected, exposed and partially protected sites was; 50, 0, and 43%, respectively. Propagule survival rates at the two tidal heights on the protected and partially protected shores were 75 and 83% at +0.3 m compared to 25 and 27% at 0.0 m. Since no propagules survived on the exposed shore, growth data for the last 3 years were confined to that from the protected and partially protected shores. Figure 3 shows a plot of propagule growth and survival during the 5 years since planting. These data indicate a significant initial loss of propagules on the exposed shore followed by a gradual reduction to zero. Growth rate of propagules during the first 2 years averaged 18.5 cm/yr but only 7.8 cm/yr during the next three years. The average height of propagules after 5 years was the same (59 cm) at both the protected and partially protected sites.

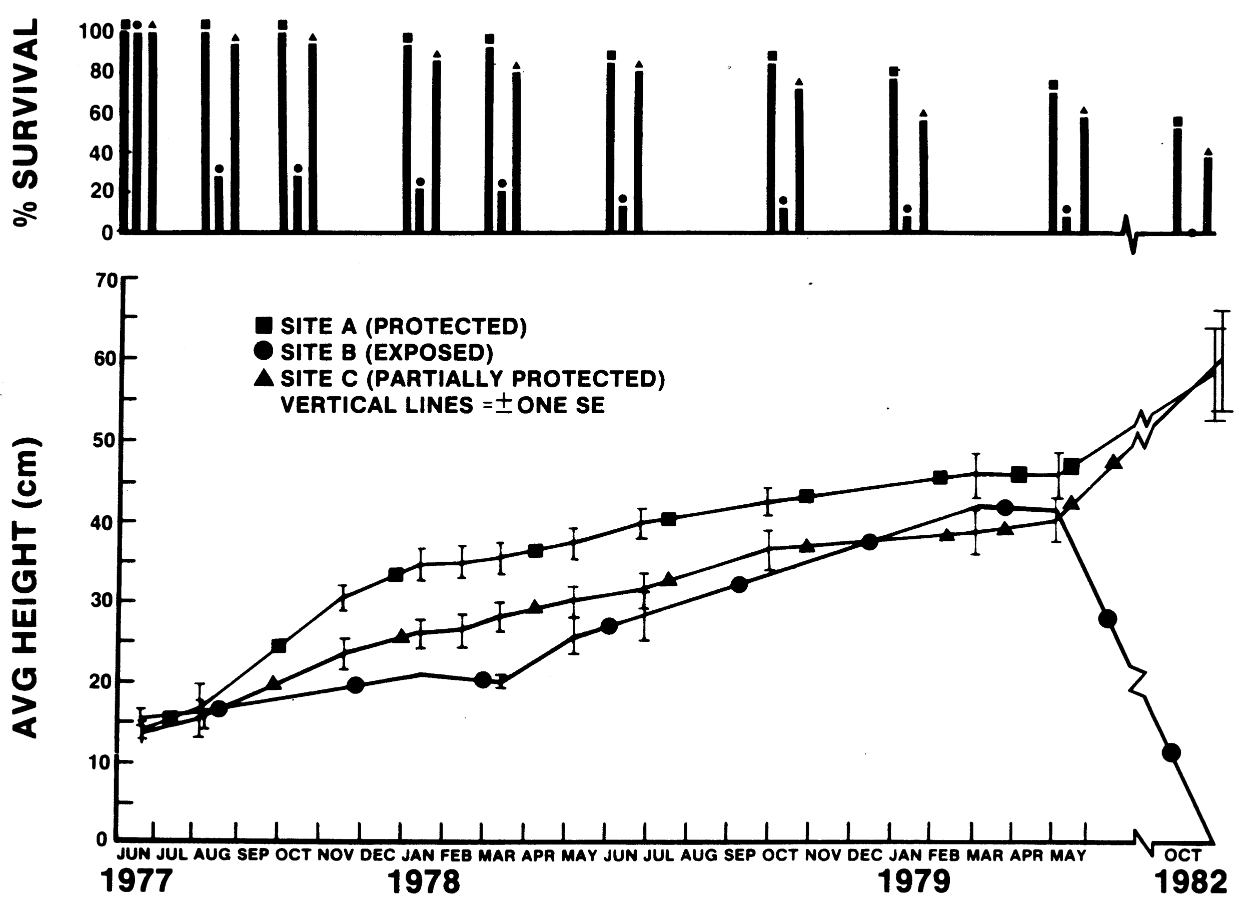


Figure 3. Survival and growth of red mangrove propagules (type A) (1977-1982).

Seedlings (type B)

Survival of seedlings after 5 years was 50, 0 and 64% for the protected, exposed and partially protected sites, respectively (figure 2). Seedling survival at the +0.3 m tidal level (100 and 78%) was significantly greater than at 0.0 m (0 and 22%) for the protected and partially protected sites, respectively. Seedling height and growth was greater on the partially protected shore (66 cm) than on the protected shore (53 cm) (figure 4). Vertical growth rates for seedlings during the 5 years was relatively steady at 3-4 cm/yr for the protected site, and 5-6 cm/yr for the partially protected site. The average height of seedlings after five years, was not significantly different from that of propagules (59 cm vs 61 cm). The total survival percentage for seedlings (36%) was also similar to that for propagules (36%). The main difference between survival of these two plant types on the exposed shore was the rate of loss. For the exposed site, the propagules had their greatest losses during the first 2-3 months, whereas seedling losses were delayed until the second year (figure 2). On all shores, the survival trends after 2 years were essentially identical for both propagules and seedlings: final survival statistic being the same after 5 years with 15 of 42 of each type plant surviving.

Small trees (type C)

Transplanted small trees exhibited the greatest survival at all sites and were the only surviving plants on the exposed shore. Survival for small trees was 79, 75 and 93% for the protected, exposed, and partially

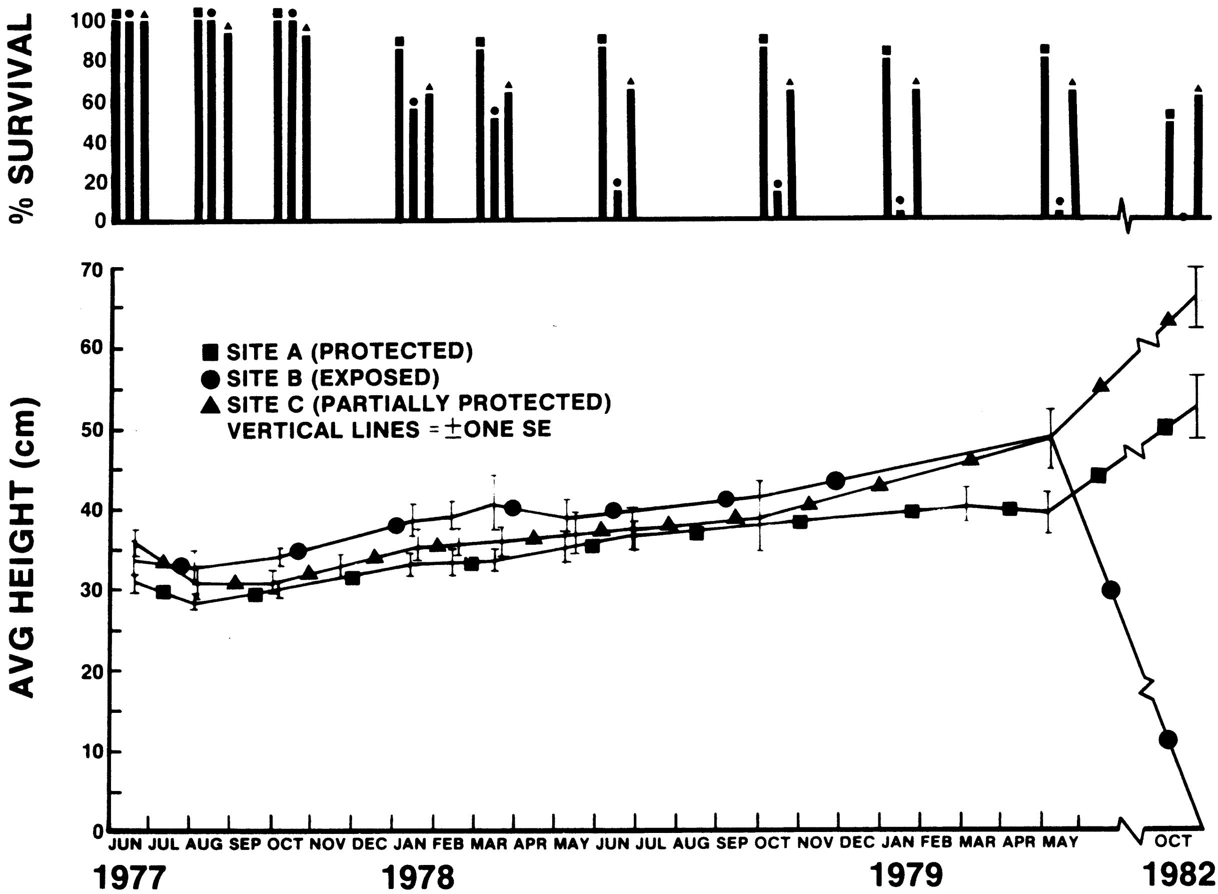


Figure 4. Survival and growth of red mangrove seedlings (type B) (1977-1982).

protected sites, respectively (figure 5). The average survival for all small trees was 80% with 34 of the original 42 plants remaining after 5 years. The greatest loss of small trees occurred during the last 3 years, decreasing from an overall survival of 98 to 80%. Growth of small trees during the first 2 years was directed towards lateral branching and prop root and leaf production. The small vertical growth (approx. 6 cm) that occurred during that time was difficult to quantify due to the settling of these plants into the substrate (see Pulver, 1976). Vertical growth during the last 3 years (1979-1982) however, was significant and averaged 6.5 cm/yr.

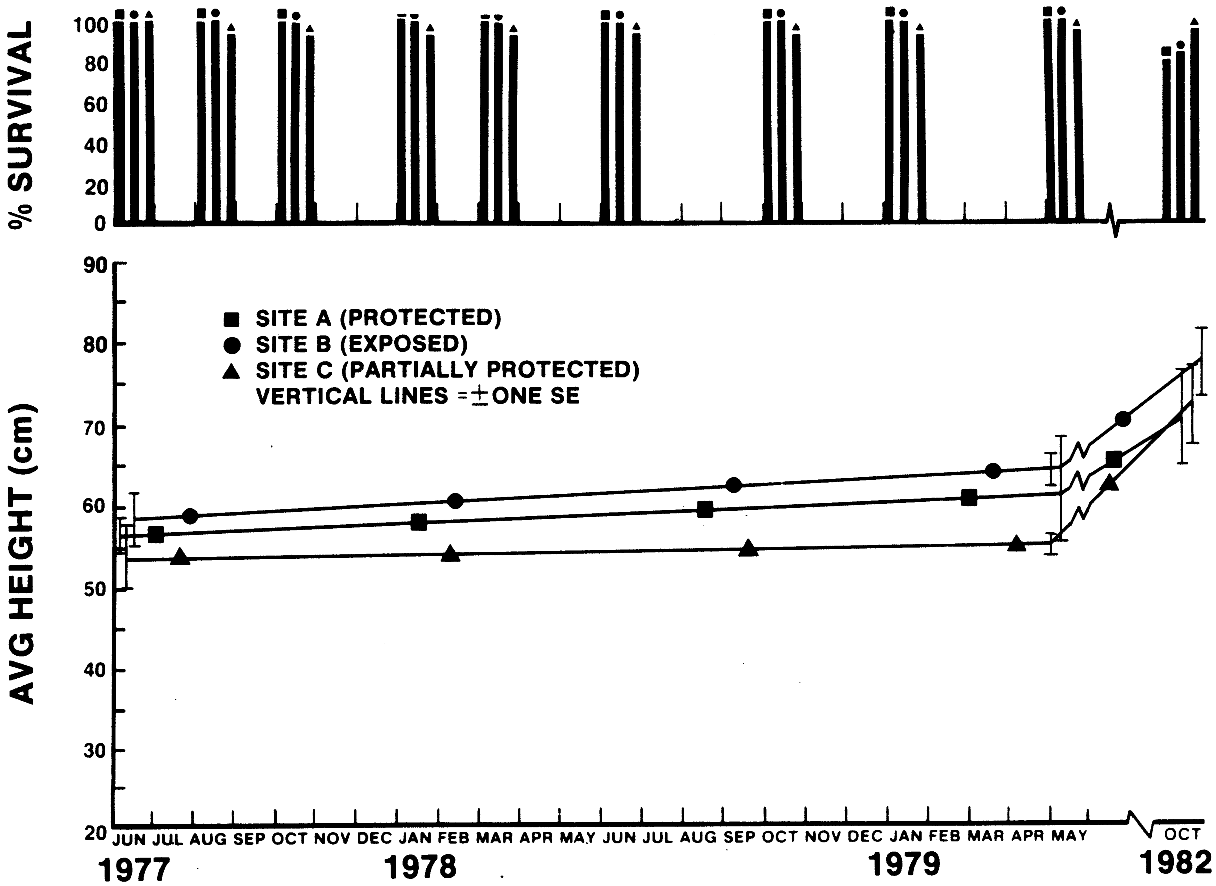


Figure 5. Survival and growth of red mangrove trees (type C) (1977-1982).

Lateral growth and prop root development of small trees continued during the last 3 years but were not quantified. The average height of all small trees was 76 cm representing an average vertical growth of 20 cm after 5 years (figure 6). Maturation and fruiting was evident in October 1982 with 19% of the small trees bearing fruits or flowers.

Figure 6. Typical growth pattern exhibited by type C plants on the exposed shoreline (site B) after 5 years (1977-1982).

Tidal Height and Degree of Exposure

Survival for all plant types was greater (68%) at the +0.3 m tidal height than at 0.0 m (29%) at all sites. Figures 7, 8, 9 and 10 show the conditions on the protected and exposed shores (sites A and B) before and 5 years after the transplant. The difference in mangrove community development at these sites is quite evident and may be explained by differences in rates of organic debris accumulation.

Figure 7. Condition of protected shoreline (site A) in 1977 (10 years after placement of marl).

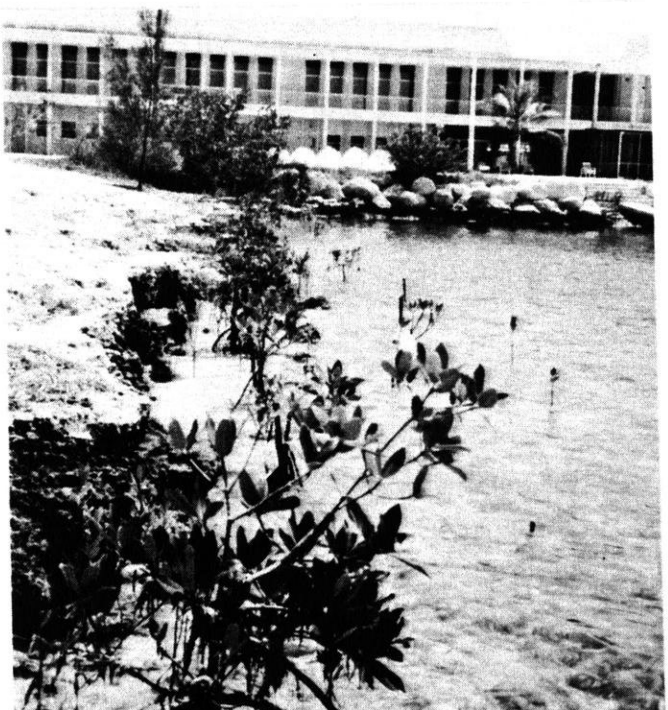


Figure 8. Condition of protected shoreline (site A) in 1982 (5 years after mangrove transplant).

Figure 9. Erroding, unvegetated condition of exposed shoreline (site B) in 1977 (10 years after placement of marl).

Figure 10. Revegetated condition of exposed shoreline (site B) in 1982 (5 years after mangrove transplant).

CONCLUSIONS

The results of this study indicate that partially protected shorelines may be planted with either propagules or 1-year-old seedlings with similar survival and growth rates. Exposed shorelines however, must be planted with small trees (at least 2 to 3 years old) to withstand erosion forces. Since all plants exhibited greater growth on the more exposed shores there appears to be a beneficial effect of accumulated seagrass and other organic detritus. However, quantitative data to support this contention were not collected in this study.

On exposed shorelines, strong winds affect the three types of transplanted mangroves differently. The shorter (avg height = 0.33 m) transplant stock (propagules and seedlings) suffered greater losses, possibly due to becoming buried under an accumulation of anaerobic debris similar to the conditions that occurred after hurricane Donna (Craighead 1971). The larger mangrove plants exhibited greater growth and survival possibly because they were taller (avg height = 0.56 m) and not buried; were better anchored (more prop roots); and they benefited from the accumulation of organic detritus around their bases. This supports the recommendation by Pulver (1976), that small trees (0.5-1.5 m tall) in lieu of seedlings, should be transplanted for rapid shoreline revegetation. The larger mangroves are more effective in capturing debris and establishing an organic mulch.

It appears that the unusually windy weather in the Keys during the past few winters may have been responsible for the observed decrease in survival of propagules and seedlings and the increase in growth of the larger trees. In sharp contrast to the transplanted shorelines, the natural vegetation of the control shores (sites D and E) was limited to a few small seedlings. Without the protection afforded by transplanted mangroves, these shores have continued to erode throughout the 5 years of this study.

Naturally occurring seedlings, like those on the control marl shores, are seen throughout the Florida Keys. However, they often die before maturing or exist in a stunted form for many years (Teas et al. 1976). The energy requirements for the initial establishment and growth of mangrove propagules on marl shorelines appear to be adequately met by storage products in the mature propagule or by nutrients on the substrate surface. Nutrients and/or energy requirements needed to support advanced growth of mangroves appear to be inadequate on most barren marl shorelines.

Tidal height of the transplant is an extremely critical factor in determining the survival of all three plant types, especially propagules and seedlings. Smaller mangrove plants, lacking prop roots and heavy foliage, are more susceptible to physical damage from drifting debris and are subjected to greater physiological stress due to greater relative submersion at the lower tidal level. Teas et al. 1976, describe how some Florida land developers have killed mangroves by altering the water levels to increase the percentage of prop root submergence. In this study and others (Lewis 1979, Lewis and Haines 1981) small differences in the tidal height of the experimental transplants produced significant differences in survival rates due to greater stresses at the lower tidal level.

The long-term success (i.e., survival, growth and maturation) demonstrated by this study supports the use of small trees and propagules of the red mangroves for the creation and restoration of shoreline vegetation. Using (1) a power auger to loosen the substrate, (2) organic amendments to augment available nutrients, and (3) stakes for anchoring transplants appears to provide the ingredients necessary for establishing mangrove communities on barren marl shorelines.

LITERATURE CITED

Boylan, R. L.. 1974. Local area forecaster's handbook for Naval Air Station, Key West, Florida. Naval Weather Service Environmental Detachment, U.S. Naval Air Station, Key West Florida. 52 pp.

Carlton, J. M. 1974. Land-building and stabilization by mangroves. Environ. Conserv. 1(4):285-294.

Craighead, F.C. 1971. The Trees of South Florida. Univ. Miami Press., 212 pp.

Davis, J. H. 1940. The ecology and geologic roles of mangroves in Florida. Carnegie Institution of Wash. Publ. 517. Papers from Tortugas Laboratory. 32:302-412.

Goforth, H.W. and J. Thomas 1979. Plantings of red mangroves (Rhizophora mangle L.) for stabilization of marl shorelines in the Florida Keys. In: Proc. Sixth Annual Conf. on Wetlands Restoration and Creation. D.P. Cole (Ed.), Hillsborough Community College, Tampa, Florida. pp. 208-230.

Lewis, R.R. III 1980. Large scale mangrove restoration on St. Croix, U.S. Virgin Islands, In: Proc. Sixth Annual Conf. on Wetlands Restoration and Creation. D.P. Cole (Ed.), Hillsborough Community College, Tampa, Florida. pp. 231-245.

Lewis, R. R. III 1981. Mangrove forests, In Creation and Restoration of Coastal Plant Communities. Lewis R. R. III. (Ed.), Pub. CRC Press Inc. Boca Raton, Florida. Chapter 8, pp. 153-171.

Lewis. R.R. III and K.C. Haines. 1981. Large scale mangrove restoration on St. Croix, Virgin Islands, In: Proc. Seventh Annual Conf. on Wetlands Restoration and Creation. D.P. Cole (Ed.), Hillsborough Community College, Tampa, Florida. p. 137.

