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Proceedings of The 14th Annual Conference on Wetlands Restoration and Creation

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May 14 - 15, 1987

Hillsborough Community College

President
Dr. Andreas Paloumpis

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Plant City Campus
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PROCEEDINGS OF THE FOURTEENTH ANNUAL CONFERENCE ON WETLANDS RESTORATION AND CREATION

May 14-15, 1987

Sponsored by
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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 organization, colleges and universities, corporations, and environmental groups with an interest in wetlands. These proceedings are a compilation of papers and addresses presented at the Fourteenth Annual Conference.

This year's conference would not have been possible without the assistance and cooperation of Mr. Roy R. "Robin" Lewis, III. Mr. Lewis has been an important contributor since the very first conference, fourteen years ago. We are grateful for his help and participation. Appreciation is also extended to Charles Deusner for providing administrative support for the conference.

The following people also deserve thanks for contributing to the conference and assisting in the preparation of the proceedings for publication: Bettye J. Broxton, Diane Crigger, Fay Crowe, Johnnie Harcleroad, Patricia Schwarzlose, David Walker, Jackie Watford, and Cecelia Weaver.

Thanks are also extended to Robin Lewis, Colleen O'Sullivan, and Bob Whitman for arranging and conducting field trips to wetland restoration/creation sites.

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

To all these people, thank you.

KEYNOTE ADDRESS

DOES MITIGATION WORK?: EPA'S WETLANDS RESEARCH PROGRAM IS CHECKING

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EPA'S WETLANDS RESEARCH PROGRAM

The U.S. Environmental Protection Agency (EPA) adopted in January of 1986 a Wetlands Research Plan (Zedler & Kentula 1986). The Plan outlined research to (1) define the water quality functions of wetlands; (2) develop a method to assess and predict the cumulative impacts associated with the incremental loss of wetlands; and (3) provide support for Agency personnel faced with decisions concerning mitigation for permitted wetland losses through creation or restoration.

The research program was funded in April 1986. The Environmental Research Laboratory in Athens, Georgia was given responsibility for the water quality research. Responsibility for the cumulative impact and mitigation research was given to the Environmental Research Laboratory in Corvallis, Oregon. The Mitigation Research Component is the subject of this paper.

THE MITIGATION RESEARCH COMPONENT

The creation or restoration of wetlands is often required as compensation for wetland losses permitted under Section 404 of the Clean Water Act. Created and restored wetlands represent a range of conditions and situations. Many are small in size, often less than an acre. However, some, as in the restoration of areas impacted by mining, are hundreds of acres. Many occur in urban areas and reflect the influence of high population densities. Others, mostly the result of highway construction, occur in rural settings. Sometimes attempts are made to create wetlands that look as natural as possible. Others look obviously man-made. Despite the size, setting or appearance, the question remains: How well do these created and restored wetlands replace the ecological functions of the wetlands that were destroyed?

This question is central to the Mitigation Research Component. Although there are other forms of mitigation that could be considered in the program, the effectiveness of mitigation through the creation and restoration of wetlands is of great concern. The number of these projects is constantly increasing, while the science of wetland creation and restoration is considered to be in its infancy.

OVERVIEW

The Mitigation Research Component has two goals. They are to determine the ecological functions of created and restored wetlands, and improve project design. Ultimately, the results of the research will be compiled into a mitigation handbook for Agency 404 personnel. It will be designed to provide guidelines by which to (1) evaluate the probability that a proposed project will succeed, (2) formulate permit conditions, i.e. set goals, and (3) determine if a project met those goals. To gather the needed information, research has been initiated to synthesize existing information and evaluate completed projects.

Research Project #1 - Synthesis of Information

Efforts are underway to synthesize two types of information on wetland creation and restoration. One focuses on the information contained in the literature; the other on that in the 404 permit record.

The literature synthesis will serve as provisional guidance until the first version of the handbook is produced. Since much of the information on wetland creation and restoration is not contained in the scientific literature, the goal is to assemble information from as many sources as possible, including personal experience.

A group of eminent wetland scientists has been commissioned to produce the document. They are primarily people who have worked on various aspects of wetland creation and restoration. Dr. Jon Kusler of the Association of State Wetland Managers and I coordinate the effort.

The document will be composed of two sections. The first will be a series of theme papers covering the wide range of topics of general application to wetland creation and restoration. These topics include succession and stability of created and restored wetlands, an overview of wetland evaluation, and applications for creating wetlands for waterfowl management. The second will be a series of regional reviews. These will discuss the status of the science of wetland creation and restoration for wetlands of various types in different regions of the country. The authors will also identify information gaps and research needs. These will be reviewed by members of the National Wetland Technical Council which will then recommend research priorities for the program.

A compilation of information from the 404 permit record will be used to characterize patterns and trends in permit-related wetland creation and restoration and identify completed projects for evaluation. To facilitate the process, a data management system was designed. The system runs on an IBM-compatible personal computer equipped with DataBase III or III+. Essentially, the commercial program was customized to streamline data entry and, thus, eliminate errors.

During the past year, the system was tested by contractors assembling databases of projects in EPA Region X (Oregon, Washington and Idaho) and California, and freshwater projects in Texas, Arkansas, Louisiana, Alabama and Mississippi. It is now being revised. The software and user's manual will be available by the end of the calendar year.

Preliminary analysis of the databases from Washington and Oregon indicates that over 90% of the projects occur West of the Cascade Mountains and in the vicinity of urban centers. Estuarine intertidal (salt marshes and mudflats) and palustrine (emergent marshes and ponds) are the wetland types most often created or restored. In addition, these databases have been used to select complete projects for evaluation.

Research Project #2 - Evaluation of Completed Projects

Completed wetland creation and restoration projects are being treated as "experiments in progress." The goals are to compare characteristics of created and restored wetlands with those of naturally occurring wetlands and determine how those characteristics change with time. An approach to conducting the evaluations is currently being tested in pilot studies in Washington, Oregon and Florida.

The first step is to identify the "test" population by searching the permit database for groups of completed projects that might form a sampling unit. For example, in Oregon the test population identified is a group of twelve, created, emergent marshes less than a hectare in size, ranging in age from six months to six years, in an urban area of the Willamette Valley.

Next, a set of reference sites are selected. A stratified random sample is made of naturally occurring wetlands with characteristics defined by the test population. Again, in Oregon this was a group of emergent marshes less than a hectare in size occurring in the urban areas of the Willamette Valley.

Finally, the sites are evaluated. Plant community structure, substrate, water quality and hydrologic variables are sampled. Where possible, various methods of sampling the same variable are used. The data obtained will be used to test the usefulness of various methods and the consistency of the results obtained when they are used by different individuals.

RESULTS FROM THE FIRST YEAR OF THE PROGRAM

The following will report the results of the research described above:

Literature Synthesis	Early 1988
Permit Database Software and User's Manual	Fall 1987
Patterns and Trends in the 404 Permit Record--Washington, Oregon, California	Fall 1988
Patterns and Trends in the 404 Permit Record--Freshwater Projects in Texas, Arkansas, Louisiana, Alabama, Mississippi	Early 1989

Procedure for Selection of Reference Sites	Summer 1987
Results of Pilot Study to Test Method to Evaluate Mitigation Sites--Washington and Oregon	Summer 1988
Results of Pilot Study to Test Method to Evaluate Mitigation Sites--Florida	Fall 1986

CONCLUSIONS

EPA's Research Plan set program research goals in relation to the plans and efforts of other groups. Since there are many unanswered questions and limited resources with which to seek the answers, it is important that duplication is avoided. Therefore, the Wetland Research Program has an information transfer component to disseminate its findings and collect those of other researchers. A mailing list is maintained and a flier that briefly describes the Program's progress is distributed. Those interested in being included on the mailing list or in supplying information on their research should contact the Wetland Research Program at the Environmental Research Laboratory, 200 S.W. 35th Street, Corvallis, OR 97333; phone: (503) 757-4666, FTS 4204666.

The first year of EPA's Wetland Research Program has been busy. The products described above are in various stages of completion. However, even with all this activity the answer to the question, "Is mitigation working?," is only beginning to be answered. Hopefully, with the efforts of the Wetlands Research Program and that of other scientists working in the field, the next time a report on the Program's progress is given there will be more of an answer.

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Zedler, J. B. and M. E. Kentula. 1986. Wetlands research plan. EPA/600/3-86-009. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Oregon. National Technical Information Service Accession Number PB86 158 656/AS.