Pulver, T. R. 1976. Transplant techniques for sapling mangrove trees, Rhizophora mangle, Laguncularia racemosa, and Avicennia germinans, in Florida. Florida Dept.of Natural Resources, Florida Marine Research Publications No. 22. 14 pp.

Savage, T. 1972. Florida mangroves as shoreline stabilizers. Florida Department of Natural Resources. Professional Papers Series No. 19. 46 pp.

Savage, T. 1978. The 1972 experimental mangrove planting an update with comments an continued research needs. In: Fifth Annual Conf. on Wetlands Restoration and Creation. D.P. Cole (Ed.), Hillsborough Community College, Tampa Florida. pp. 43-71.

Teas, H. J., H. R. Wanless, and R. Chardon. 1976. Effects of man on the shore vegetation of Biscayne Bay. In: Biscayne Bay: Past, Present and Future. A. Thorhaug (Ed.), Univ. of Miami Sea Grant Report No. 5. pp. 133-156.

Thom, B. G. 1967. Mangrove ecology and deltaic geomorphology: Tabasco, Mexico. J. of Ecology. 55:301-343.

Wanless, H. R. 1974. Mangrove sedimentation in geological perspective. In: Environments of south Florida: Present and Past. P. J. Gleason (Ed.), Miami Geol. Soc. Mem. 2. pp. 190-200.

WETLAND RESTORATION: A MODEL CHARACTERIZATION APPROACH USING AERIAL PHOTOGRAPHY AND FIELD ANALYSIS

Daniel J. Homblette

Boyle Engineering Corporation

320 East South Street

Orlando, Florida

32801

ABSTRACT

A restoration plan was prepared for a degraded shallow cattail marsh (steep man-made banks with limited transition zone, eroded shorelines, invasion of aggressive exotic plant species, limited wildlife diversity, extensive algal and submergent blooms during summer months) in Madison, Wisconsin, from site condition data collected using a model characterization approach. Five natural wetlands, or models, were studied in order to establish background conditions for comparison with the degraded site. Results of the study indicated that: (1) hydrologic conditions in the degraded wetland were characteristic of urban watersheds: extreme fluctuations in water level (20 centimeters) with generally high levels maintained over most of the year and virtually no periods of substrate exposure; (2) upland-wetland transition zone slopes in the degraded wetland were uncharacteristic of natural sites, ranging from 5-30% with an average of 12% while the models ranged from 0.1-5%; (3) soil profiles in the degraded wetland consisted primarily of varied depths of compacted fill over muck; (4) species diversity was highest in the disturbed site, while the vegetative cover/open water ratio was lowest (1:7). The high species diversity indicated good restoration potential, however, the extreme slopes combined with prolonged high water levels resulted in conditions that were not conducive to vegetation development beyond the very narrow transition zone.

INTRODUCTION

If restored or newly created wetlands are to function as an integral part of the landscape, it is important that the restoration process go beyond perfunctory measures (such as fill removal, coarse site grading, ditch plugging and water level manipulation for waterfowl enhancement) and incorporate ecologically based concepts leading to conditions characteristic of undisturbed ecosystems. Recently, the ecological approach has been pursued both for restoration (Swanson and Shuey, 1980; Gilbert, 1981) and in creation-oriented projects (Lahti, 1977; David, 1980). Site hydrology, topography, soil type, and vegetation are being studied as interconnected components of a complete system. This approach was incorporated into a restoration plan for a shallow cattail marsh located in Madison, Wisconsin, where five model wetlands were studied in 1981 in order to prepare a restoration plan. The site data were collected using traditional field methods, complemented by aerial photography that was used for mapping vegetation and documenting surrounding upland conditions.

While the methodology utilized in this study was applied to shallow cattail marshes in Wisconsin, it is applicable to all types of wetland related studies - both freshwater and coastal.

STUDY AREA

The wetlands studied in this project are located in south central Wisconsin in Dane County (Figure 1). The restoration site - the Class of 1918 Marsh - is located on the Univeristy of Wisconsin campus in Madison, with the five natural model wetlands scattered throughout the central part of the county.

The gently rolling terrain is of glacial origin and consists of end moraine, glacial outwash and scattered drumlins.

The Class of 1918 Marsh is a 17-acre shallow cattail marsh that was created in the early 1970's within a much larger wetland basin that had been drained for agriculture and later used as a dump for excess fill and construction debris from development on the University campus. Through efforts by University faculty and students, city residents, and campus planning, combined with funding from the Class of 1918, a shallow marsh was created in part of the original basin. The intent was to retain some of the original natural character of the area, improve wildlife habitat and provide an area for scientific research. Stormwater retention was incorporated into the site with the outlet consisting of a weir and pump to

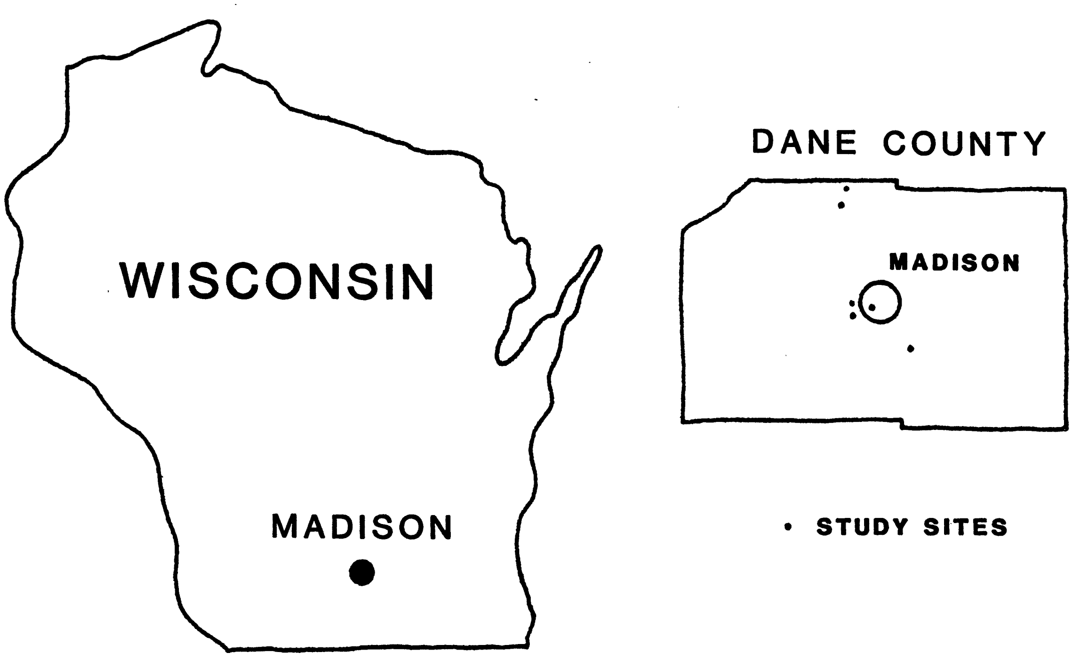


Figure 1: Study site locations

drain away excess flow. The model sites range in size from 23 to 35 acres (natural wetlands with minimal human impact and closer in size to the Class of 1918 Marsh were not found in Dane County) and are each located in shallow glacial depressions. Three of the sites - Kutz-Meek, Hahn-Jackson and Onsrud - are in rural areas with the uplands consisting of cropland and oak-hickory woods. The remaining two sites are located within the City of Middleton adjacent to or surrounded by residential development. While human activity has impacted each of the wetlands to some degree (limited ditching, stormwater input), they are much less impacted than the Class of 1918 Marsh and more closely represent natural conditions. Over 60 wetlands were surveyed in the area during the selection process. The five that were selected ranked highest when basin profile, size, hydrologic regime, vegetation composition and degree of impact factors were analyzed.

METHODS AND MATERIALS

Field Analysis

A review of available literature on wetland ecology, especially freshwater shallow and deep marshes, indicated that the primary site factors associated with natural vegetation diversity, wildlife habitat dynamics, and overall maintenance of a well balanced wetland ecosystem were basin shape, transition zone slope, water depth and fluctuation, and to a lesser degree soil texture and chemistry (Dane 1959, Dix and Smeins 1967, Walker and Coupland 1968, Whittaker 1967, Lahti 1977, Clambey and Landers 1978, Anderson et al. 1980, Weller 1981 and others). The site conditions studied in this project included soils, transition zone slope, hydrology and vegetation composition. Basin shape, also an important site parameter as noted above, was minimized as a factor in comparing the model sites to the Class of 1918 Marsh, by selecting natural wetlands with basins characteristic of shallow cattail marshes. Field conditions were documented along transects positioned perpendicular to the upland-wetland edge (across the environmental gradients) at randomly selected locations. The use of aerial photographs to complement the field data helped minimize the number of transects required at each site (5 in each model and 13 in the Class of 1918 Marsh). Transects were variable in length, with ultimate length determined in the field. Generally when open water or expansive monotypic stands of vegetation were encountered the transect was stopped.

Soil type and texture were documented along each transect using a hand-held probe to collect soil cores. The samples were taken at five meter intervals through uniform vegetation stands and on either side of locations where major vegetation composition changes occurred.

Elevation changes along each of the transects were obtained with a Dumpy Level and Philadelphia Rod at five meter intervals or more frequently across abrupt edges. For comparison purposes all edge slopes were calculated for a ten meter distance centered on the point where the surface water met the soil substrate on a particular date. Surface water elevations were established during the initial survey and fluctuations monitored during the study were referenced to stakes placed along each transect. Water levels in the Class of 1918 Marsh were measured at the outlet weir.

Vegetation composition and distribution were documented using a one-half by two meter frame quadrat. The sampling was started by placing the frame at the upland reference stake and each following sample was taken by moving the frame along the transect in two meter increments. Species presence was recorded, and the dominant species noted in each quadrat.

As a permanent record of each transect, photographs were taken from the upland end of the transect out into the wetland along the transect centerline. A 35mm camera was mounted on a four meter boom that was supported in a shoulder-waist harness at a forty-five degree angle. The camera frame at the end of the boom allowed for adjustment of the field-of-view, so that the entire transect could be contained in a single photo. Elevation of the camera eliminated the usual problem of an obstructed view at eye level due to tall vegetation. The ground photos were used extensively with the transect data and the aerial photographs.

Aerial Photography

Color and color infrared vertical aerial photographs were taken of each wetland in September. This time of year is generally considered favorable for taking aerial photos of freshwater wetlands for vegetation mapping because the highest percentage of species are nearest their peak in growth and most visible on the photos. A hand held system of two 35mm Pentax K1000 cameras was used to take the photographs from a Cessna 172 aircraft. To obtain the vertical orientation the pilot banked the plane over the center of the wetland at a predetermined flying height (880 meters) and at the full extent of the banking the photos were taken out of the open side window. Trigger grips allowed for simultaneous tripping of the camera shutters to obtain identical photos with two film types. A north orientation was maintained in each frame so that error resulting from the camera angle not being perfectly vertical was consistent in each photo. The cost of renting the pilot and plane was $130 for the two hour flight ($17/hr for the pilot, $48/hr for the plane).

RESULTS

It can be seen in the field data that the transect distances were highly variable both within each wetland and between wetlands (Table 1). As noted earlier each of the transects was started in the upland away from the wetland edge and proceeded into the wetland until open water or monotypic vegetation stands were encountered. In this way the upland-wetland edge was accurately sampled, and sufficient data was collected to characterize each wetland and provide ground verification for interpretation of the aerial photographs.

Soils

Soil conditions in the natural wetlands can generally be characterized as partially decomposed organic matter (usually 10-20cm) over organic muck (30-60cm) with the base substrate material consisting of sand and/or clay. Similar conditions originally existed in the vicinity of the Class of 1918 Marsh, however with the impacts of drainage and later filling, the soil conditions were significantly altered. The existing soil profile consists of clean fill over mixed fill that includes construction related debris, underlain by silt that has eroded from the steep edge slopes and partially decomposed organic matter. Muck deposited prior to disturbance of the

Table 1

Site Characteristics

Wetland	Size (Acres)	Transects #	Transects Length(m)	Soils	Edge Slope Range	Edge Slope Avg.	Vegetation[1] Transect	Vegetation[1] Site
Kutz-Meek	29	5	30-75	Muck/Sand and Clay	0-3%	1%	9-33	30/36
Hahn-Jackson	35	5	40-60	Muck/Sand and Clay	1-3%	2%	6-11	16/19
Tiedeman	21	5	20-45	Muck/Sand and Clay	1-4%	2%	9-14	18/29
Stricker	21	5	30-75	Muck/Clay	.1-5%	2%	3-13	12/17
Onsrud	28	5	25-100	Muck/Sand	1-2%	2%	23-32	34/45
Class of 1918 Marsh	17	13	10-30	Fill/Silt/Muck/Clay	5-30%	12%	8-24	34/60

[1] Range of species (lowest-highest) recorded along the transects and total species recorded at each wetland site (wetland species/total species)

original wetland, and clay form the lower layers.

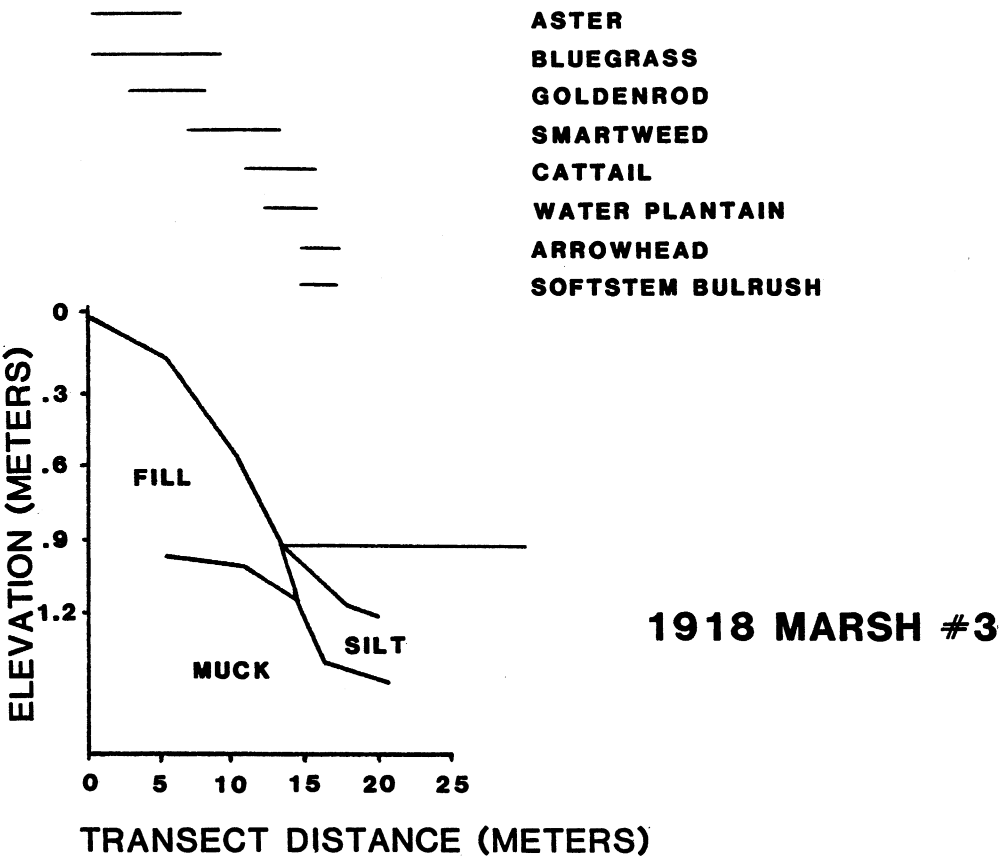
Upland-Wetland Edge Slopes

Edge slopes within the natural wetlands ranged from 0-5% with an average of 2% (Table 1), while more extreme conditions were recorded in the Class of 1918 Marsh. The range of 5-30% and an average of 12% are indicative of the method used to create the shallow basin - dredging. Vertical elevations from the water surface to the upland range from only .3m to 1.5m, however the extremely abrupt edge results in the steep edge slope. This factor combined with the lack of natural cyclical water fluctuations is the major reason why the degraded conditions exist in the Class of 1918 Marsh. In natural wetlands as water levels recede portions of the basin are often exposed providing a substrate for seed germination and/or shallower conditions where vegetative reproduction may occur. However in the Class of 1918 Marsh there is only a narrow perimeter where wetland vegetation exists and as water levels recede additional substrate is not exposed because of the steep slopes. Figure 2 is a comparison of two of the wetland edge profiles at a natural site (Hahn-Jackson) and in the Class of 1918 Marsh.

The data collected at all transects (scope and size limits of this paper preclude inclusion of the data) was recorded in the format shown in Figure 2. This allowed for a more direct comparison between individual transects and between site conditions in wetlands.

Hydrology

From a hydrologic standpoint the natural model wetlands can be charac-



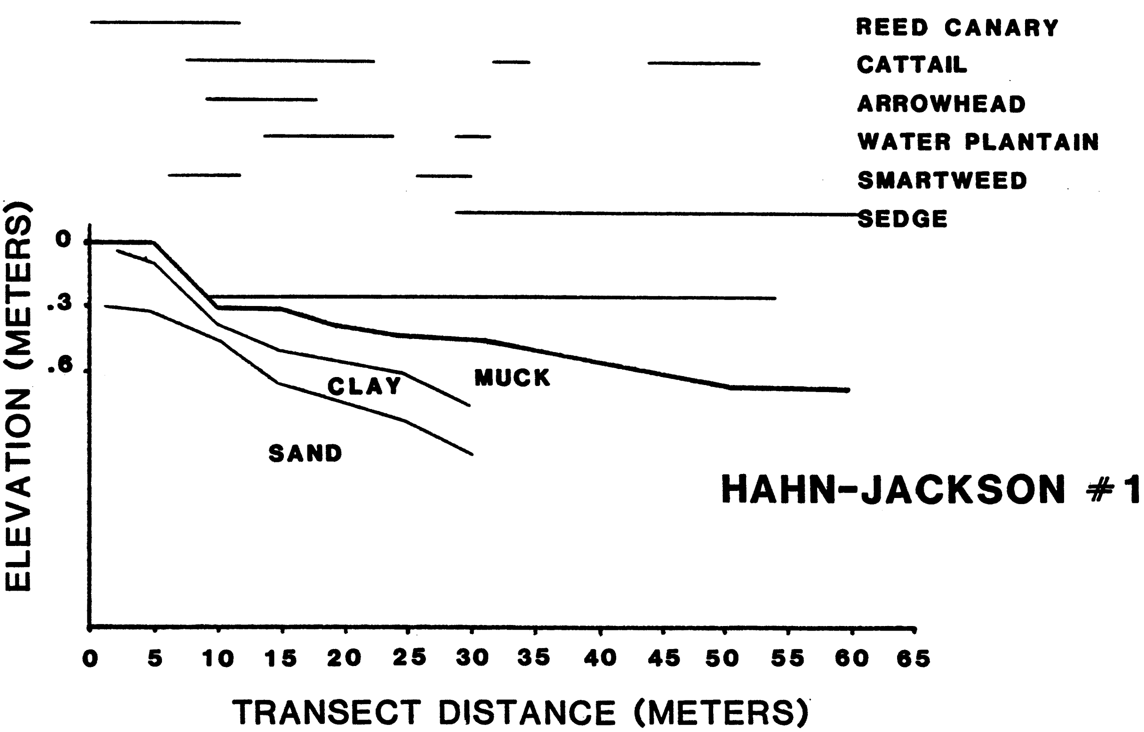


Figure 2: Transect profiles

terized as isolated shallow drainage depressions with no outlets. Inflow to the depressions is from upland runoff through shallow drainageways and sheetflow. Within the two urbanized sites the inflow is higher than normal levels because of upland development. Water levels are highest in the spring and gradually decrease into late summer, with only temporary shallow peaks due to the larger rainfall events. In contrast water levels are maintained artificially high in the Class of 1918 Marsh because of the weir at the outlet. In addition significant long term peaks occur from spring through fall because of direct stormwater input from adjacent developed sites. The average depth in the basin is 40cm with some storm peaks adding an additional 20cm in a matter of hours.

Vegetation

The numbers of wetland species sampled in the models and the Class of 1918 Marsh were highly variable, ranging from a low of 12 in Stricker's to a high of 34 in both Onsrud and the Class of 1918 Marsh (Table 1). The high number of species in the degraded site is one indication of the potential the site has for restoration. However as noted earlier the vegetation is very limited in areal extent forming only a narrow band around the perimeter that results in a vegetative cover/open water ratio of 1:7.

Restoration Plan

The background data gathered using the model characterization approach was incorporated into a restoration plan for the Class of 1918 Marsh. Visual and statistical comparison of the data substantiate the preliminary analyses that high water levels and steep edge slopes are the two major limiting factors at the degraded site. Of all the natural conditions documented throughout this study, those in Tiedeman's proved to be most useful in preparing the restoration plan. Tiedeman's, like the Class of 1918 Marsh is impacted by stormwater inflow, however it continues to function as a more ecologically balanced wetland because the edge slopes are relatively shallow (1-4% with an average of 2%). A 1:1 vegetative cover/open water ratio is maintained because even though higher water levels exist, the shallower slope allows for vegetative reproduction and during lower water periods, seed germination.

Briefly the restoration plan for the Class of 1918 Marsh consisted of a series of management activities phased over a five year period. The initial phases would consist of water level manipulation to more closely parallel normal cyclical fluctuations and edge slope grading to provide better substrate conditions for vegetation. Close monitoring of the wetland, including field measurements and periodic mapping using aerial photographs, would indicate if the two major modifications were successful. Subsequent steps would include vegetation restoration through plantings and/or seeding and weedy species maintenance.

CONCLUSIONS

The model characterization approach to documenting site conditions in wetland ecosystems was an effective method of gathering and organizing significant amounts of physical and biological data. The profile format of recording the information (Figure 2) proved very useful in comparing conditions within and between wetlands which is essential when preparing a restoration plan. The approach can be modified for many applications, and significantly reduced in scope for smaller projects.

Incorporation of aerial photography into the approach significantly reduced the amount of fieldwork necessary to accurately map vegetation in the wetlands. The method described, using hand-held 35mm cameras, was very cost effective, and resulted in products that are easy to use when enlarged to 5x7 prints and which form a permanent record for long term monitoring.

This paper is a brief summary of part of a more extensive study conducted as partial fulfillment of the requirements for a Master of Science Degree in Environmental Monitoring from the University of Wisconsin-Madison (1982).

LITERATURE CITED

Anderson, P.H.; Lefor,M.W.; and Kennard, W.C. "Forested Wetlands in Eastern Connecticut: Their Transition Zones and Delineation." Water Resources Bulletin, No. 2 (1980), pp. 248-255.

Clambey, G.K. and Landers, R.Q. "A Survey of Wetland Vegetation in North-Central Iowa." Proceedings of Fifth Midwest Prairie Conference. Ames, Iowa: Iowa State University, 1978.

Dane, C.W. "Succession of Aquatic Plants in Small Artificial Marshes in New York State." New York Fish and Game Journal 6 (January 1959): 57-76.

David, G. "Indian Lake Restoration Project." NatureScape (January-February 1980): 12.

Dix, R.L. and Smeins, F.E. "The Priarie Meadow, and Marsh Vegetation of Nelson County, North Dakota." Canadian Journal of Botany 45 (1967): 21-58.

Gilbert,T.; King, T.; and Barnett, B. "An Assessment of Wetland Habitat Establishment at a Central Florida Phosphate Mine Site." U.S. Fish and Wildlife Service, 1981. p. 96.

Lahti,T.C. "Restoration of a Small Southern Wisconsin Wetland." In Wetlands, Ecology, Values and Impacts: Proceedings of the Waubesa Conference on Wetlands. Edited by C. DeWitt and E. Soloway. Madison, WI: University of Wisconsin-Institute of Environmental Studies, 1977. pp. 136-163.

Swanson, L.J., Jr. and Shuey,A.G. "Freshwater Marsh Reclamation in West Central Florida." In Proceedings of the Seventh Annual Conference on Wetlands Restoration and Creation. Edited by D. Cole. Tampa, FL: Hillsborough Community College, 1980. pp. 51-61.

Walker, B.H. and Coupland, R.T. "An Analysis of Vegetation-Environment Relationships in Saskatchewan Sloughs." Canadian Journal of Botany 46 (1968): 509-522.

Weller, M.W. Freshwater Marshes: Ecology and Wildlife Management. Minneapolis,MN: University of Minnesota Press, 1981.

Whittaker, R.H. "Gradient Analysis of Vegetation." Biological Review 42 (1967): 207-264.

LONG-TERM MONITORING OF THE APALACHICOLA BAY WETLAND HABITAT DEVELOPMENT SITE

CHARLES J. NEWLING

MARY C. LANDIN

STEPHEN D. PARRIS

Environmental Laboratory
U. S. Army Engineer Waterways Experiment Station
P. O. Box 631
Vicksburg, MS 39180

Abstract: The Apalachicola Bay wetland habitat development site was created by hydraulic deposition of silty dredged material over existing coarse-grained material. In 1976, intertidal and supratidal zones were planted with smooth cordgrass (*Spartina alterniflora*) and saltmeadow cordgrass (*S. patens*). From 1980 through 1982, annual visits were made to the area to observe the physical condition, plant community, and wildife use at the habitat development site and at three nearby natural marshes used as reference sites. The results indicate that the habitat development site was highly successful because: (1) the vigorous marsh plant community that dominates the site is comparable to those at the natural reference marshes, and (2) the site receives especially heavy wildlife use by numerous bird species, particularly waterbirds and clapper rails (*Rallus longirostris*). Wind-driven waves from a long south fetch are eroding the main containment dike on the south side of the site, but the habitat development marsh itself is still intact and is biologically productive.

INTRODUCTION

As part of Dredged Material Research Program (DMRP) conducted between 1973 and 1978, the Apalachicola Bay wetland development site was one of seven wetland and upland habitat development field sites established and studied at selected, suitable locations in United States waterways where dredged material had been deposited. These sites were built for the purpose of demonstrating the feasibility of using dredged material for the establishment of productive plant and animal habitats. Because of the questions raised concerning plant and animal succession, stability, and ecological contribution of the sites, continued monitoring was conducted at the sites from their inception through 1982. From their inception through 1977, monitoring was conducted by Kruczynski et. al. (1978) under the DMRP (Smith 1978), and from 1978 through 1982 under the auspices of the Dredging Operations Technical Support (DOTS) Program (Newling 1981, Newling and Landin 1983).

Site History and Description

Apalachicola Bay wetland development site is located on Drake Wilson Island in Apalachicola Bay, Florida (Figure 1). The bay is one of the most productive and least polluted estuaries in the United States (Kruczynski et al. 1978).

Rainfall averages 142.8 cm yearly and summers are hot and humid. Average annual temperatures are 20.4^{o}C with an average of only five days of below 0^{o}C weather. Tidal range is only $\pm$0.5 m in the bay; the salinity of the bay ranges from brackish to sea strength. The bay supports considerable commercial fishing for oysters, blue crabs, and shrimp, on which the local economy is based. Sport fishing in the bay

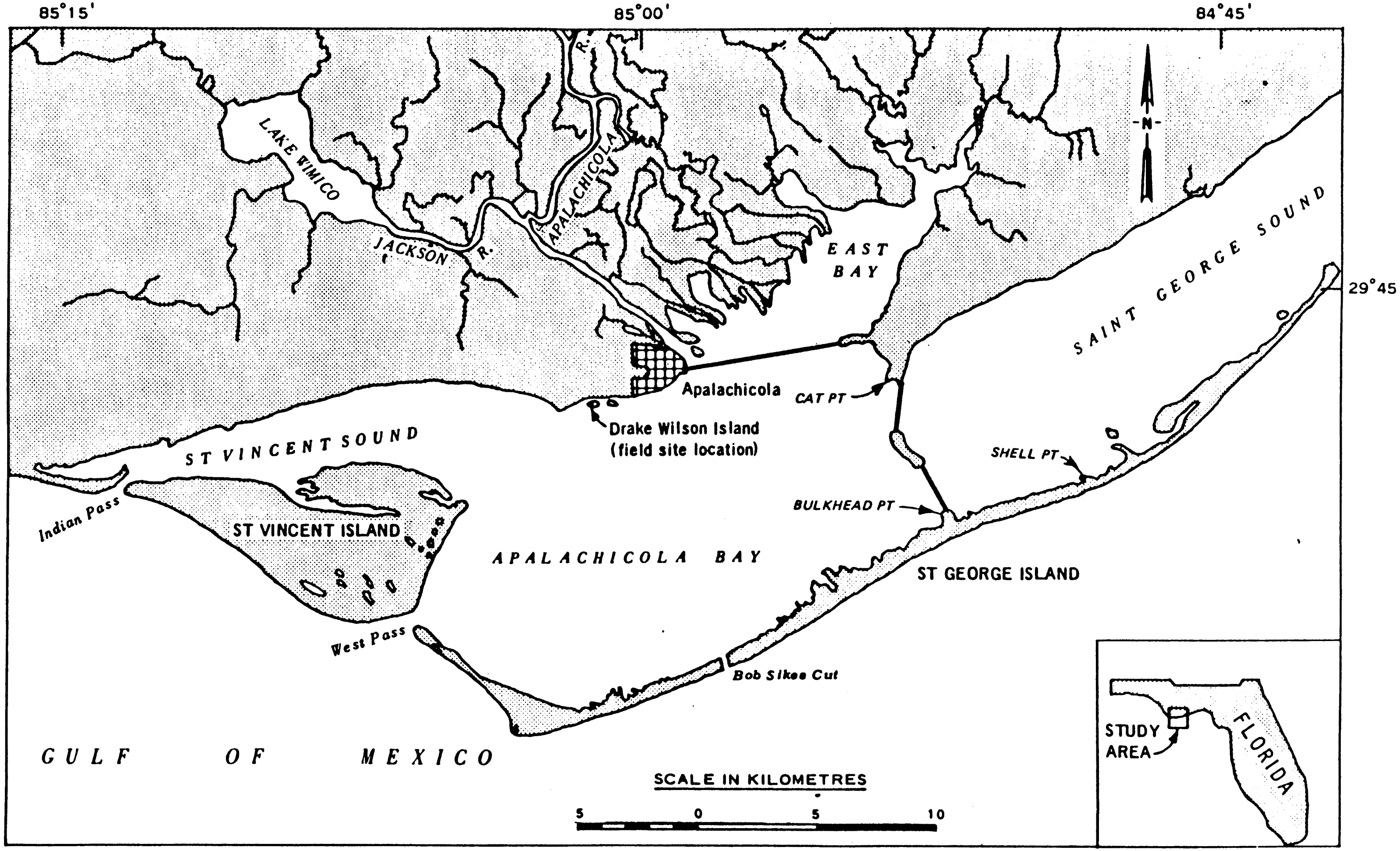


Figure 1. Location of the Apalachicola Bay wetland habitat development field site and three naturally developed reference areas, Cat Point, Bulkhead Point, and Shell Point (after Kruczynski et al. 1978).

for seatrout, redfish, sheepshead, whiting, and flounder, plus the mild weather, also draws a large tourist business to the area (Kruczynski et al. 1978).

Primary wildlife use in the estuary in the vicinity of the experimental site is by waterbirds, shorebirds, and some waterfowl. Herons, egrets, gulls, terns, and skimmers frequent the site. Small mammals such as raccoons and muskrats are common on the mainland.

Drake Wilson Island is one of two islands that were enlarged (Figure 2) in 1976 by diking with sandy clay dredged material from the bay, then by pumping coarse-grained sandy dredged material from adjacent Two-Mile Channel (Kruczynski et al. 1978).

Site Research and Development

The experimental site itself was enlarged by hydraulically pumping sandy clay dredged material from the bay bottom forming a triangular diked containment area island on the west side of and adjacent to Two-Mile Channel. The containment area was then filled with sandy dredged material from the channel. After filling, the dikes were graded to a gentle slope and a weir was placed on the bay side to allow tidal water influx. Finally, silty material was pumped over the sand to provide a more fertile and suitable substrate for planting. Despite attempts to achieve a more general distribution, the silty material dispersed primarily into the intertidal area, leaving the highest portions of the island with coarse sandy substrate. Engineering details are discussed in Kruczynski et al. (1978).

Transplants of smooth cordgrass and saltmeadow cordgrass were planted in 1976 in the intertidal zone within the diked interior of

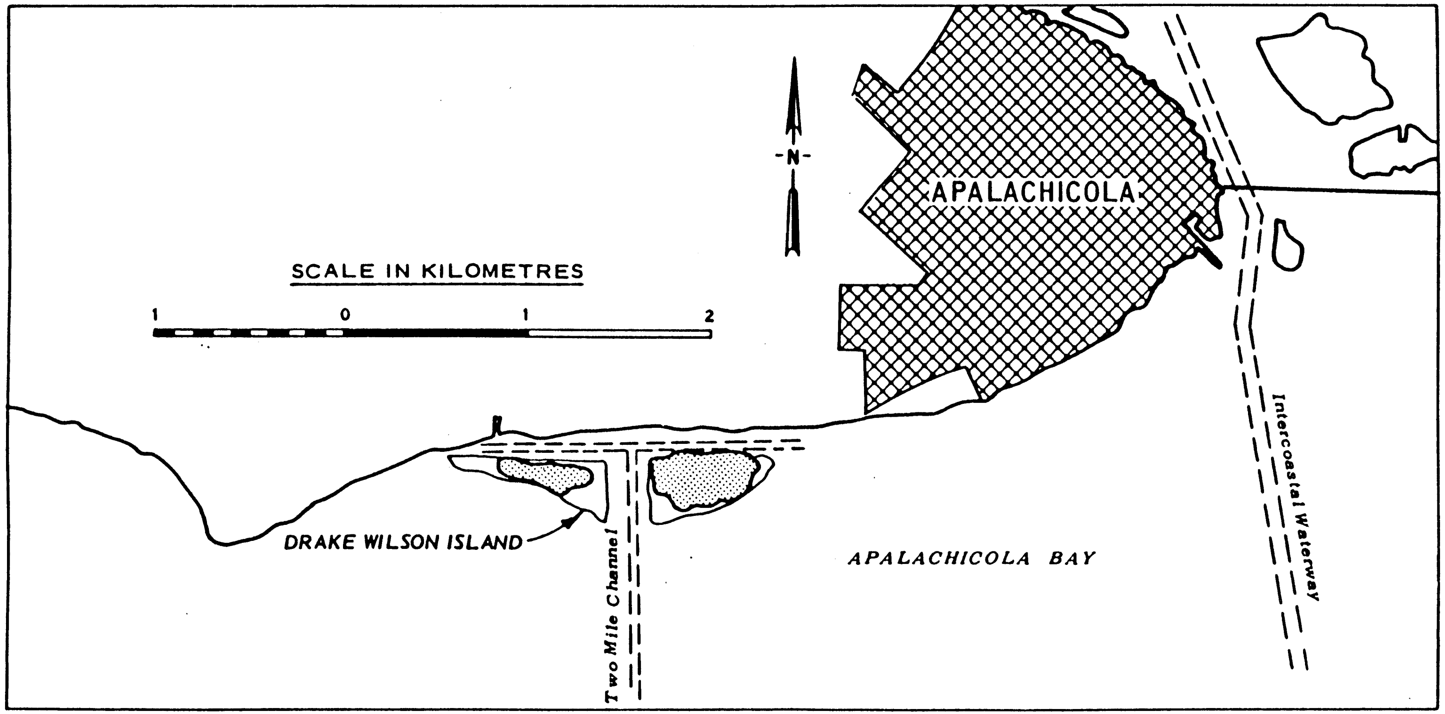


Figure 2. Location of two Mile Channel and the two disposal sites that resulted from its construction. The shaded areas indicate the approximate sizes of the disposal sites prior to disposal during the 1976 maintenance operation (after Kruczynski et a. 1978).

the island (Figure 3). Smooth cordgrass was planted at lower elevation and saltmeadow cordgrass was planted at the mean high tide line and above. Plantings were monitored for percent survival, percent cover, seed production, culm density, biomass, and numbers of new shoots. Methods, techniques, and analyses are discussed in Kruczynski et al. (1978). Various spacings for plantings were tested.

By the end of two growing seasons (Kruczynski et al. 1978), spacings of 1.0 m or less had 100 percent cover by smooth cordgrass. The plants were healthy, spreading rapidly, and producing seeds. At wider spacings, transplants either washed out or survived but grow poorly possibly due to tidal energy. Hand planted and mechanically planted transplants grew equally well.

In the higher marsh area, saltmeadow cordgrass showed vigorous growth response at spacings wider than 1.0 m. After two growing seasons (Kruczynski et al. 1978), all plots were growing and producing seeds; 100 percent cover was obtained by the end of the second year. Plantings at higher elevations survived better than those at mean high tide.

Forty-two plant species had invaded the planted site on the island by September 1977. Saltgrass (<u>Distichlis spicata</u>) was the dominant invader covering a large area by the end of 1977, especially in open areas between the smooth cordgrass and saltmeadow cordgrass plots (Kruczynski et al. 1978).

Soil samples collected prior to and after dredged material disposal indicated no contaminants in the sediments (Kruczynski et al. 1978). The sandy dredged material was fairly infertile and the silt dredged material contained abundant plant nutrient levels. Excessive

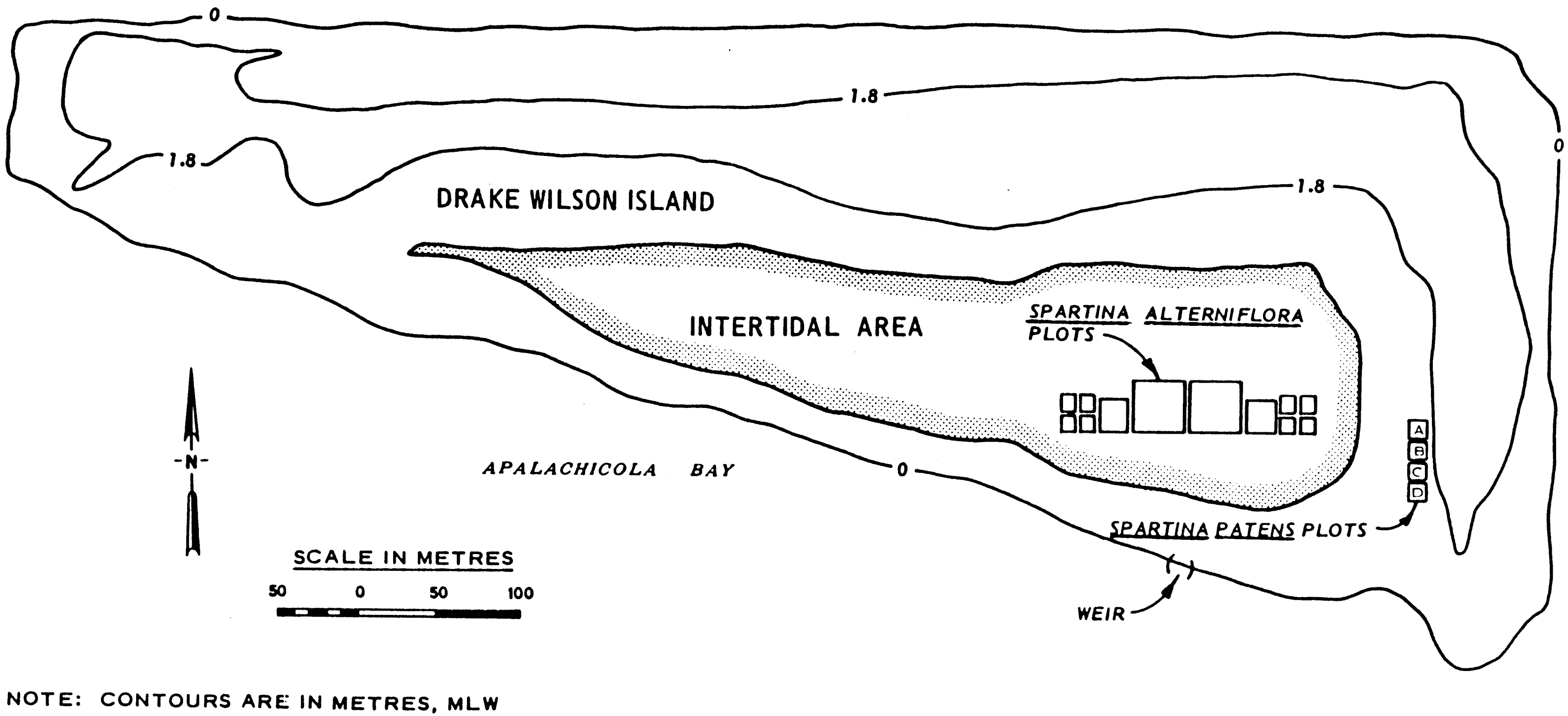


Figure 3. Locations of the original test plantings (June 1976) at the Apalachicola Bay wetland habitat development site on Drake Wilson Island (from Kruczynski et al. 1978).

soil salinity was not a problem for plant growth.