A REVIEW OF BEACH PRISMS: THEIR APPLICATION FOR WETLANDS CREATION UNDER MODERATE TO HIGH ENERGY CONDITIONS

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ABSTRACT

Beach Prisms are modular, precast, preassembled erosion control systems which function as permeable detached breakwaters and affect deposition of sediments suspended in the water column. Triangular prisms, ranging in size from 2-12 feet, are cast from high quality sulfate resistant concrete designed for extended exposure in marine environments. The concrete admixture is a 5000 psi compressive strength ASTM -C.-150 type. Individual castings are post tensioned into 10-25 foot units with a corrosion resistant stainless steel tie-rod system. Integrally cast-in turbulating elements were designed using wind tunnel data generated at Ketron, Inc. These elements serve to induce turbulence into inshore water flow thereby dissipating the kinetic energy, which causes suspended solids to separate from the water by gravity. Sediments are first deposited behind the prisms and eventually out in front as the shoreline profile becomes modified to a gradual run-up configuration. Accreted sediments then may be stabilized using native vegetation. This paper reports on the accretion patterns, rates, sediment characteristics and the feasibility of vegetative stabilization of the accreted material. This cost effective system offers a practical alternative for controlling and stabilizing shorelines in moderate to high energy areas.

INTRODUCTION

The ecological significance of wetlands is widely recognized. Communities of wetlands plants provide food and add structural complexity to shallow coastal ecosystems, thus creating habitats capable of supporting abundant and diverse animal populations (Odum et al. 1984; Martin et al. 1961; Needham 1938). Wetlands are also important for nutrient cycling and exert marked influence on sedimentological processes through the stabilization of subtidal soils (Eleuterius 1985) and the dissipation of tidal and wind-generated wave energy (Ward et al. 1984).

Despite the importance of wetlands for maintaining environmental quality, total acreage of this resource has been greatly reduced. Of the 215 million acres of wetlands existing in the conterminous United States in the mid-1500s, only 99 million acres remained by 1975 (Cowardin et al. 1979). Most of this loss was a direct result of land management practices supporting agricultural, commercial and recreational development (Woodhouse & Knutson 1982). In recognition of the need to preserve wetlands resources, numerous forms of legislation have been enacted by federal, state and local governments to protect existing wetlands and to support efforts to create additional wetland areas (Dillmann 1985).

Maryland has placed a high priority on protecting and augmenting the aquatic resources of the Chesapeake Bay. In 1986, critical areas legislation was passed which restricts residential development along shorelines and requires 100 foot buffer zones of vegetation on farmland bounded by waterways. The Maryland Department of Natural Resources offers a 50-50 cost-sharing program to property owners using non-structural methods of shoreline erosion control (NSSEC). In spite of these efforts over 1,000 acres of wetlands are lost per year of which a significant portion can be attributed by the processes of erosion (Zabawa et al. 1982). In Anne Arundel County, with 468 miles of navigable waterway and the greatest number of registered boats per capita, a principal cause of shoreline erosion and wetlands loss is wave energy generated by recreational boat traffic. Attempts to curtail erosion using structural methods of shoreline stabilization accounts for much of the loss. Through the replacement of potential wetlands habitat with various types of bulkheads and revetments, NSSEC is ineffective for stabilizing shorelines through the constructions of wetlands because of the degree of recreational utilization (Maryland Standards and Specifications 1983).

The creation of wetlands in high energy areas requires the regrading of slopes or preferably the use of some form of artificial permeable breakwater (Silvester 1984). Homeowners are reluctant to sacrifice property for regrading projects and the availability of suitable fill materials for beach nourishment is often limited (U.S. Army Corps of Engineers 1987). Similarly, the materials commonly used for breakwater construction such as tires, concrete rubble and rip-rap are often objectionable in residential settings for reasons of aesthetics, durability and performance (Szuwalski 1981). An alternative is the use of breakwaters having a uniform appearance which are amenable to vegetative plantings (Department of the Army 1984; U.S. Army Corps of Engineers 1971).

Beach Prisms are modular, precast, preassembled erosion control systems which function as permeable detached breakwaters and promote accretion of sediments suspended in the water column. Triangular prisms ranging in size from 2-12 feet are cast from high quality sulfate resistant concrete designed for extended exposure to a marine environment. The concrete admixture is a 5000 psi compressive strength ASTM -C.-150 type. Individual castings are post tensioned into 10-25 foot units with a corrosion resistant stainless steel tie-rod system. Integrally cast-in turbulating elements were designed using wind tunnel data generated at Ketron, Inc. These elements serve to induce turbulence into inshore water flow thereby dissipating the kinetic energy, which causes suspended solids to separate from the water by gravity.

Sediments are first deposited behind the prisms and eventually out in front as the shoreline profile becomes modified to a gradual run-up configuration. Accreted sediments may be stabilized using native vegetation. This paper reports on the accretion patterns, rates, sediment characteristics and the feasibility of vegetative stabilization of the accreted material.