METHODS

Annual visits were made from 1980 to 1982 during which qualitative observations of physical condition, plant community, and wildife use were recorded for Apalachicola Bay habitat development site and the three nearby reference areas, Bulkhead Point, Shell Point, and Cat Point (Figure 1). The reference areas were naturally developed marshes of similar species composition, elevation, tidal regime, protection from typical storms, and size. In 1982, quantitative observations were also made on the plant communities at the experimental site and three reference areas. Random 0.5 m^2 quadrats were sampled within the intertidal zone: eight at the experimental site and four each at the reference sites. In each quadrat, species occurrence, stem density by species, mean height of ten randomly selected stems, and number of flowering stems were recorded. Within each of the four original supratidal zone saltmeadow cordgrass plots (Kruczynski et al. 1978), four random 0.25 m^2 quadrats were sampled for species occurrence and percent cover.

RESULTS AND DISCUSSION

The unconsolidated condition of the silty material in the intertidal zone of the experimental site has not changed appreciably since the time of disposal. While the material will not support the weight of a man, it will support the dense growth of smooth cordgrass which now dominates most of the site. Strong wind driven waves across the long fetch from the south have battered the south dike appreciably, eroding it in many places almost to the mean high tide elevation. On these areas as well as other low portions of the south dike, smooth cordgrass is the dominant plant cover. Storm tides appear to overtop the south levee fairly regularly. The weir is essentially non-functional although regular tidal exchange does occur by way of two small natural channels: one that has eroded around the weir itself, and a second that has breached the dike approximately 200 meters west of the weir.

Results of quantitative sampling in 1982 of the intertidal planting in the diked disposal area and the three reference sites are listed in Table 1. Measurements for the Apalachicola Bay habitat development site fell within the range of variability of those measured at the intertidal plant communities at the three reference sites. The quantitative measurements support qualitative observations noting strong comparability among the sites and in some cases, even more vigorous growth at the experimental site. Since 1980, the interior of the diked disposal area at the Apalachicola Bay site has changed from a wetland stand consisting of a patchwork of ponded openings to a totally closed stand with a dense cover of smooth cordgrass except for one small ponded area of subtidal elevation in the very center of the site (Figure 4c). The ponded area is heavily used for feeding by wading birds, primarily herons and egrets. The intertidal marsh supports a large population of clapper rails. In a band 20-30 meters wide extending into the intertidal portion of the disposal area from the sandy upland areas on the east and north (Figure 4d), the smooth cordgrass is regularly interspersed with saltmarsh bulrush (Scirpus robustus). The presence of this less salt tolerant species near the upland edges of the intertidal marsh may be in part the result of the influence of freshwater runoff from the upland.

Table 1. Summary of vegetation data collected at the Apalachicola Bay habitat development site and three reference areas on 24 August 1982.

Location	Species	Stem Density (Stems/ m^2)	Mean Stem Height (cm)	Frequency of Occurrence (%)	Mean No. Flowering Stems/ m^2
Apalachicola Bay H. D. Site* (Drake Wilson Island)	Smooth cordgrass (*Spartina alterniflora*)	137.8	93.7	100	0
	Saltmarsh bulrush (*Scirpus robustus*)	1.8	80.1	25	0
	Total	139.6	91.9		
	Mean percent cover = 73%				
Bulkhead Point**	Smooth cordgrass	83.5	79.2	100	0
	Total	83.5	79.2		
	Mean percent cover = 49%				
Shell Point**	Smooth cordgrass	190.0	52.3	100	0
	Total	190.0	52.3		
	Mean percent cover = 75%				
Cat Point**	Smooth cordgrass	161.0	111.4	100	0
	Total	161.0	111.4		
	Mean percent cover = 89%				

* Based on eight 0.5-m^2 quadrats.

** Based on four 0.5-m^2 quadrats.

a. July 1976

b. September 1977

c. October 1981

d. August 1982

Figure 4. Photographs of smooth cordgrass plantings (*Spartina alterniflora*) within the diked interior of the Apalachicola Bay wetland habitat development site. Vegetation in foreground of Fig. 4d is a combination of both smooth cordgrass and saltmarsh bulrush (*Scirpus robustus*).

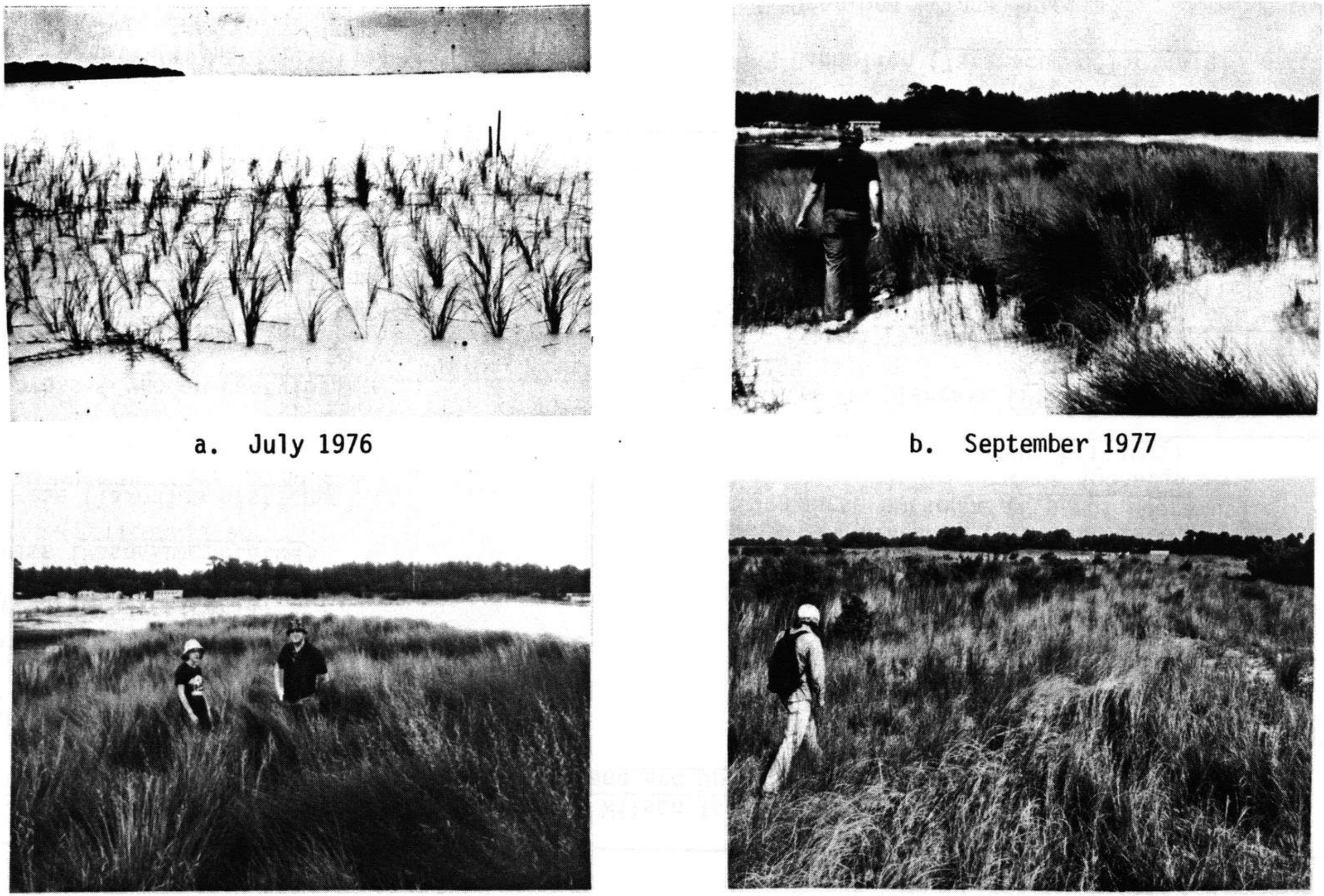
The results of the 1982 survey of experimental plots planted with saltmeadow cordgrass at and just above the mean high tide elevation are listed in Table 2. Percent cover and dominance of saltmeadow cordgrass are reduced markedly from earlier observations (Kruczynski et al. 1978). Invading species now dominate the plots. These differences are due largely to physical changes on the plots that were most likely induced by the successful, dense growth of the original saltmeadow cordgrass plantings themselves (Figure 5). Loose sand blowing from the adjacent, unplanted area was apparently trapped by the saltmeadow cordgrass, initiating formation of low dunes over the plots. Much of the original saltmeadow cordgrass plantings appear to have been smothered by 30-50 cm of sand. The site was sufficiently stable to allow invasion by species tolerant of the rather hostile environment of the experimental plots. The plots now exhibit a shifting, somewhat sterile sandy substrate with alternating conditions of drought or saturation and the added stress of periodic salt spray. Despite the rigorous conditions, the plots now support moderate plant cover as described in Table 2.

Table 3 lists the plants observed on the Apalachicola Bay habitat development site during the site visit in 1982. The 97 plant species observed represented a substantial increase over the 52 reported by Kruczynski et al. (1978). Of special note were the freshwater wetland species observed such as royal fern (_Osmunda regalis_), pennywort (_Hydrocotyle_ sp.), pilewort (_Erechtites hieracifolia_), and coarse rush (_Juncus biflorus_). None of these species are salt tolerant yet they all occurred in flats or depressions at the base of dunes and just above the mean high water mark. Most, particularly royal fern, exhibited a stressed, stunted growth form. Their presence is another evi-

Table 2. Mean percent cover (based on four 0.25 m^2 quadrats and species composition of saltmeadow cordgrass experimental plots) at the Apalachicola Bay habitat development site on 24 August 1982.

Species	Mean Percent Cover Plot A	Plot B	Plot C	Plot D
Bahia grass (*Paspalum notatum*)	14	3	3	5
Beardgrass (*Andropogon* sp.)	1	13	3	31
Blazing star (*Liatris* sp.)		1		
Brome grass (*Bromus* sp.)	1			
Club moss (*Lycopodium* sp.)	22	35	21	6
Coarse rush (*Juncus biflorus*)	1	1		2
Dog fennel (*Eupatorium capillifolium*)	2	2	1	2
Groundsel tree (*Baccharis halimifloria*)			1	
Marsh loosestrife (*Lythrum lineare*)			3	4
Pennywort (*Hydrocotyle* sp.)	3			
Pilewort (*Erechtites hieracifolia*)			1	trace*
Royal fern (*Osmunda regalis*)	1			
Saltmeadow cordgrass (*Spartina patens*)	5	9	14	9
Unidentified Composite				1
Unidentified Forb			trace*	
Unidentified Moss			3	11
Unidentified Sedge				10
TOTALS	50	64	50	81
Saltmeadow cordgrass (September 1977)	75-100	50-75	25-50	0-10
Saltmeadow cordgrass original spacing (m)	0.3	0.6	1.8	2.7

* trace $<$ 1 percent cover.



a. July 1976

b. September 1977

c. September 1978

d. August 1982

Figure 5. Photographs of the saltmeadow cordgrass (Spartina patens) experimental plots at the Apalachicola Bay habitat development site.

Table 3. Plant species observed on Drake Wilson Island including the Apalachicola Bay habitat development site compared to similar observations from nearby natural reference areas on 24 August 1982.

Drake Wilson Island (Wetland and Upland)
American three square (*Scirpus americanus*)
Arrowhead (*Sagittaria* sp.)
Bagpod (*Sesbania vesicaria*)
Bahia grass (*Paspalum notatum*)
Bald cypress (*Taxodium distichum*)
Barnyard grass (*Echinochloa crusgalli*)
Bermuda grass (*Cynodon dactylon*)
Big smartweed (*Polygonum pennsylvanicum*)
Bitter panic grass (*Panicum amarum*)
Black cherry (*Prunus serotina*)
Black needlerush (*Juncus roemerianus*)
Blazing star (*Liatris* sp.)
Broadleaf cattail (*Typha latifolia*)
Brome grass (*Bromus* sp.)
Centipede grass (*Eremochloa ophiuroides*)
Chufa (*Cyperus esculentus*)
Climbing hempweed (*Mikania scandens*)
Club moss (*Lycopodium* sp.)
Course nutsedge (*Cyperus odoratus*)
Coarse rush (*Juncus biflorus*)
Common plantain (*Plantago virginica*)
Common ragweed (*Ambrosia artemisiifolia*)
Common reed (*Phragmites australis*)
Crabgrass (*Digitaria sanguinalis*)
Curly-leaf dock (*Rumex crispus*)
Dallis grass (*Paspalum dilatatum*)
Dandelion (*Taraxacum officinale*)
Deer pea (*Vigna luteola*)

Table 3 (Concluded)

Spurge (*Euphorbia dentata*)	Yerba (*Eclipta alba*)
St. Augustine grass (*Stenotaphrum secundatum*)	Yucca (*Yucca* sp.)
Plants observed in or above the high marsh zone on one or more of the reference areas but not seen on Drake Wilson Island	
Common greenbriar (*Smilax bona-nox*)	Prickly pear cactus (*Opuntia* sp.)
Glasswort (*Salicornia* sp.)	Sea lavender (*Limonium carolinianum*)
Grapes (*Vitis* spp.)	Sea oates (*Uniola paniculata*)
Perennial foxtail grass (*Setaria geniculata*)	Wooly croton (*Croton capitata*)
Poison ivy (*Rhus radicans*)	Yaupon (*Ilex vomitoria*)

Table 3 (Continued)

Broom sedge (*Andropogon virginicus*)
Bushy beardgrass (*Andropogon glomeratus*)
Cabbage palm (*Sabal palmetto*)
Camphor weed fleabane (*Pluchea camphorata*)
Fleabane (*Erigeron* sp.)
Globe nutsedge (*Cyperus globosus*)
Green ash (*Fraxinus pennsylvanica*)
Ground pine (*Lycopodium obscurum*)
Groundsel tree (*Baccharis halimifolia*)
Lead Plant (*Amorpha herbacea*)
Lichens
Live oak (*Quercus virginiana*)
Loblolly pine (*Pinus taeda*)
Longlead pine (*Pinus palustris*)
Marsh elder (*Iva frutescens*)
Marsh loosestrife (*Lythrum lineare*)

Dog fennel (*Eupatorium capillifolium*)
European beachgrass (*Ammophila arenaria*)
Fall panic grass (*Panicum dichotomiflorum*)
Fimbristylis (*Fimbristylis castanea*)
Pigweed (*Amaranthus* sp.)
Pilewort (*Erectites hieracifolia*)
Plantain (*Plantago* sp.)
Pokeweed (*Phytolacca americana*)
Red rattlebox (*Sesbania punicea*)
Rose mallow (*Hibiscus* sp.)
Royal fern (*Osmuda regalis*)
Saltgrass (*Distichlis spicata*)
Saltmarsh cattail (*Typha domingensis*)
Saltmarsh fleabane (*Pluchea purpurascens*)
Saltmarsh morning glory (*Ipomoea sagittata*)
Saltmarsh sand spurry (*Spergularia marina*)

Table 3 (Continued)

Marsh rose mallow (Hibiscus moscheutos)
Mosses
Nutsedges (2 unid.)
Ogeechee plum (Nyssa ogeche)
Onion (Allium sp.)
Panic grass (Pancicum sp.)
Pepperbush (Clethra alnifolia)
Peppervine (Ampelopsis arborea)
Perennial saltmarsh aster (Aster tenuifolius)
Small white morning glory (Ipomoea lacunosa)
Smooth cordgrass (Spartina alterniflora)
Softstem bulrush (Scirpus validus)
Southern dewberry (Rubus trivialis)
Spiderwort (Tradescantia virginiana)
Spikerush (Eleocharis sp.)
Spiny sandspur (Cenchrus echinatus)
Saw grass (Cladium jamaicensis)
Sea oxeye (Borrichia frutescens)
Sea purslane (Sesuvium portulacastum)
Seashore mallow (Kosteletzkya virginica)
Seaside goldenrod (Solidago sempervirens)
Sedge (Carex sp.)
Sensitive fern (Onoclea sensibilis)
Shortleaf pine (Pinus echinata)
Sicklepod (Cassia obtusifolia)
Swamp dock (Rumex verticillatus)
Switchgrass (Panicum virgatum)
Water hemp (Amaranthus cannabinus)
Water hyssop (Bacopa monnieri)
Water pennywort (Hydrocotyle bonariensis)
Water smartweed (Polygonum punctatum)
Wax myrtle (Myrica certifera)

dence of freshwater runoff through the dunes, probably forming a layer that floats above a saline of brackish water table as the freshwater reaches the saltmarsh elevation at approximately the mean high water elevation. The stunted growth forms are probably the results of periodic drought and salt stress.

Wildlife use of the entire site was substantial (Table 4). Habitat interspersion on the island itself as well as interspersion of the entire island and its habitats in relation to other nearby areas is probably a major factor in its success for wildlife use. Most conspicuous is the use of the wetland habitat development site by wading birds for feeding. The site supports a large population of clapper rails which appeared to be much greater than that observed at any of the reference areas.

Table 4. Wildlife observed on Drake Wilson Island including the Apalachicola Bay habitat development site compared to similar observations on three nearby natural reference areas on 24 August 1982.

Drake Wilson Island including the Habitat Development Site	Bulkhead Point
Barn swallow	Clapper rail
Belted kingfisher	Fish crow
Black-bellied plover	Laughing gulls
Black vulture	Yellow-crowned night heron
Brown pelican	Unidentified plovers
Clapper rail	Fiddler crabs
Common crow	
Double-crested cormorant	**Cat Point**
Fish crow	Black-bellied plover
Great blue heron	Great blue heron
Great egret	Laughing gulls
Laughing gull	Eastern mole
Least tern	Fiddler crab
Little blue heron	
Louisiana heron	**Shell Point**
Red-tailed hawk	Clapper rail
Royal tern	Laughing gull
Sandwich tern	Osprey
Snowy egret	Robin
Cottontail rabbit	
Eastern mole	

Table 4 (continued)

Drake Wilson Island including <u>the Habitat Development Site</u>
Unidentified small rodents (mice, voles, shrews)
Fire ants
Native ants
Blue crab
Fiddler crab
Hermit crab

SUMMARY

The habitat development site at Apalachicola Bay was highly successful. A vigorous marsh plant community comparable to those at nearby natural marsh areas dominates the site. Erosion from wind driven waves from the long south fetch are eroding the main containment dike on the south side but the site is still intact. The site receives heavy wildlife use.

LITERATURE CITED

Kruczynski, W. C., R. T. Huffman, and M. K. Vincent. 1978. Habitat development field investigations, Apalachicola Bay marsh development site, Apalachicola Bay, Florida; summary report. Technical Report D-78-32. U. S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi. 39 pp.

Newling, C. J. 1981. Monitoring of Dredged Material Research Program habitat development sites. Dredged Material Research Notes, News, Reviews, etc. U. S. Army Engineer Waterways Experiment Station. Vicksburg, Mississippi. Vol. D-81-1. pp. 1-8, 12.

Newling, C. J., and M. C. Landin. 1983. Habitat development investigations at seven upland and wetland dredged material disposal sites. Technical Report, in preparation. U. S. Army Engineer Waterways Experiment Station. Vicksburg, Mississippi.

Smith, H. K. 1978. An introduction to habitat development on dredged material. Technical Report DS-78-19. U. S. Army Engineer Waterways Experiment Station. Vicksburg, Mississippi. 40 pp.

COMPARISONS OF NURSERY PRACTICES FOR GROWING OF *Rhizophora* SEEDLINGS

J. B. REARK, ECOLOGIST

BIOLOGICAL RESEARCH INSTITUTE
SOUTH MIAMI. FLORIDA

ABSTRACT

Four methods for germinating and growing of *Rhizophora* seedlings were simultaneously tested in a slathouse under uniform conditions, glass jars, nursery pots, tree pots, and Jiffy 7 peat balls. All watering was from municipal tap, with weekly soluble acid fertilizer, and no other chemicals applied. Results showed the clear superiority of the Jiffy 7 under test conditions. Mortalities and comparative costs of production are discussed, including the planting out process. Jiffy 7 methodology proved most cost efficient, and both Jiffy 7 and tree pots had longer shelf life than the other methods tested.

INTRODUCTION

Within the past decade mitigation plantings have become commonplace in wetlands everywhere in the State of Florida. As an observation, the growing of plants for mitigation plantings has not been in the hands of skilled horticulturalists generally. Understandably, a decade ago when care of the environment during development passed from the realm of the dream of the few to the legal necessity for all, academicians were nearly the only the ecologic technicians available for a field which had no capital, no literature, and almost no experience. The plants were not available commercially as a rule, a condition which will hopefully be changing as did large scale landscape design in the sixties. We should be moving from the back yard growing of mitigation plants in discarded plastic dishpans, fastfood foam coffeecups, and salvaged tin cans to the standardized production of high quality planting stock for which trained Florida nurserymen have justly become renowned. It would also be desireable to see all such efforts planned by demonstrably skilled ecosystemic architects and constructed by licensed contractors. None of that will make any sense unless state and county agencies get serious about boat wakes, off-the-road four-wheelers, and the variety of tracked and airplane engine vehicles one finds in our wetlands, and totally banning the presence of any of them except in designated areas. If it is to be legal to require environmental protection and re-establishment from owners of lands, the corrollary must be that police powers be employed for the protection thereof.