STUDY SITE

Forked Creek (Figure 1) is a small tributary, of the Magothy River, approximately 1 km in length. A natural sand spit extends 2/3 of the distance across the mouth of the creek. A large shoal area exists on the river side of this barrier, a deep water lagoon on the creek side. The lagoon provides safe harborage during storms for adjacent communities, while the shoal provides a favorable environment for the growth of submersed aquatic vegetation. The spit itself was heavily vegetated by *Spartina alterniflora*, *S. patens*, *Solidago sempervirens*, *Scirpus americanus*, and *S. robustus* until the late 1970s. Increased development of the area, including the construction and expansion of marinas and mooring piers, produced a significant increase in the number of boats using the area.

Erosion of the spit accelerated, and by 1985 the spit was reduced to the configuration indicated by the boundaries of vegetation (Figure 2). At that time a boating speed restriction was placed on Forked Creek to reduce wave energy and the entire area was revegetated using *S. alterniflora* and *S. patens* at 8 inch centers. This approach was effective for all areas except a portion where wave energy originating from the river was deflected by an existing wooden groin. Scouring from this diversion threatened the integrity of the spit. Such a loss would have a profound effect on the characteristics of the site, including a loss of valuable shallow water habitat and tidal wetlands.

METHODS

In September of 1986, at the time of prism installation, transects were established to identify beach configuration and relative elevations (Figure 3). Three 10 foot sections of 2 foot beach prisms were placed to buffer wave energetics at the site. Placement was made to negate the energy of waves deflected from the wooden groin and from the exposed northern face fronting the Magothy River. Additional surveys of beach configurations and profile were made in February and April of 1987. Core samples were taken to evaluate changes in the sediment profiles and to measure the particle sizes of existing and accreted deposits in front of, behind, and outside of the test area. Vegetative planting of *S. alterniflora* and *S. patens* were planned for June of 1987 to stabilize accreted sediments.