In a small effort to begin a look at quality in mitigation plantings, an experiment was devised to test various methods currently in use by the growers of both mitigation and physiology red mangroves, *Rhizophora mangle* L.,and to test a departure from those practices that had great theoretical promise.

MATERIALS AND METHODS

The experiment was performed on benches in a wood slathouse designed to produce 50% shade, the 2x4cm slats spaced 4cm and running north-south. The location was in South Miami, Dade County, Florida, USA.

Mangrove propagules were collected from healthy trees in Key Largo, Florida, and were culled visually for symptoms of insects or disease. 100 propagules from 15 to 25cm in length were divided into four groups of 25, each exhibiting the full size range. Each group was planted uniformly as follows:

Group 1. Placed into 124ml babyfood jars with ca.100ml of tap water. This method approximates that of laboratory physiologists who might also use seawater or formulated experimental waters.

Group 2. Planted into plastic pots 66mm OD x63mm, using a 1:1 sterilized soil mix of everglades peat and silica sand that is commercially available to nurseries in the Dade-Broward market area. This method approximates the mean of current nursery practice.

Group 3. Planted into plastic tree pots measuring 58mm square and 122mm deep using commercial sponge clippings to not only seal the open bottom of the pot, but to furnish a hygroscopic aid to moist soil conditions, a method much utilized in the Bahamas nurseries.

Group 4. Planted directly into Jiffy 7s, a manufactured product from Norway. The product is a compressed disk of peat measuring 9x47mm as packed complete with a planting depression and in a nylon net which controls the shape after wetting. 15 minutes after soaking the disk, it expands to 30x50mm, and is then ready for planting. Plastic trays measuring 28x55.5cm are manufactured to contain 55 Jiffy 7s.

Watering was daily, using tap water as distributed by the Dade County Water and Sewer Authority. All plants were fertilized weekly using an end-of-hose siphon sprayer and Stern's Miracid soluble fertilizer diluted at the manufacturer's recommended rate of 1.27g/l (1 tsp/gal). The acid formula was chosen because a previous experiment had indicated that such a product might avoid chlorosis (Reark, 1982), and no chlorosis developed during the course of this experiment. No rooting hormones, fungicides, insecticides, or any other chemical were applied to the propagules at any time. The experiment was begun April 23, 1983, reported on May 19, 1983, and concluded on June 5, 1983.

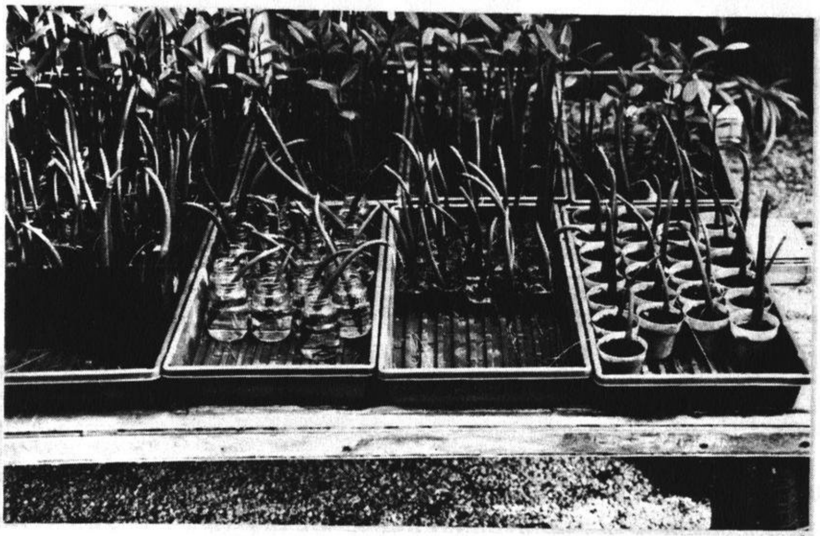


Fig.1. Photograph taken April 23, 1983 showing the benched experiment. Left to right, Group 3 - plastic tree pots, Group 1 - glass jars, Group 4- Jiffy 7s, and Group 2 - standard nursery pots. Each group is in its own 28x55.5cm plastic tray, and all face south. Note the comparative space that 25 planted propagules occupies in the standard tray.

RESULTS

The first rooting became apparent in Group 1, the glass jars, on May 14, at which time each method was analyzed for presentation and photographed. Group 1 at that time had 3 propagules with definite roots to 4mm, Group 2 had 5 propagules with roots to 10mm, Group 3 had 13 propagules with roots to 15mm, while Group 4 had 18 propagules with roots showing through the fabric of the Jiffy 7 to a distance of over 20mm from the center (Fig.2). Each propagule was returned to its own pot or jar, and the observations continued until a pattern of time of leaf formation could be established.

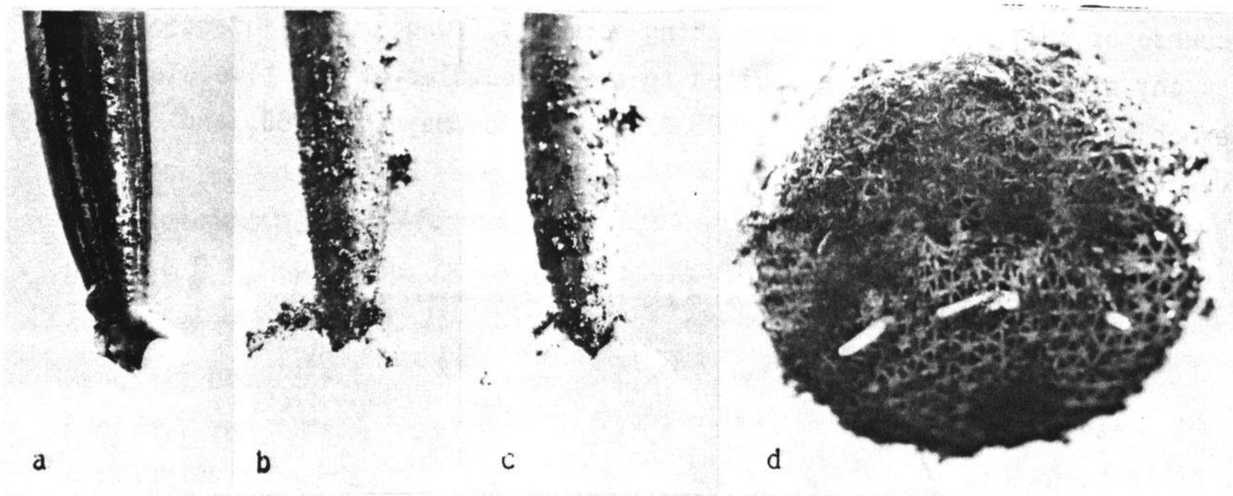


Fig.2. Rooting after three weeks, red mangroves. (a) Group 1 - glass jars, (b) Group 2 - nursery pots, (c) Group 3 - tree pots, and (d) Group 4 - Jiffy 7, the last showing typical root emergence. All photos x 1.2 natural size.

By June 5, sufficient growth had occurred to compare the methods for a judgement of nursery efficiency and cost comparisons.

GROUP	ROOTING	FOLIAGE
1 - glass jars	8	5
2 - nursery pots	12	7
3 - tree pots	17	17
4 - Jiffy 7s	22	22

Table 1. Summary of state of red mangrove propagules at the end of six weeks. Foliage indication is leaf vernation after shedding of the stipule. It should be noted that groups 1 and 2 had no expanded leaf pairs at this time, while groups 3 and 4 had 10 and 13 respectively.

Due to the short length of the experiment, no mortality figures could be assembled. However, the high mortality using standard #6 plastic nursery pots (ca. 15cm ID), which approached 50% under freshwater conditions, is what prompted the search for a better method. Experience with many trays of both tree pot and Jiffy 7s have given tree pot growth after one year at an average of 40cm with 3 pairs of leaves and a mortality which averages at 7 per tray and none after plant out, while the Jiffy 7s give a much more uniform growth of 50cm, 3 pairs of leaves, and mortality of 2.6 per tray (Fig.3), as compared with 200 #6 pot plants with averages of 35cm growth, 3 pairs of leaves, and mortality of 42%. There is a further mortality after plant out of the #6s, against none for tree pots or Jiffy 7s to date.

Fig.3. Exuberant growth exhibited by trays of red mangroves in Jiffy 7s planted July 15, 1982 and photographed May 14, 1983.

DISCUSSION

The results of this experiment can be readily translated into dollars per unit area. The plastic tray accupies an area of 1554cm^2 and will hold 36 of the 124ml glass jars (group 1), 30 of the plastic pots of group 2, 36 of the group 3 tree pots, and 55 Jiffy 7s. Two rows per bench plus one aisle 45.7cm wide (18 in) will give a realistic unit area per plant and this can be integrated with the current market average price of $ 1.00 to give the following tabular results:

GROUP	UNITS M^2	(SqFt)	NET SALE M^2	(SqFt)
1	164.4	15.22	$ 164.40	$ 15.22
2	136.9	12.68	$ 136.90	$ 12.68
3	164.4	15.22	$ 164.40	$ 15.22
4	251.2	23.26	$251.20	$ 23.26

However, when mortalities are figured into the tabulation, the figures change dramatically:

GROUP	% MORTALITY	NET SALE M^2	(SqFt)
1	32%	$ 111.80	$ 10.35
2	42%	$ 79.40	$ 7.35
3	19%	$ 133.16	$ 12.33
4	5%	$ 238.64	$ 22.10

To return to the analysis of production on a per plant basis, we get the following costs:

mangrove propagule collection @ $ 0.06 - 0.08
care and watering per month @ 0.012
nursery set-up per month 0.92/M^2 - 0.085/SqFt
@ varies by group, see tabulation u/a
costs of pot ready to use: Group 1. Cheap if you have a baby + dishwasher
Group 2. $ 0.04 - 0.06 with soil
Group 3 $ 0.06 - 0.08 with soil/sponge
Add $ 0.51 for plastic tray, varies u/a
Group 4. $ 0.071 w/tray

The production costs per seedling at six months will thus be theoretically be close to $ 0.30 - 0.35 each, but application of the mortalities again widens the gap between the Jiffy 7 and other methods tested. All mortalities have been derived from 11 years of horticultural experience with red mangroves at the South Miami site.

The final cost information under the conditions of an inland site using freshwater irrigation is that the shelf-life of Groups 1 & 2 is between 3 & 6 months before transplantation is necessary, and that Groups 3 & 4 will not only endure for at least 15 months, but can be planted out as is at the rate of one per minute by virtually anyone at no loss.

Under the conditions of this experiment, the Jiffy 7 method demonstrated better growth and a higher economic efficiency than did other techniques.

LITERATURE CITED

REARK, J.B. An in situ fertilizer experiment using young *Rhizophora*. in Webb, F.J., ed. 1982 Proceedings of the Ninth Annual Conference on Wetland Restoration and Creation. Hillsborough Community College, Tampa, Florida.

WETLAND VEGETATION AND AGRONOMY

PRELIMINARY REPORT

ROBERT M. SNYDER, P.E.

Snyder Oceanography Services, Inc.
Jupiter, Florida

LEONARD J. GREENFIELD, Ph.D.

Rio Palenque Research Corporation
Coral Gables, Florida

DONALD SHACKELFORD, P.E.

The Bankers Land Company
Palm Beach Gardens, Florida

STEVE BEEMAN, M.S.

Ecoshores, Inc.
South Daytona, Florida

PRELIMINARY REPORT

Beginning in the mid-1970's, two adjacent projects in northern Palm Beach County adjoining the Intracoastal Waterway were permitted and constructed. Comprising about 10 miles of canal banks, the projects are unique in that almost the entire shoreline was sculpted and revegetated with cordgrass and red mangroves. Planting took place over a 12 month period starting in the autumn of 1980. Spartina alterniflora establishment utilized about 650,000 plants, the majority being transplanted clumps from a lease area with nursery grown sprigs used in about 30% of the planting area. Approximately 250,000 Rhizophora mangle nursery seedlings were planted interspersed with the cordgrass. Observation over the next 18 months indicated a wide variation in growth and appearance of the cordgrass. Close scrutiny indicated a number of variables such as soil properties, groundwater flow and shading that might influence growth and survivability of the plants. Many of the variables are not normally considered in permit monitoring requirements. It was concluded that, because of the expanse of the projects and the fact that more than fifty acres of fringe saltmarsh were created from essentially upland property, there was a unique opportunity to monitor plant response over an extended period with the view of developing agronomic best management practices for use in future projects.

Following a review of the literature, a new monitoring program was designed and approved by DER to replace that called for in the permit documents. The project location is shown in Figure 1 and the summary of parameters is shown in Table 1.

To date, three quarterly field investigations have been completed and detailed data are being worked up. The study is scheduled to run a total of 2 years and is expected to lead to publication of a number of papers and technical notes as well as practical guides for salt-marsh fring establishment. Preliminary review of the field studies have led to the following general observations. Other than perturbations such as storm related washouts and slumping, the major influences of plant growth apparently result from (a) variations in horizontal groundwater flow leading to invasion of freshwater species at the elevation of high tide and in the upper banks, (b) variability in soil characteristics and (c) shading of new growth following seasons of lush growth and dieback. The invasion by red mangrove seedlings is nearly triple the amount originally planted in areas of considerable flushing. This supports earlier observations that the presence of cordgrass or other inter-tidal grasses leads to the natural capture of seedlings. It also points to the possibility of mangrove establishment by the release of seeds and seedlings in areas where cordgrass has been established. The invasion of white mangroves is considerable and some black mangroves have also volunteered.

The work is being supported by The Bankers Land Company which holds title to the properties and is being coordinated by the Palm Beach Soil and Water Conservation District which is conducting a county wide inventory of estuarine conditions. It is hoped that the information developed will assist the newly formed Florida Association of Conservation District's Coastal and Wetland Resources Committee in establishing statewide guidelines for the creation and enhancement of coastal estuarine conditions.

Table 1 - Summary of Parameters

1. STATIONS: Frenchmans Creek - 6 vegetation; 6 water/sediment
   Admiral's Cove -20 vegetation; 8 water/sediment

2. FREQUENCY: Quarterly + post-event measurements

3. DETERMINANTS:

   a) population size and growth
   b) relationship of above to environmental conditions

4. POPULATION AND GROWTH FACTORS:

   a) number of plants per quadrat
   b) growth rate
   c) proliferation by root stock
   d) seed production
   e) gross production
   f) protein content and key minerals (selected stations)

5. ENVIRONMENTAL FACTORS:

   a) salinity, temperature, and dissolved oxygen
   b) nutrients
   c) visibility
   d) plankton
   e) bottom fauna (diversity)
   f) bank soil quality (physical and chemical) vs. elevation
   g) groundwater flow and chemistry
   h) finfish count

6. VISUAL OBSERVATIONS:

   a) relative appearance throughout projects
   b) anomalous behavior
   c) invasion
   d) dieout
   e) response to climatological events

7. PLANT EXPERIMENTS:

   a) Distichlis
   b) Sporobolus
   c) S. patens
   d) cross-planting

8. REPORTS:

   a) quarterly data reports to Bankers and DER
   b) semi-annual analysis
   c) annual results and conclusions
   d) preparation of papers for delivery and publication

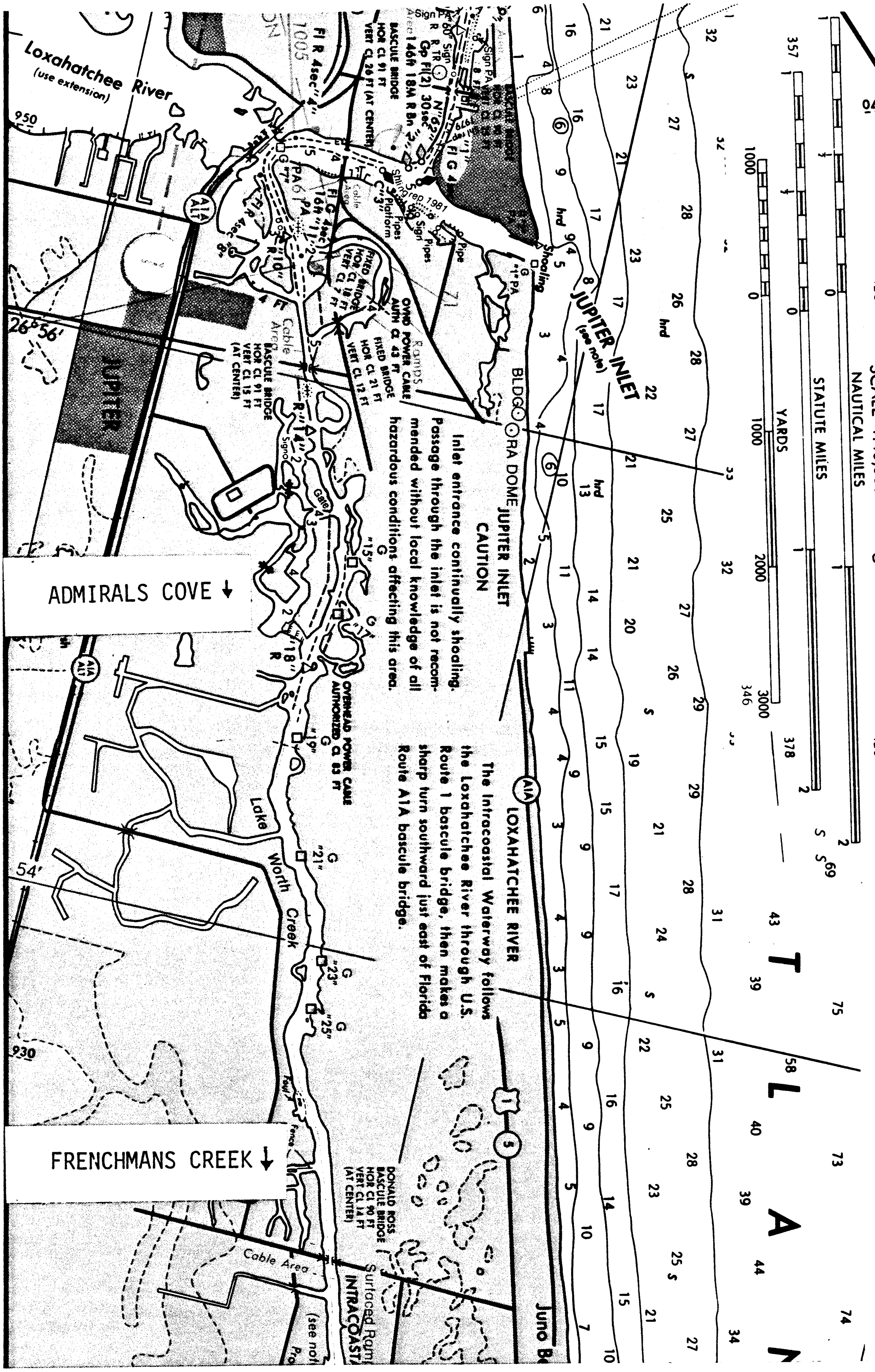


Figure 1. Project Location - Northern Palm Beach County, Florida
Complete works not shown.

MANGROVE RESTORATION IN NAPLES, FLORIDA

MICHAEL F. STEPHEN, Ph.D.

Coastal Engineering Consultants, Inc.
3883 Davis Boulevard
Post Office Box 8306
Naples, Florida 33941

August, 1983

ABSTRACT

During the initial dredging of the navigation channel into Naples Bay, dredge spoil was sidecast onto adjacent Bay fringe wetlands. In addition to direct wetland destruction, the spoil placement caused alternation and blockage of normal tidal flushing into the landward mangrove forest.

Restoration of these spoil sites was proposed as part of the development plan for the Windstar Golf and Multi-family Community located on upland properties immediately east of Naples Bay. Construction activities for the restoration were begun in May, 1982 with clearing and scraping of the restoration sites. After the design grade was achieved, red mangrove propagules were hand plugged over the fifteen acre site.

A status survey completed in January, 1983 revealed a 97% survival rate for the propagules. Most propagules had sprouted multiple leaf clusters and were 45 to 53 cm (18 to 21 inches) in height. In addition, there were numerous "volunteer" black mangrove established along the site perimeter. In the shallow water of the interior, small oyster communities were observed.

This project provides vivid documentation as to the feasbility and value of wetland restoration.