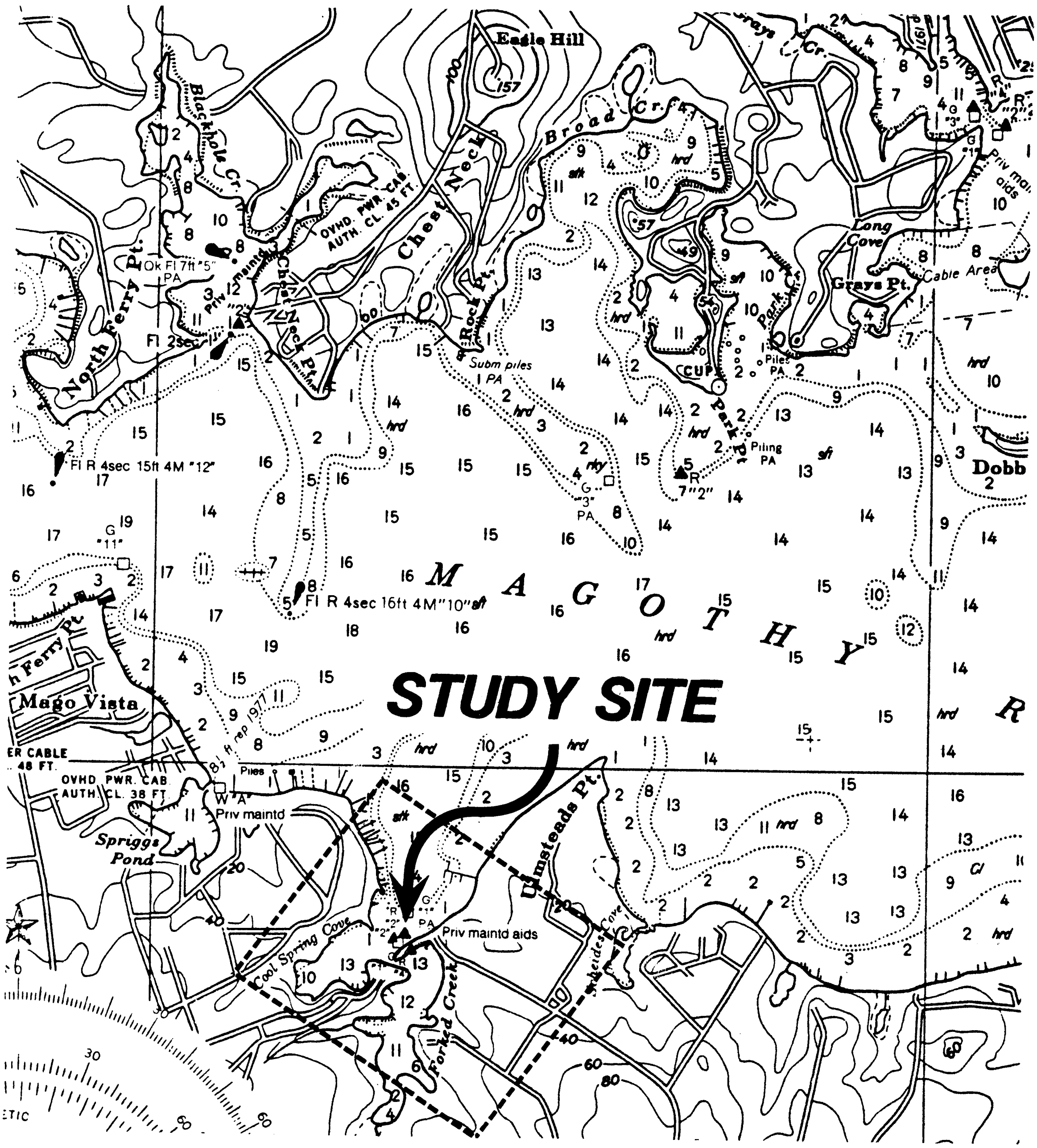

Figure 1. Study site location, Anne Arundel County, Maryland, from NOAA Navigation Chart 12282, 25th ed.

RESULTS

Changes in the beach profile at the control site (transect C, Figure 2) as measured by changes in relative elevation are given in Figure 3. No significant changes in beach elevation above MLW were observed during the test period. Below MLW an average accumulation of 2.5 +/- 1.0 inches of sediment was deposited for a distance of 30 feet. Such variation is consistent with seasonally observed changes at similar sites.

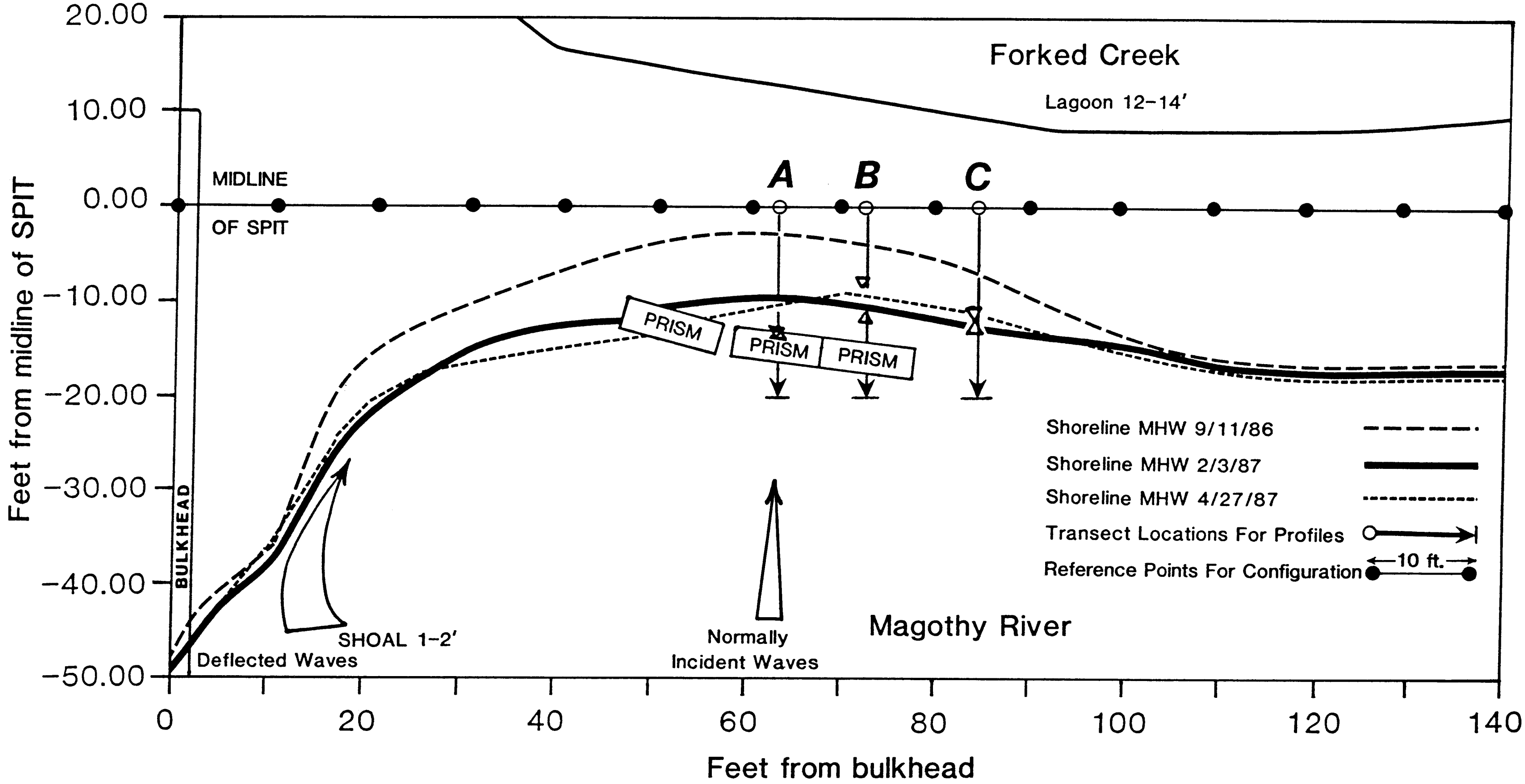

Figure 2. Changes in the shoreline configuration at Ulmstead Beach, 1986-87. Three 3 ft. prisms 10 ft. in length were placed at MLW. Prisms were positioned to minimize normally incident and deflected wave energies. Locations for measuring beach configuration (actual) and profiles (relative elevations) are indicated.

Sediment accretion above the westernmost prism (transect B, Figure 2) was significant at the highest and lowest points averaging 4.8 +/- 1.0 inches (Figure 4). No significant changes were observed at the mid-elevations or below MLW when compared with the control. The sediments accreted were coarse sands which did not differ in size from that of the control site. Sediment deposition was maximal in the fall. A significant loss of sediment occurred in all areas above MLW between the February and April monitorings.

Sediment accretion at the middle prism (transect A, Figure 2) followed the pattern observed at the far prism (transect B, Figure 5) above MLW. Significant sediment deposition also occurred below MLW to a distance of 8 feet from the base of the prism. This is thought to be a result of greater sediment availability caused by current diversions from the wooden bulkhead combined with offshore sources. Again, no differences in sediment composition between this site and the control were observed.

DISCUSSION

Great care must be taken in the evaluation of the efficacy of products serving as offshore breakwaters since performance is largely a function of the characteristics of the test site and the design of a given installation. In addition, complete evaluation can only be made by monitoring sites over a number of years. In this preliminary study the following observations can be made:

1. On the sand substrate where the maximum recommended weight limit of a structure was 15 to 30 lbs./sq. in., the 0.75 lbs./sq. in. foot print pressure of 2 foot Beach Prisms was sufficient to prevent sinking.

2. A net accretion of coarse sand sediments was observed both above and below MLW behind the prisms after 9 months. Similar increases at the control site were not found.

3. Accreted sediments were not stable and could be lost, possibly, as a result of storm surges and spring tides.

4. Accreted sediments were uniformly coarse sand, the predominant soil type of the area.

Beach Prisms as an artificial permeable offshore breakwater offer a number of distinct advantages over more conventional construction materials. Installation is more rapid and less destructive to existing habitats than stabilization methods employing crushed stone and grade change (U.S. Army Corps of Engineers 1986). The completed structure is uniform in appearance and is more resistant to displacement by vandals, icing, and storm surges than comparable structures constructed of rip-rap or tires. The structures are capable of promoting beach enrichment and are compatible with vegetative methods of stabilization. An effort