INTRODUCTION

Development activities in the coastal area of Florida have historically produced significant damaging impacts to the natural wetland and bay ecosystems that are prevalent along our coastlines. In his excellent summary of Mangrove distribution, productivity and revegetation techniques, Lewis (1982) indicates that approximately 44% of the wetlands surrounding Tampa Bay and approximately 82% of the wetlands surrounding Biscayne Bay have been lost due to development. As a result of such activity, stringent environmental regulations have been employed to restrict future development in an attempt to protect what is left of the fringing bay and wetland environmental habitat.

While preservation is necessary, it is important to recognize that there are many locations in these areas which suffer from the activities of the past, but which, with proper planning, design and implementation, can be restored to their natural functional value. Typically it has not been cost effective for the private land owners to voluntarily restore significantly altered or damaged wetland environments. However, if close scrutiny is applied to the coastal zone areas by engineers and scientists when development or redevelopment activities occur, a number of methods may be employed to restore productive and viable wetland habitat. This report is an example of just such a case.

Historical development activities in Naples Bay, Florida (see Figure 1) involved dredging of the natural mangrove wetlands to create the classic Florida development of finger canals and filled spits or fingers used for residential upland development. Additionally, a channel was dredged in the central portion of Naples Bay to form a navigation channel from the Gulf of Mexico to the head of Naples Bay.

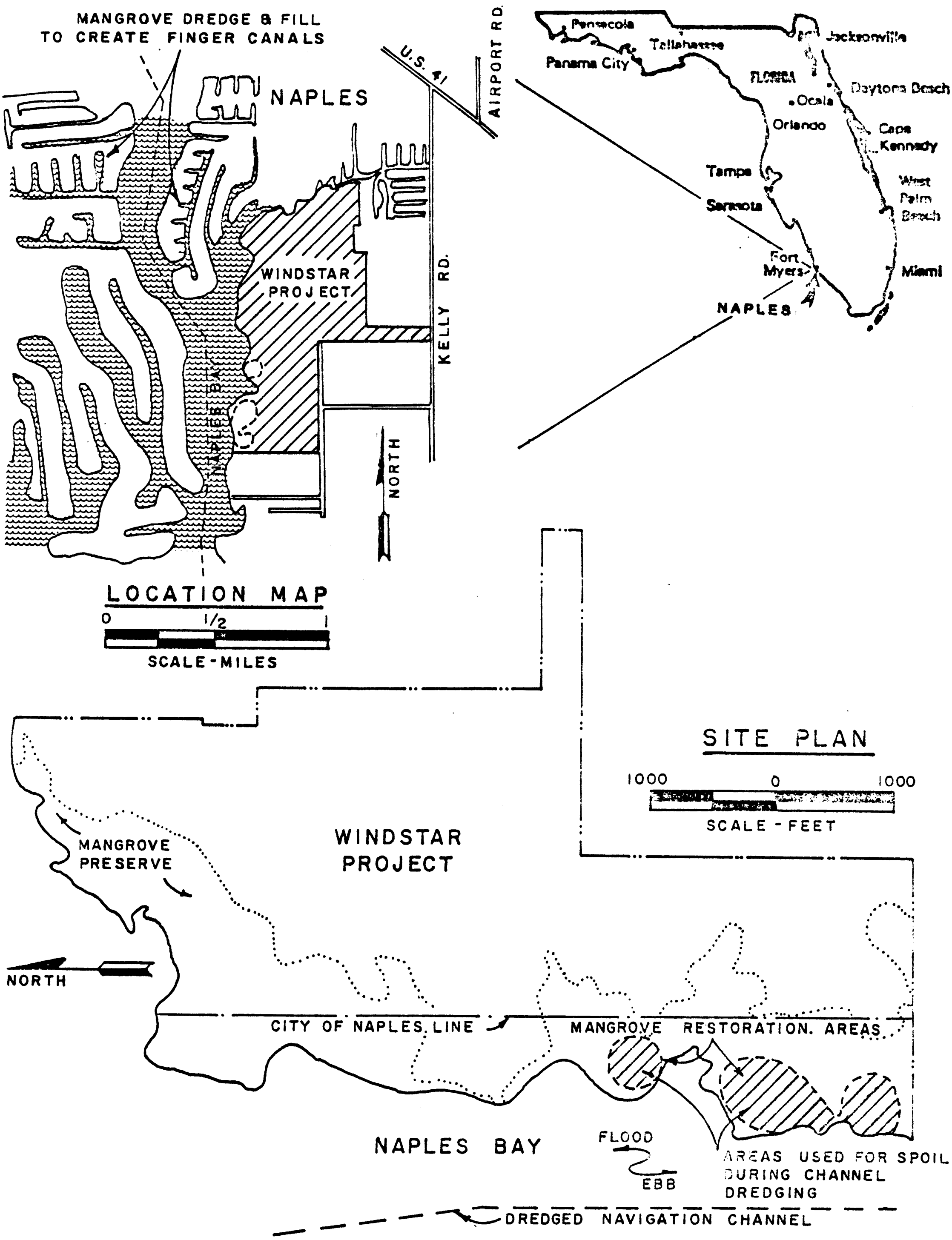


Figure 1. Location of Project

During the initial dredging, spoil was sidecast onto the adjacent wetlands along the east side of the Bay. This activity, in addition to direct wetland destruction, caused alteration and blockage of normal tidal flushing into the landward mangrove forest, thereby reducing nutrient return to the Bay and stunting the remaining mangrove community. These spoil piles were gradually overgrown with noxious, exotic vegetation, such as Australian Pine (Casarina equisetifolia) and Brazilian Pepper (Schinus terebinthifolius).

The restoration of these spoil sites to recreate mangrove wetlands was proposed as part of the development plan for the Windstar Golf Course and Multi-family community. The project, located on the east side of Naples Bay (Figure 1), involved a 500 acre tract that in addition to the golf course and residential development would retain over 100 acres to be mangrove preserve. After the appropriate permit approvals were obtained, construction activities for this restoration were begun in May, 1982.

Initial field surveys of the property also identified an extensive ditch network that had been cut through the mangrove fringe and into the interior of the project. This ditch system had apparently been created by Collier County in an attempt to drain interior ponded low areas to benefit mosquito control. This mosquito ditch network, while adversely affecting saltwater intrusion in the area did support a heavy lateral growth of fringe mangroves along the sides. In order to utilize the upland property, it became necessary to fill the ditching system and, therefore, eliminate the $\pm$2.0 acres of fringe mangrove in the ditches. This restoration of the approximately 15 acres of mangrove served as a mitigation to allow filling of these mosquito ditches.

MATERIALS AND METHODS

Construction Methods

Initial activities involved the careful flagging of the mangrove perimeter which surrounded the old spoil sites. The flagging served to identify clearing limits for the contractor; then an entrance road alignment from the upland development area, through the mangrove forest and into the spoil sites was carefully flagged.

This enterance road was cleared as the first step in construction. After careful clearing and stump removal to avoid disturbance of the sub-soil, a filter cloth layer was put down and covered with sand to create the roadbed. This road was carefully constructed, (1) so that the sand did not extend beyond the edges of the filter cloth and into the mangroves on each adjacent side, and (2) so that it could be removed after the completion of grading and planting on the interior sites. Once the interior work was complete this sand could be scraped off the filter, the filter cloth pulled out and these entry areas replanted.

Once the entry road was completed, the contractor progressed into the interior to clear the spoil site. A chipper was used initially to avoid excessive disturbance of soil. The chipper was able to mulch all of the thick underbrush. The remaining stands of large trees were then felled towards the interior. The trees felled towards the inside and away from the adjacent edge of the mangroves were then placed in piles and burned on the site.

Following the clearing of each site large pans were used to scrape the material down to a level where the equipment could no longer run and be supported by the underlying sand layer. At this point, a large backhoe was brought in to do the final excavation to the design grade which was elevation +45 cm (+1.5 ft.) NGVD. Survey stakes were placed over the site to provide elevation control for the design grade. Design requirements stipulated that the site be totally flooded at mean high tide. Florida State Department of Natural Resources tidal bench marks were used to establish the approximate elevation of mean high water. In addition, the elevation of the surrounding natural mangrove's peat surface was surveyed and used as a guide for establishment of the control elevation. Two excellent papers, one by Teas (1977) and another by Lewis (1982) provide significant information pertaining to site characteristics, water depth and transplanting procedures as well as general background and comparitive information regarding other similar projects of this type.

After the scraping was completed the project was then hand planted with red mangrove (Rhizophora mangle) plugs bundled in pairs. The mangrove propagules were collected from the numerous bays and imbayments located in the Ten Thousand Islands to the south of the project. Over 70,000 propagules were planted on this site. The average original height of the propagules varied between 20 to 25 centimeters (8-10 inches). None of the propagules were rooted at the time of their placement. The propagules were similar in form to type B as defined by Teas (1975). Propagules placement was done by stringing lines across the site, and then the plugs were hand placed by Mangroves Systems, Inc. (Tampa, Florida) personnel. Bundled pairs were plugged at a spacing of 1 meter on center. The use of bundled pairs was suggested by the project environmental consultant, Kevin Erwin, as a method of achieving a higher rate of survival for the plugged propagules.

RESULTS AND DISCUSSION

The planting was completed in August, 1982. A six month follow-up survey was completed in January, 1983. During the follow up survey, a visual inspection of the planting sites was made. Based on this visual inspection, it was estimated that approximately 97% of the plugged propagules had survived the first six month period since installation. Most propagules had sprouted multiple leaf clusters and were +45 to 53 cm (18 to 21 inches) in height. This was a fairly rapid growth rate. Growth rates reported by Goforth and Thomas (1980) indicated average heights of 35 to 38 centimeters after a period of approximately 6 to 7 months from planting, 35 to 50 centimeter heights were not achieved until a full year after planting in their survey.

In addition, to propagule growth there were numerous volunteer black mangroves (Avicennia germinans) established along the site perimeter. In ponded shallow water throughout the interior small oyster communities were observed. Extensive algae growth covered the bottom and was providing oxygen enrichment of the warm shallow water and supporting small fish, which in turn induced numerous birds to enter and feed.

Table 1 was developed from Figure 2 which shows a plot of mangrove growth height vs. elevation. A maximum growth height occurred at about elevation +39 cm (+1.3 ft.) HGVD. Optimal propagule growth centered between elevation +36 to 45 cm (+1.2 ft. to +1.5 ft.) It is interesting to note that design grade, +45 cm (+1.5 ft.) was selected on the basis of surveys on the surrounding mature mangroves. However, the preliminary data in Figure 2 indicate that the best growth occurred at elevation +39 cm (+1.3 ft.).

Contrary to what Goforth and Thomas (1979) describe as significant increases in propagule survival at +0.1 m, no progagules survived at that elevation in this study. Hannan (1975) states that placing transplants as little as 10 centimeters too low in the tidal zone caused heavy mortality. Teas (1976) indicated that mangroves were absent in Biscayne Bay at land elevations less than 9 centimeters below sea level. It was postulated by Teas (1976) that this was probably due to inhibition of rooting and early development under anaerobic conditions. Davis (1940) relates poor mangrove propagule growth in areas of standing, stagnant water. It is clear, therefore, that the planning elevation is critical for a successful restoration project.

Figure 3 is the height vs. elevation plot with site characteristics superimposed. This figure yields useful information for planning of future restoration projects in that it suggests a succession of conditions for optimal growth initiation of different wetland/bay community types. For example, dense clusters of volunteer black mangrove were limited to the higher elevations with lower soil moisture along the outer perimeter of each site. Volunteer oysters were limited to isolated low areas which retained standing water throughout the tidal

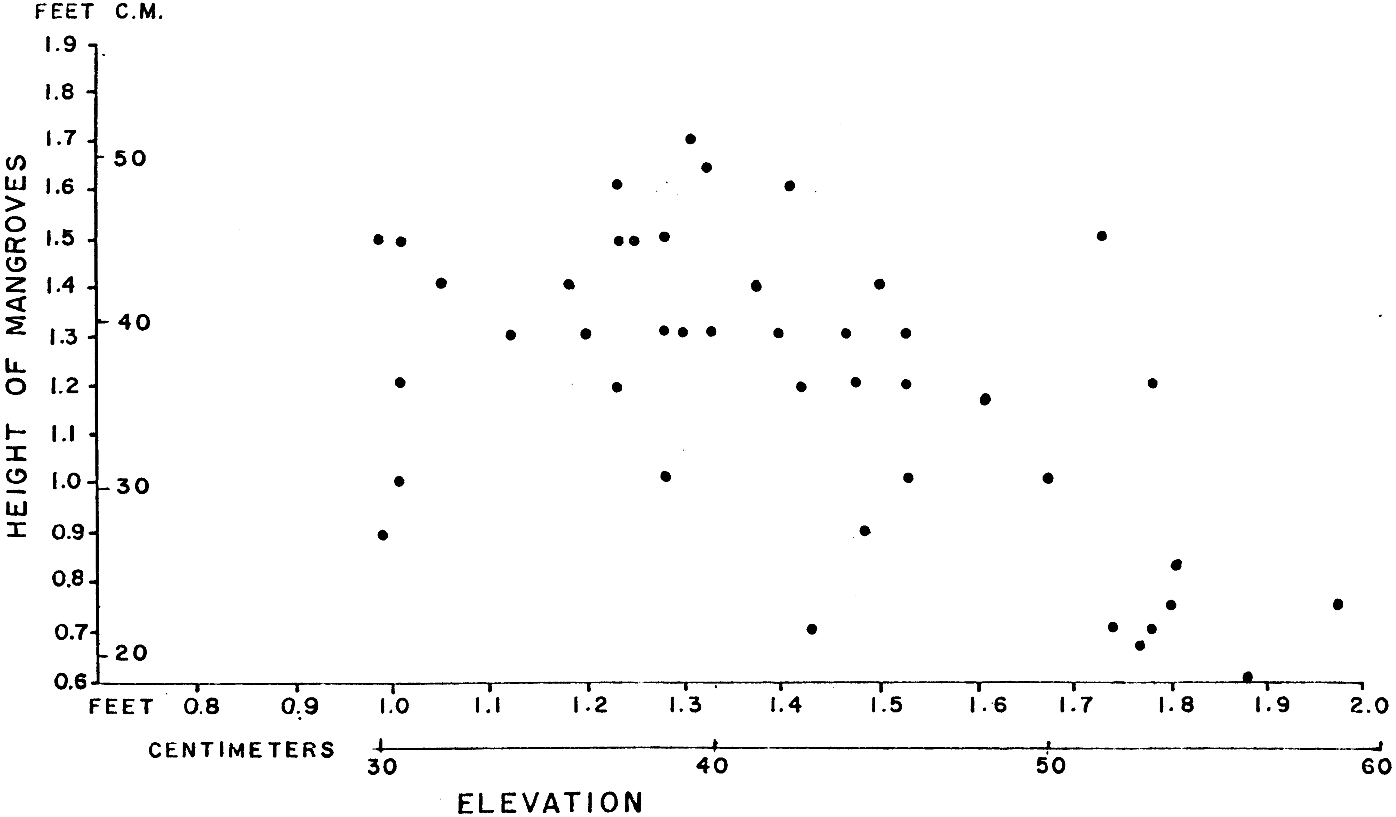


Figure 2. Mangrove Height vs. Elevation

The following Table I, based on a field survey of January, 1983, illustrates the relative growth as a function of elevation and soil conditions over the site.

TABLE I

Mangrove Characteristics	Elevation in CM & Ft. Above 0.0 NGVD	Site Characteristics
Volunteer crabs, small fish, wading birds, warm water, large O_2 production, NO MANGROVES	24 -30 cm (0.8-1.0')	Relatively deep 10-15 cm (4"-6') open water devoid of mangroves.
Volunteer oyster clutches, FEW and STUNTED MANGROVES 30.5 cm 12" in height maximum	30 - 36 cm 1.0-1.2'	Ponded open waters 5-10 cm (2"-4") deep.
WELL DEVELOPED MANGROVE plugs, multiple leaf growth. Mangrove height 38-56 cm (15"-22")	33 - 45 cm (1.1-1.5')	2.5-5 cm (1" to 2") of standing water to no standing water. High soil moisture.
BEST GROWTH ELEVATION mangrove height 48-56 cm (19"-22") Largest well developed plants multiple leaf growth	39 cm (1.3')	No standing water, good soil moisture.
Volunteer black mangroves and stunted red mangrove plugs	51 - 57 cm (1.7'-1.9')	Poor soil moisture typically the edge of the restoration area.

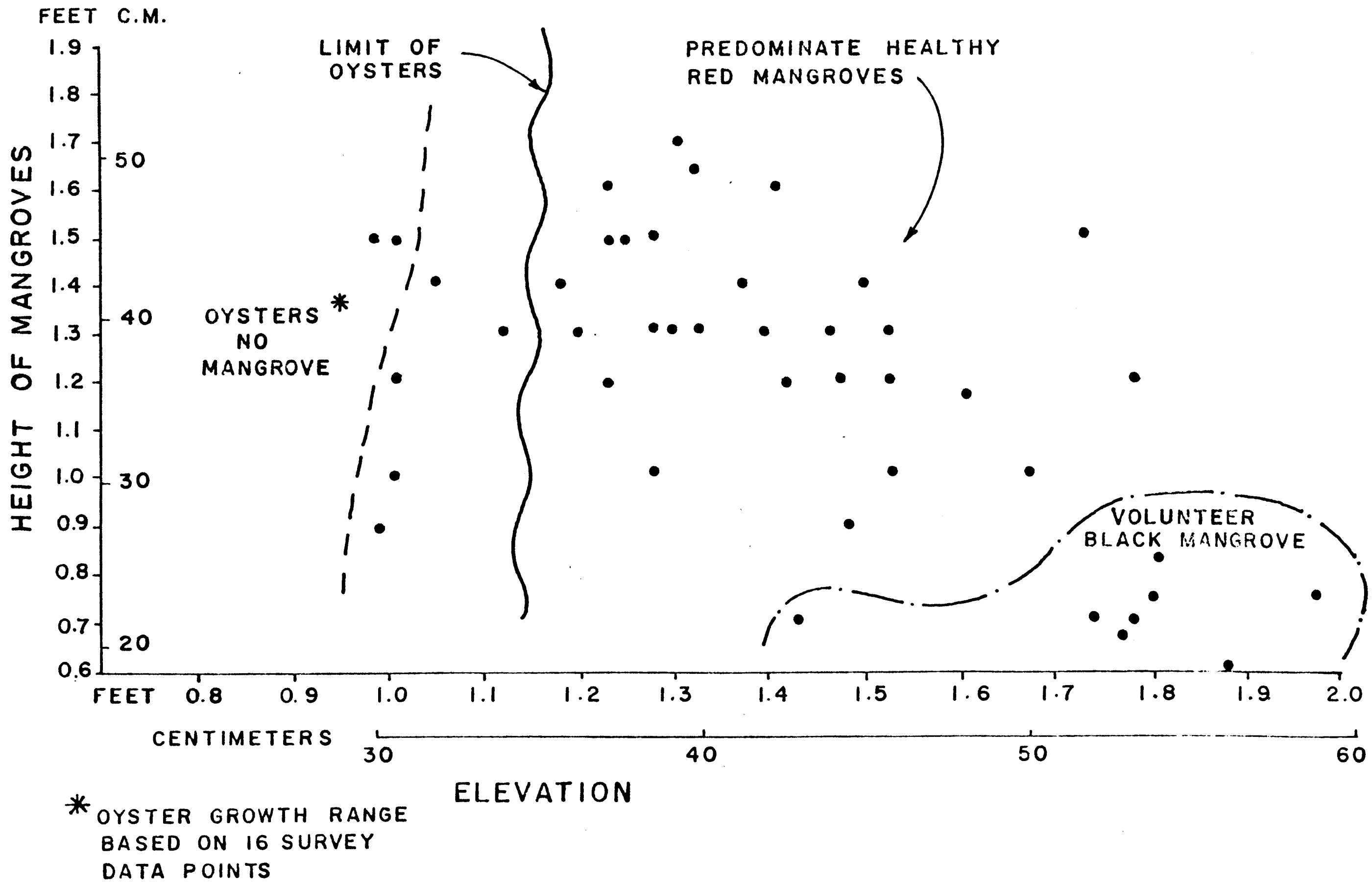


Figure 3. Mangrove Height vs. Elevation Plot

cycle. Successful red mangrove propagule growth covered the widest range but was optimal at a lower elevation than anticipated in well drained areas with consistent high soil moisture. This deviation could be related to normal sediment accumulation which would tend to elevate the surface grade elevation in a mature mangrove stand over a period of time.