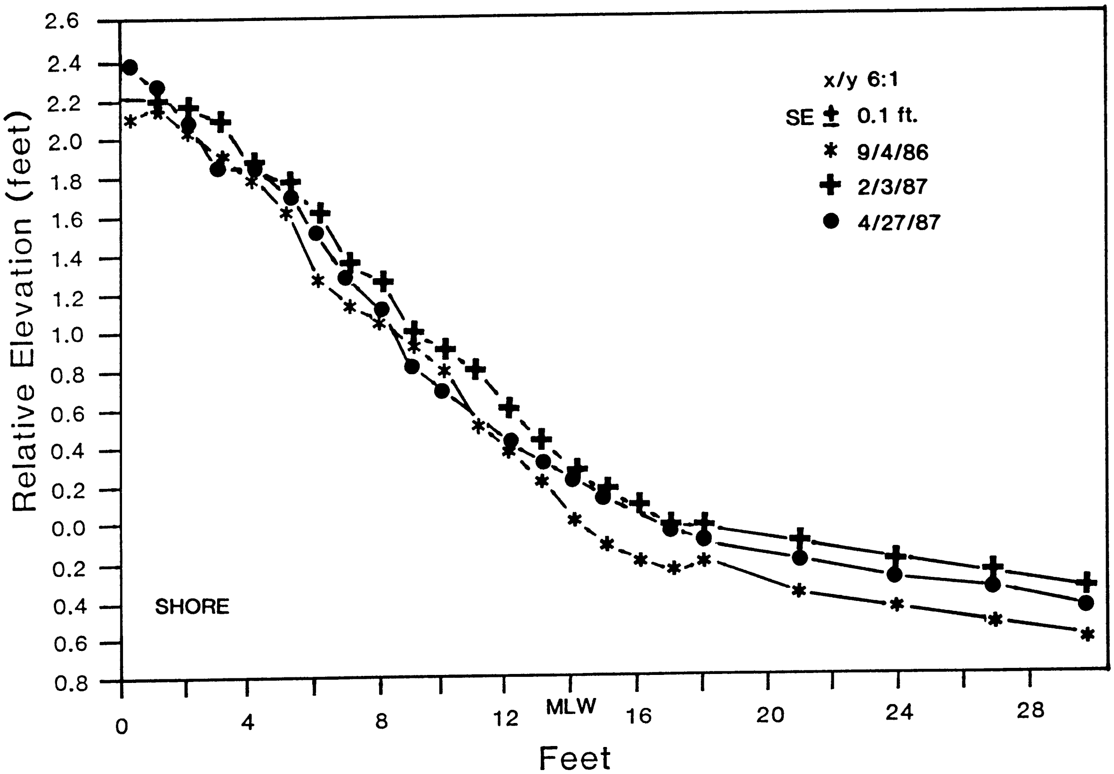

Figure 3. Changes in the beach profile as measured by relative elevation at Transect C, Fig. 2.

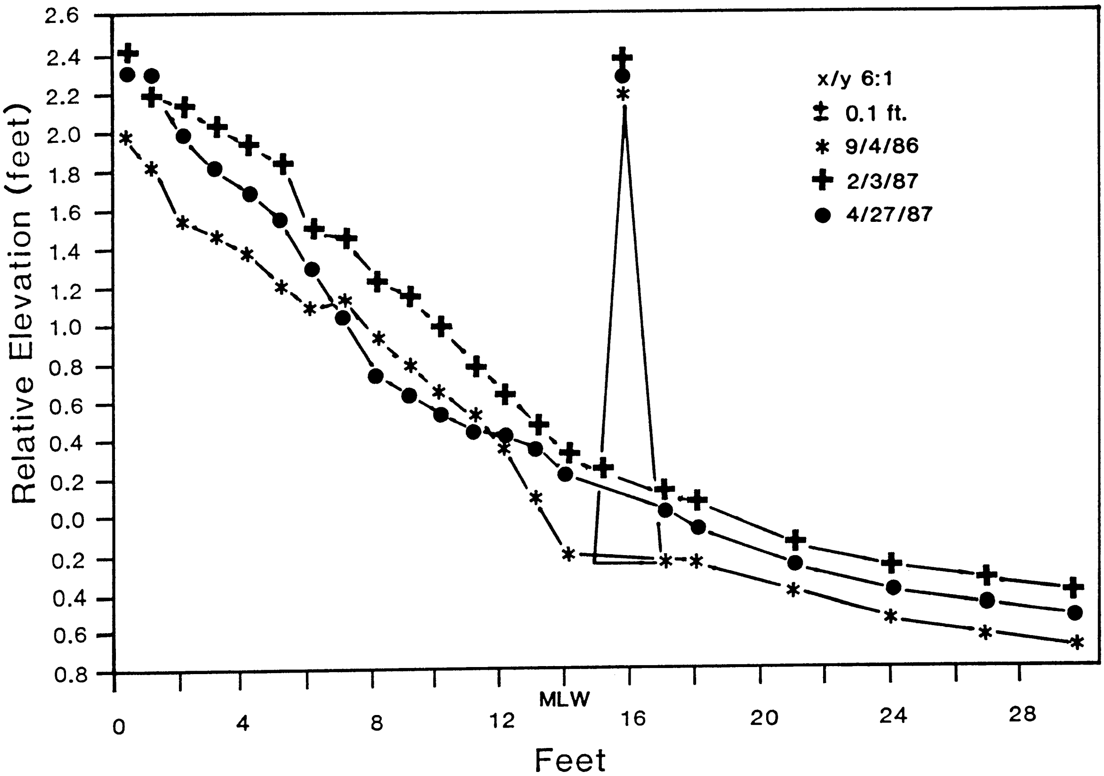

Figure 4. Changes in the beach profile as measured by relative elevation at Transect B, Fig. 2.