Construction tolerances ranged approximately +15 cm (6 inches) above and below grade which is not excessive considering the type of equipment and the problems that are encountered in construction of this type. However, Figure 3 certainly points out the need for extreme care in the grading of these sites.

In order to address the problem of stagnant, standing water described by Davis (1940) it is also important to note that a part of the design included the creation of small flushing channels, a network of vein like channels that would provide an opportunity for water to penetrate into the interior and around the outer perimeter of the restoration site. Soil moisture is extremely critical to successful growth. It is entirely necessary to have moving water in and out of the site.

It may, however, be advantageous, in the future design of restoration areas of this type, to intentionally provide some water impoundment. Where it occurred, unintentionally, these areas exhibited volunteer growth of oyster communities and numerous small fish to remain living in the area. It seems much more conductive to development of the natural succession of a wetland community to provide a variety of depths. Deeper water species can then transition into the more shallow water species. It appears to be a definite benefit to provide some of this type of habitat. Some black mangrove volunteers were expected but the oyster volunteers were an extremely welcomed surprise.

PROJECT COSTS

The total cost of this project was $250,000. The 64,000 cubic yards of fill were removed from the spoil sites and placed on the upland project. This produced a net cost of $3.40 per yard for the fill, which was somewhat less then the cost to transport commercial fill in from the site. Therefore, it was economically viable in terms of the development activity. Further, it allowed the improvement of bay shoreline by adding 15 acres of oxygen production, shallow water and mangrove habitat. Additionally, it allowed the reestablishment of flushing into the surrounding existing mature mangrove, thus having the benefit of adding an additional 30 to 40 acres of flushing to the bay.

CONCLUSION

Optimal growth of red mangrove propagules occurred between elevations +33 to 45 cm (+1.1 to +1.5 ft.) NGVD. Consideration should be given to intentional variation of grade and creation of permanently ponded areas to provide habitat for small fish, wading birds, algae, and oysters. Wetland restoration and creation is demonstrably successful and can and should be a positive element that is encouraged in all types of coastal development activities. It can be made cost effective and where cooperation is achieved between the development activity, the developer and environmental permit agencies, then it has a positive effect on the quality of life, environment and benefits all. The survey results indicate that elevation control is critical, that very close tolerances are required and a qualified, conscientious contractor is necessary to accomplish work of this type.

ACKNOWLEDGEMENTS

I am pleased to have the opportunity to present the results of this work and wish to credit the following individuals and organizations.

Mr. Kevin Erwin, a consulting environmental ecologist in Ft. Myers, Florida, worked closely with us and was extremely important to the success of this project. He assisted in terms of field mapping, determination of design elevation, and in monitoring construction activities.

Mangroves Systems, Inc. and Mr. Robin Lewis, provided contract work on the planting of the mangrove propagules, and collection of the propagules.

We wish to recognize Aberdeen Heavy Equipment Services, who did the excavation, grading and clearing work. Their conscientious efforts were essential to the unqualified success of this project.

Finally, we wish to recognize Mr. Lloyd Sheehan and the Windstar Development Group for their support of this project throughout its entirety.

LITERATURE CITED

Davis, J.H., 1940, The Ecology and Geologic Role of Mangroves in Florida: Carnegie Institute, Washington, Papers from the Tortugas Laboratory, 32 (16), pp. 304-409.

Goforth, H.W., and Thomas, J.R., 1979, Plantings of Red Mangroves (Rhizophora Mangle L.) for Stabilization of Marl Shorelines in the Florida Keys: in Proc. 6th Annual Conference Wetlands Restoration Creation, (Cole, D.P., Ed.), Hillsborough Community College, Tampa, Florida. 1980, p. 207.

Hannan, J., 1975, Aspects of Red Mangroves Reforestation in Florida: in Proc. 2nd Ann. Conf. on Restoration of Coastal Vegetaion in Florida, (ed., R.R., Lewis), Hillsborough Community College, Tampa, Florida; pp. 203.

Lewis, R.R., 1982, Mangrove Forests in Creation and Restoration of Coastal Plant Communities: (R.R. Lewis, ed.), CRC Press, Inc. Boca Raton, Florida.

Teas, H. J., Jurgens, W., and Kimball, M.C., (1975), Plantings of Red Mangroves (Rhizophora Mangle L.) in Charlotte and St. Lucie Counties, Florida, in Proc. 2nd Annual Conference Restoration Coastal Vegetation Florida, (Lewis, R.R., Ed.), Hillsborough Community College, Tampa, Florida, p. 132.

Teas, H.J., 1976, Productivity of Biscayne Bay Mangroves: Past, Present, Future: in Biscayne Bay, (Ed. A. Thorhog). University of Miami Seagrant Special Report No.5, University of Miami Coral Gables, Florida; p. 315

Teas, H. J., 1977, Ecology and Restoration of Mangrove Shorelines in Florida: Environmental Conservation, Vol. 5, No.1, Spring, 1977; The Foundation for Environmental Conservation, Switzerland, p. 51-58.

THE ATCHAFALAYA RIVER DELTA: A "NEW" FISHERY NURSERY, WITH RECOMMENDATIONS FOR MANAGEMENT

Bruce A. Thompson

Linda A. Deegan

Coastal Ecology and Fisheries Institute

Center for Wetland Resources

Louisiana State University

Baton Rouge, LA 70803

ABSTRACT

The dynamics of coastal Louisiana's fish fauna have been greatly influenced by the switching cycle of growth and decay of river deltas and the change in hydrologic and salinity regimes that accompany these cycles. Diversion of Mississippi River water down the Atchafalaya River is forming a new delta and creating wetlands in the northeast section of the Atchafalaya Bay. These wetlands represent the emerging subaerial phase of a new Mississippi River delta lobe that will continue to expand.

Earlier reports hypothesized that as freshwater discharge into Atchafalaya Bay increased, the nursery capacity would be lost. In addition, it was suggested that cold associated with winter and spring floods would significantly depress the productivity of the system.

Data from the present study strongly suggest that with the emergence of the delta islands, the region has regained its nursery capacity. Most of the nekton found in the delta are young-of-the-year and juveniles. Distribution and abundance of young during periods of cold water show a dependence on island configuration. Certain growth patterns create morphologies that favor the formation of "temperature refugia" for nekton and thus present the possibility of having great management potential. Appraisal of the configuration of present habitats functioning as refugia indicate artificial creation is possible from thoughtful placement of dredge spoil from the Atchafalaya navigation channel.

INTRODUCTION

The main coastal area of Louisiana was formed by successive delta complexes as a result of significant changes in the course of the Mississippi River over the past 7000 years (Kolb and Van Lopik 1966, Frazier 1967). As time progresses, each delta goes through a cycle of growth, abandonment, and finally deterioration (Gagliano and Van Beek 1975). New delta complexes develop over older deltaic lobes with growth in one area coexisting with deterioration in other areas.

The dynamics of the coastal Louisiana fish fauna have been greatly influenced by these delta cycles and the alteration of the configuration of the coastal zone. At present, the biomass of fish communities in this region of the Mississippi Delta is very large and the area has been called the "fertile fisheries crescent" (Gunter 1967).

The Atchafalaya River is currently forming a new delta at its mouth in the northeast section of the Atchafalaya Bay (Figure 1). This is unique along the present Louisiana Gulf coast where coastline retreat is now considered a major problem (Gagliano and Van Beek 1975; Craig, Turner and Day 1979; van Heerden and Roberts 1980a).

Examination of the fish assemblages currently utilizing the delta region provides insight into geomorphological processes and human interventions that may affect the levels of productivity of our biological (principally fishery) resources. Continued studies, resulting in suggestions for management alternatives in the area, could enhance our capability towards using techniques to increase our fishery resources or at least minimize deleterious effects.

The general objective of this study was to determine the fishery potential of the Atchafalaya delta. Specifically, the study was aimed at: 1) determining which species presently use the delta region; 2) which species primarily used it as a nursery; 3) the differential habitat utilization of these species; and 4) management possibilities to enhance the fish fauna.

AREA DESCRIPTION

General

The Atchafalaya River branches from the main Mississippi River near Simmesport, Louisiana, and flows 226 km before entering the northeastern section of the Atchafalaya Bay (Figure 1). Van Beek et al. (1979) gives a detailed description of the basin environment. It is a broad floodplain between the natural levees of the Mississippi-Lafourche and Teche-Mississippi systems. The lower basin is a wide swamp forest, grading to marshes in the Fourleague Bay area to the east of the delta (Figure 1).

Atchafalaya Delta

The delta, at present, is about 6 km across and occupies about 1200 ha of land. Adams and Baumann (1980) and Baumann and Adams (1981) provide a review of the geological and hydrological setting for the formation of the Atchafalaya delta. Figure 2 shows the present configuration of the Atchafalaya delta and van Heerden and Roberts (1980a and b) and van Heerden, Wells and Roberts (1981) discuss the processes responsible for the shape and extent of these new islands.

MATERIALS AND METHODS

Sampling started in March 1981 and ended November 1982. Based on an examination of habitat types in the delta, a series of sampling stations was established, covering both natural and artificial areas (Figure 2).

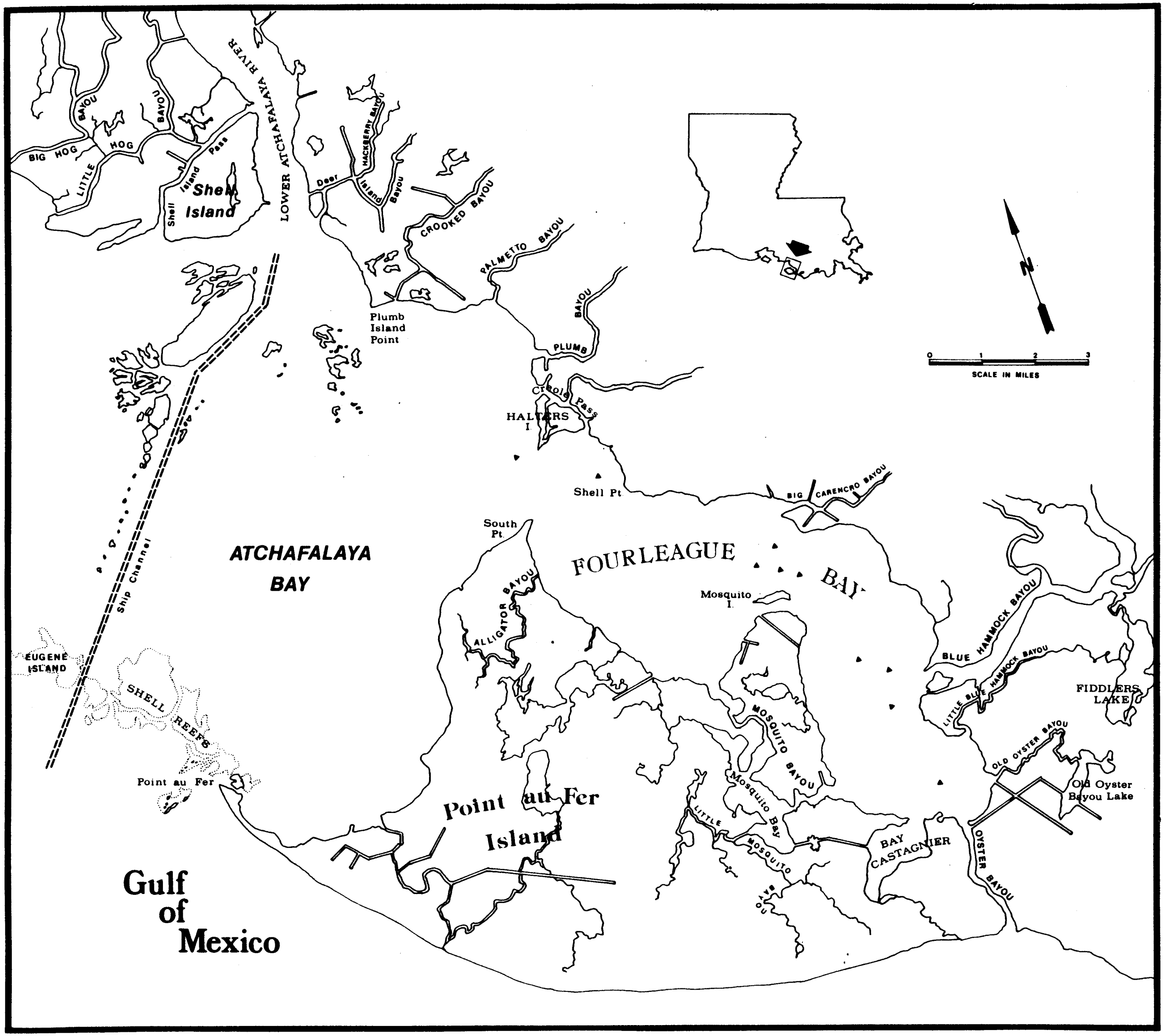


Figure 1. Atchafalaya Bay Region.

At stations Two and Eight, replicate 10 minute samples using a 4.88 m trawl with a 19 mm mesh body and 5 mm mesh cod-end, were made monthly. All other stations were sampled monthly with a 15.24 m, 5 mm mesh bag seine. During low water periods, a 3.1 m, 5 mm mesh straight seine was used. An attempt was made, where possible, to keep the sampling areas uniform to reduce sampling bias in station comparison. Samples were preserved in 10% formalin in the field and returned to the laboratory for processing. Identification of all young-of-the-year fishes to species was emphasized in order to evaluate the delta as a nursery.

Conductivity, salinity, and water temperature were determined with a YSI model 33. A Secchi disc was used to measure water clarity.

Each fish species was classified into one of four ecological affinity groups (patterned after McHugh 1967) to help understand the use of the delta by different components of the fish community.

Freshwater (FW) - affinities primarily with freshwater (below 0.5 ppt.). Spawns in fresh water; have slight to moderate salinity tolerances.

Estuarine (ES) - spends most of life cycle within the estuary. Generally has greatest spawning in the estuary; wide salinity tolerances.

Estuarine-Marine (ESM) - time spent in estuary is primarily as young of the year. Spawn in either nearshore or offshore marine habitat; very wide salinity tolerances; species often referred to as "estuarine dependent."

Marine (MA) - spends most of life in nearshore or offshore marine habitat; generally intolerant of freshwater conditions; spawns in marine habitat.

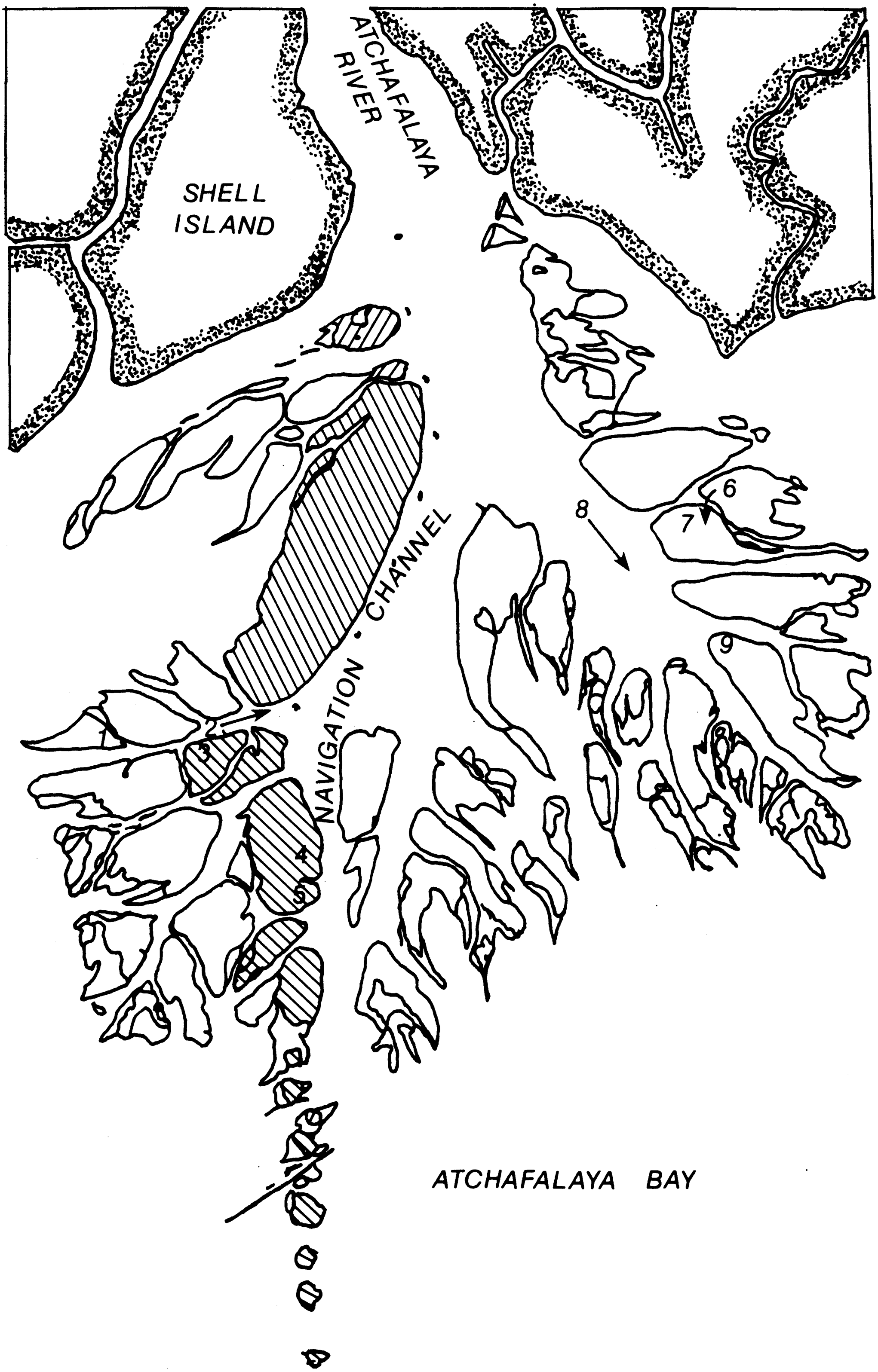


Figure 2. Atchafalaya Delta islands showing details of land building, with location of dredge islands (slash lines). Numbers show habitats sampled for nekton.

RESULTS AND DISCUSSION

Estuarine Nurseries

Gunter (1945, 1950), Turner and Johnson (1973), Peterson and Peterson (1979) and Dovel (1981) have discussed the importance of inland coastal waters as fish nurseries. The concept of "estuarine dependent" species has arisen from the widespread pattern of many nekton species spending at least some part of their first year of life in estuaries. An area must offer three major environmental factors to be a significant nursery (Joseph 1973). These are:

1) physiologically suitable in terms of chemical and physical factors.
2) an abundance of food and/or minimum competition for critical resources.
3) protection from predation

Using these same criteria, we can expand the estuarine nursery concept to include those estuarine fishes that complete their entire life cycles there and freshwater species that utilize the area.

The Atchafalaya delta is the nursery ground for at least 56 species of fish (Table 1). Estuarine-marine forms are the most numerous with 19 species, followed by freshwater (17) and estuarine (15), with only five marine species. In terms of numerical abundance, estuarine and estuarine-marine species predominate (Table 2). Three very abundant estuarine species, *Cyprinodon variegatus*, *Fundulus grandis*, and *Menidia beryllina* spawn in the delta and account for most of the true larvae found there.

Many of our valuable recreational and commercial fish, mostly freshwater and estuarine-marine, are found as postlarvae and juveniles in among the delta islands (Table 1). Gulf menhaden (*Brevoortia patronus*), striped mullet (Mugil cephalus), Atlantic croaker (Micropogonias undulatus) and blue catfish (Ictalurus furcatus) are the most abundant (Table 2).

The concept that this area is a "new" nursery area needs further explanation. In the Atchafalaya system, a logical hypothesis would be that the three past delta cycles affecting this region (Frazier 1967) produced conditions not unlike what the fish fauna is presently experiencing. Therefore, the local fish populations could be "pre-adapted" from the outcome of past geological events to be very tolerant of environmental changes, particularly salinity. The Mississippi deltaic plain may have local stocks or populations of many marine or estuarine-marine fish species that are "especially tuned" to exist in low salinity regimes.

Recent historical data from the 1950s (Hoese 1976) indicate the fish community of the Atchafalaya Bay to have been primarily estuarine-marine and marine. The estuarine-marine species reported by Hoese (1976) most likely used Atchafalaya Bay as a nursery area in a manner similar to their use of other Louisiana estuaries (Gunter 1938a,b,c, 1957; Gunter and Shell 1958; Thompson and Verret 1980). Studies of the physiography of the bay during this same period (Shlemon 1975) showed that the Atchafalaya Bay bottom had changed little since 1858 with the region being a relatively shallow (2-m depth) open-water bay.

Habitat and Management

Sampling stations in the delta (Figure 2) were chosen to represent the differential habitat usage by the fish fauna. Major factors used in station selection were: 1) natural vs. artificial substrates; 2) degree of openness or accessibility to the main channel; 3) size of the inner island channel; 4) salinity, temperature, and turbidity regime; and 5) range of water depth. It should not be surprising that many of these

Table 1. Fishes using the Atchafalaya delta as a nursery.[1]

Freshwater (17)

Dorosoma cepedianum
D. petenense
Cyprinus carpio
Ictiobus bubalus
Ictalurus furcatus
I. punctatus
Pylodictis olivaris
*Gambusia affinis**
Heterandria formosa
Poecilia latipinna
Morone chrysops
M. mississippiensis
Lepomis macrochirus
L. microlophus
L. punctatus
Micropterus salmoides
Aplodinotus grunniens

Estuarine (15)

*Cyprinodon variegatus**
*Fundulus grandis**
*F. jenkinsi**
F. pulvereus
Lucania parva
Membras martinica
*Menidia beryllina**
Lagodon rhomboides
*Dormitator maculatus**
Eleotris pisonis
*Evorthodus lyricus**
*Gobionellus boleosoma**
G. oceanicus hastatus
*G. shufeldti**
Trinectes maculatus

Estuarine/Marine (19)

Elops saurus
Brevoortia patronus
Anchoa mitchilli
Arius felis
Bagre marinus
Eucinostomus argenteus
Archosargus probatocephalus
Bairdiella chrysoura
Cynoscion arenarius
C. nebulosus
Leiostomus xanthurus
Micropogonias undulatus
Pogonias cromis
Sciaenops ocellatus
Stellifer lanceolatus
Mugil cephalus
Citharichthys spilopterus
Paralichthys lethostigma
Archirus lineatus

Marine (5)

Myrophis punctatus
Caranx hippos
Peprilus burti
Prionotus tribulus
Symphurus plagiusa

[1]Nursery defined by presence of larvae, postlarvae, small juveniles or ripe adults(*).

Table 2. Fifteen most abundant fish collected in Atchafalaya delta, March 1981 - March 1982.

	SEINE	TRAWL	TOTAL NUMBER	PERCENT TOTAL	ECOL. AFFINITY
1. Cyprinodon variegatus - Sheepshead Minnow	42,081	0	42,081	24.4	ES
2. Brevoortia patronus - Gulf Menhaden	32,957	3	32,960	19.1	ES/M
3. Anchoa mitchilli - Bay Anchovy	32,106	514	32,710	18.9	ES/M
4. Fundulus grandis - Gulf Killifish	20,013	0	20,013	11.6	ES
5. Mugil cephalus - Striped Mullet	11,126	1	11,127	6.4	ES/M
6. Micropogonias undulatus - Atlantic Croaker	8,455	436	8,891	5.2	ES/M
7. Ictalurus furcatus - Blue Catfish	227	4,140	4,367	2.5	FW
8. Menidia beryllina - Inland Silversides	4,223	0	4,223	2.4	ES
9. Citharichthys spilopterus - Bay Whiff	2,207	45	2,252	1.3	ES/M
10. Lucania parva - Rainwater Killifish	2,138	0	2,138	1.2	ES
11. Gobionellus shufeldti - Freshwater Goby	1,975	9	1,984	1.1	ES
12. Gambusia affinis - Mosquitofish	1,826	0	1,826	1.1	FW
13. Gobionellus boleosoma - Darter Goby	1,786	2	1,788	1.0	ES
14. Cynoscion arenarius - Sand Seatrout	982	250	1,232	0.7	ES/M
15. Membras martinica - Rough Silversides	987	1	988	0.6	
Totals	163,179	5,401	168,580	97.5	

factors were found to be interrelated in their influence of the fish assemblages.

The salinity regime in the delta is almost freshwater for the entire year (Table 3). The stations on the eastern side of the delta (7-9) are in a direct line with the river flow and are fresh all year. Maximum

Table 3. Salinity (ppt) in the Atchafalaya delta.

STA.	1	2	3	4	5	7	8	9
MONTH								
MAR	FW	FW	FW	FW	FW	FW	FW	FW
APR	FW	FW	FW	FW	FW	FW	FW	FW
MAY	FW	FW	FW	FW	FW	FW	FW	FW
JUL	FW	FW	FW	FW	FW	FW	FW	FW
AUG	1.5	1.2	1.6	1	1	FW	FW	.5
SEP	FW	FW	.5	.6	.6	FW	FW	FW
OCT	1	1	.7	1.5	1.5	FW	FW	FW
NOV	.9	.8	FW	.5	1	.5	FW	FW
DEC	FW	FW	FW	FW	.6	FW	FW	FW
JAN	FW	FW	FW	FW	FW	FW	FW	FW
FEB	FW	FW	FW	FW	FW	FW	FW	FW
MAR	FW	FW	FW	FW	FW	FW	FW	FW

FW = Less than 0.5 ppt

salinity found was at Sta. 3, a semi-enclosed island pond. The small amount of saline water that reaches the delta between August and November is caused by the low river flows, strong south winds off the Gulf, and fall Gulf of Mexico high water level.

Water temperature for each sampling station also shows the direct influence of Atchafalaya River water (Table 4). Stations with the most direct influence of Atchafalaya River are depressed the most by the cold winter waters. Hoese (1981) discussed this problem for the region and predicted suppressed fishery productivity because of the extreme cold waters.

Table 4. Water temperature (C°) in the Atchafalaya delta.

STA.	1	2	3	4	5	7	8	9
MONTH								
MAR	18	15	19.5	M	M	13	22	12.8
APR	23	21	28	33	23	24	24	24
MAY	23.5	22	31	35	21	25	23	23
JUL	31	31	31.5	32	30	35	35	35
AUG	29	29	32	37	30	28	27	38
SEP	27	26	30	32	26	29.5	25.5	37
OCT	23	23	22	25	23	19.5	19.5	19.5
NOV	19	19	22	25	19	21	17	20
DEC	12.5	15	15	26	15	13	14	14
JAN	9	12.5	22	22	9	14	9	10.5
FEB	12.5	7	14	22	6.5	12	11.5	8
MAR	15	14.5	17	23.5	15.5	14	14.5	14

Data from the present study strongly suggest that with the emergence of the delta islands, certain protected areas are now acting as "temperature refugia" against cold, winter riverine waters. The habitats arranged by degree of "openness" or direct influence of the river water are:

	Most Exposed					Most Protected
Sta.	5 -	1 -	9 -	4 -	7 -	3
Species	31	38	37	23	38	48
$\bar{x}$ No. Species	10.5	11.9	13.7	6.5	15.5	15.7
$\bar{x}$ No. Specimens	2184	711	1048	2014	3579	4414
Min. Temp.	6.5	9	8	22	12	14

A significant correlation was found between water temperature and total number of fish species (ρ=.3152, p=.0005, n=119) and total number of specimens collected at each station (ρ=.3279, p=.0229, n=48). The response by the fish fauna to the cold water can be seen by the numbers of species and specimens taken each month (Tables 5 and 6). During the periods of extreme cold water (January and February), shoreline habitats heavily utilized by fish at other times, became nearly barren. The deeper channels were also sampled and their reduced diversity and productivity (only 11 specimens of a single species) during this period indicate that the deeper waters are not protection against the negative influence of the coldest riverine flow. These riverine waters force out all but the most cold-tolerant species, significantly reducing the evenness (J of species diversity index), this being inversely correlated (ρ=.3224, p=.0003, n=119) with water temperature. During periods of warmer air temperatures in the delta the protected shallows warm quickly, but this warming does not extend into the riverine-influenced areas, where the cold water overrides any potential water temperature increase. Habitats isolated from direct influence of the river maintained healthy, even very large fish concentrations (see Sta. 7, January 1982). The two areas that were the most protected (Sta. 7 and 3) maintained the highest populations (Tables 5 and 6) giving support to the concept of this being the type of habitat most valuable to form either through natural deltaic processes or the activities of man.

Van Heerden (1983) discussed the processes responsible for the creation and subsequent isolation of the secondary and tertiary delta channels that are the primary protected nursery habitat for fishes and Johnson and McGuiness (1975) and Schnick et al. (1982) reviewed many of

Table 5. Number of fish species collected by seine in the Atchafalaya delta, March 1981 - March 1982.

Station	Month M	A	M[1]	J	A	S	O	N	D	J	F	M	Sta. $\bar{X}$
1	10	13	10	19	10	17	18	12	9	7	5	13	11.9
3	12	23	21	17	19	10	12	17	14	11	13	19	15.7
4	--	15	15	7	2	3	6	7	4	3	5	4	6.5
5	--	13	16	14	10	4	11	15	11	8	3	10	10.5
6	10	16	17	----------------------station discontinued--------------------									
7	10	15	16	21	12	21	15	21	17	15	10	13	15.5
9	--	13	21	19	18	10	23	17	11	8	6	5	13.7
Month $\bar{X}$	10.5	15.4	16.5	16.2	11.8	10.8	14.2	14.8	11.0	8.7	7.0	10.7	
Month Total	14	35	37	32	28	28	35	31	25	22	18	26	

[1]No sample made June 1981.

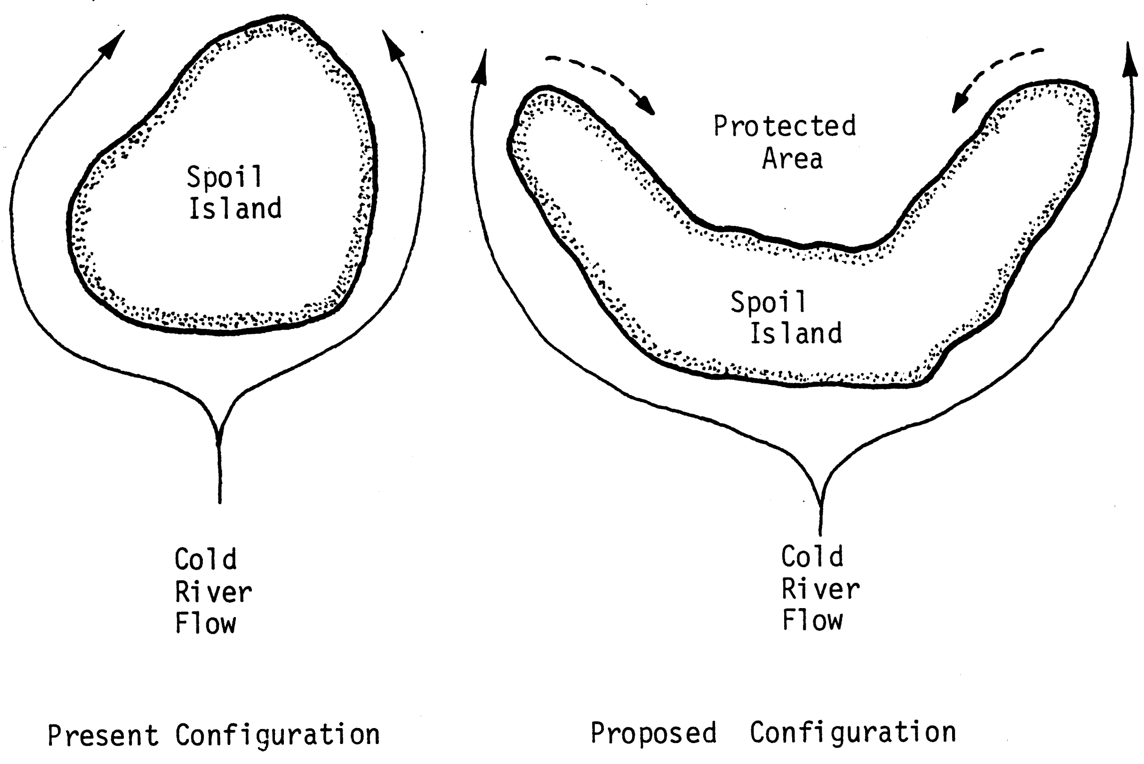
Table 6. Number of fish collected by seine in the Atchafalaya delta, March 1981 - March 1982.

Station	Month M	A	M[1]	J	A	S	O	N	D	J	F	M	Sta. Tot.	Sta. $\bar{X}$
1	781	1682	140	385	105	3595	1031	352	228	141	7	85	8532	711
3	825	32312	2108	759	1982	1561	1652	1718	1784	3193	616	4452	52962	4414
4	--	7685	8383	1411	755	1019	466	804	1127	66	137	301	22154	2014
5	--	3886	2640	2388	5642	3477	969	3292	539	167	5	1016	24021	2184
6	346	1984	2597	----------------------station discontinued----------------------										
7	1205	1817	938	1548	288	1593	756	1223	1118	29865	913	1688	42952	3579
9	--	763	2346	2789	554	2928	480	769	731	94	10	61	11525	1048
Mo. $\bar{X}$	789	7161	2736	1547	1554	2362	892	1360	921	5588	281	1267		
Mo. Total	3157	50129	19152	9280	9326	14173	5354	8158	5527	33526	1688	7603		

[1]No sample made June 1981.

the aspects of navigational dredging and stressed the importance of understanding the local physical, chemical, and biological processes critical to the area. These two studies and Kennedy et al. (1979) listed four main criteria in the design of dredge spoil islands. These are: 1) size--smaller (5 to 25 acres) are more likely to have rapid ecological development; 2) configuration--water currents and elevation must be taken into account to produce as diverse an island as practical under local conditions; 3) substrate--the type of substrate may not be beneficial to all types of biological resources; 4) elevation--the type of vegetation desired and biological resource using the area should determine the elevation variation.

Figure 3.



It is recommended that any future navigation channel dredging that occurs in the Atchafalaya delta deposit the spoil materials in a manner that creates more protected and more diverse habitats. Figure 3 presents a diagrammatic comparison between present spoil island configuration and a suggested altered morphology that would provide habitat with reduced cold water riverine influence. Other island shapes should also be investigated for their engineering feasibility. The fishery resources of the delta need areas of "temperature refugia" that function as stabilizing factors and lessen the impact of cold river water during the most critical time of the year for many young-of-the-year nekton utilizing the delta.

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LITERATURE CITED

Adams, R. D. and R. Baumann. Land building in coastal Louisiana: emergence of the Atchafalaya Bay delta. Sea Grant Publ. No. LSU-T-80-02. 1980.

Baumann, R. and R. Adams. The creation and restoration of wetlands by natural processes in the lower Atchafalaya River system: possible conflicts with navigation and flood control management. Pp. 1-24 in R. Stovall, ed. Proc. 8th Ann. Conf. Wetlands Rest. Creat. Hillsborough Comm. Coll., Tampa, Fla. 1981.

Craig, N., R. Turner, and J. Day. Land loss in coastal Louisiana (U.S.A.). Environ. Manage. 8(2):133-144. 1979.

Dovel, W. L. Ichthyoplankton of the lower Hudson estuary, New York. N.Y. Fish and Game J. 28(1):21-39. 1981.

Frazier, D. E. Recent deltaic deposits of the Mississippi River, their development and chronology. Trans. Gulf Coast Assoc. Geol. Soc. 17:287-315. 1967.

Gagliano, S. M. and J. L. van Beek. An approach to multiuse management in the Mississippi Delta system. Pp. 223-228 in M. L. Broussard, ed. Deltas, Models for Exploration. Houston Geol. Soc. 1975.

Gunter, G. Notes on invasion of fresh water by fishes of the Gulf of Mexico, with special reference to the Mississippi-Atchafalaya River system. Copeia (2):69-72. 1938a.

Gunter, G. The relative numbers of species of marine fish on the Louisiana coast. Amer. Nat. 72:77-83. 1938b.

Gunter, G. Seasonal variations in abundance of certain estuarine and marine fishes in Louisiana, with particular reference to life histories. Ecol. Monogr. 8(3):314-346. 1938c.

Gunter, G. Studies on marine fishes of Texas. Publ. Inst. Mar. Sci., Univ. Texas 1(1):7-190. 1945.

Gunter, G. Distributions and abundance of fishes on the Aransas National Wildlife Refuge, with life history notes. Publ. Inst. Mar. Sci., Univ. Texas 1(2):89-101. 1950.

Gunter, G. Some relationships of estuaries to the fisheries of the Gulf of Mexico. Pp. 621-638 in G. H. Lauff, ed. Estuaries. Amer. Assoc. Adv. Sci., Spec. Publ. 83. 1967.

Gunter, G. and W. Shell. A study of an estuarine area with water level control in a Louisiana marsh. Proc. La. Acad. Sci. 21:5-34. 1958.

Hoese, H. D. Study of sport and commercial fishes of the Atchafalaya Bay region. U.S. Fish and Wildlife Serv. Rept. 53 pp. 1976.

Hoese, H. D. Some effects of fresh water on the Atchafalaya Bay system. Pp. 110-124 in R. D. Cross and D. L. Williams, eds. Proc. Natl. Symp. Freshwater Inflow Estuaries. Vol. II. U.S. Fish and Wildlife Serv. FWS/OBS-81/04. 1981.

Johnson, L. E. and M. V. McGuinness. Guidelines for material placement in marsh creation. U.S. Army Engr. Waterways Exp. Sta., Vicksburg, Miss. Cont. Rept. D-75-2. 230 pp. 1975.

Joseph, E. B. Analysis of a nursery ground. Pp. 118-121 in A. L. Pacheco, ed. Proc. Workshop on Egg, Larval and Juvenile Stages of Fish in Atlantic Coast Estuaries. Mid. Atl. Coast. Fish. Center. Tech. Publ. No. 1. 1973.

Kennedy, D., K. Jonas, R. Nicklaus, W. Fernholz, J. Holzer, P. Thiel, T. Lovejoy, N. Gulden, and B. Hawkinson. Experimental island creation for habitat enhancement. Fish and Wildlife Work Group, Great River

Environ. Act. Team. U.S. Dept. Int. Fish and Wildlife Serv., Group Rept. 1979.

Kolb, C. and J. Van Lopik. Depositional environments of the Mississippi deltaic plain, southeastern Louisiana. Pp. 17-61 in Deltas and Their Geologic Framework. Houston Geol. Soc. 1966.

McHugh, J. L. Estuarine nekton. Pp. 581-619 in G. A. Lauff, ed. Estuaries. AAAS Publ. No. 83. 1967.

Peterson, C. H. and N. Peterson. The ecology of intertidal flats of North Carolina: a community profile. U.S. Fish and Wildlife Serv. FWS/OBS-79/39. 73 pp. 1979.

Schnick, R. A., J. M. Morton, J. C. Mochalski, and J. T. Beall. Mitigation and enhancement techniques for the Upper Mississippi River System and other large river systems. U.S. Dept. Inst., Fish and Wildlife Serv., Res. Publ. 149. 714 pp. 1982.

Shlemon, R. L. Subaqueous delta formation--Atchafalaya Bay, Louisiana. Pp. 209-221 in M. L. Broussard, ed. Deltas, Models for Exploration. Houston Geol. Soc. 1975.

Thompson, B. A. and S. J. Verret. Nekton of Lake Pontchartrain, Louisiana, and its surrounding wetlands. Pp. 711-864 in J. H. Stone, ed. Environmental Analysis of Lake Pontchartrain, Louisiana, Its Surrounding Wetlands, and Selected Land Uses. CEL, CWR, LSU, Baton Rouge, La. Prepared for U.S. Army Engineers, New Orleans District, Contract No. DACW29-77-C-0253. 1980.

Turner, W. and G. Johnson. Distribution and relative abundance of fishes in Newport River, North Carolina. NMFS-SSRF No. 666. 23 pp. 1973.

van Beek, J. L., A. L. Harmon, C. L. Wax, and K. M. Wicker. Operation of the Old River Control Project, Atchafalaya basin: an evaluation from a multiuse management standpoint. U.S. Env. Protect. Agency. EPA-600/4-79-073. 72 pp. 1979.

van Heerden, I. Deltaic sedimentation in eastern Atchafalaya Bay, Louisiana. Ph.D. Diss., Louisiana State Univ., Baton Rouge, La. 151 pp. 1983.

van Heerden, I. and H. H. Roberts. The Atchafalaya delta: rapid progradation along a traditionally retreating coast (south-central Louisiana). Z. Geomorph. N.F., Suppl.-Bd. 34:188-201. 1980a.

van Heerden, I. and H. H. Roberts. The Atchafalaya delta--Louisiana's new prograding coast. Trans. Gulf Coast Assoc. Geol. Soc. 30:497-506. 1980b.

van Heerden, I., J. Wells, and H. H. Roberts. Evolution and morphology of sedimentary environments, Atchafalaya delta, Louisiana. Trans. Gulf Coast Assoc. Geol. Soc. 31:399-408. 1981.