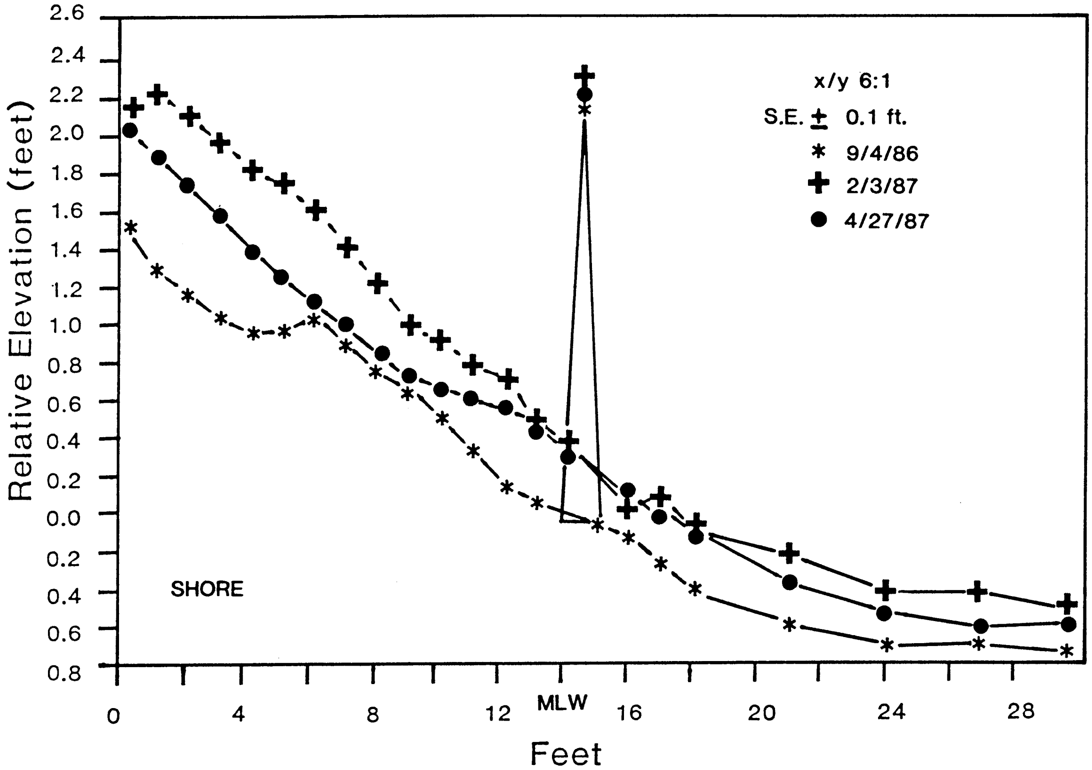

Figure 5. Changes in the beach profile as measured by relative elevation at Transect A, Fig. 2.

is currently underway to revegetate the areas behind the prism installation to minimize the loss of accumulated material during storm events. Preliminary measurements are promising and will be available in the spring of 1988 (Allstock unpublished).

At present the principal disadvantages of Beach Prisms are cost and availability. Cost effectiveness is closely linked with the availability of alternative construction materials and the size of the installation (Gaythwaite 1981; Power et al. 1979). In Maryland, the use of Beach Prisms for shoreline stabilization is becoming increasingly popular. Designs incorporating vegetative stabilization have been accepted by the cost-sharing NSSEC program of the Maryland Department of Natural Resources (Zabawa 1987). Many of these newer installations are being monitored and more comprehensive studies are planned within the next two years. It is hoped that the development of these and similar bio-structural shoreline erosion control methodologies can be used to maintain the natural integrity of heavily trafficked watersheds.

This work was made possible by the donation of the Beach Prisms from Beach Prism's Ltd., Main St., P.O. Box 179, Queenstown, Maryland 21658.

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A QUALITATIVE ASSESSMENT OF WETLANDS RECLAIMED AS NATURAL SYSTEM HABITAT

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ABSTRACT

A majority of large-scale, freshwater wetland reclamation in Florida is associated with projects initiated by the phosphate mining industry. Although that industry for more than six years has been required by law to reclaim mined wetlands on at least a 1:1 acreage ratio, the design, size, quality and success of such projects has varied greatly. This study presents a qualitative habitat assessment synopsis of three wetland reclamation projects established by the International Minerals and Chemical Corporation (IMC) in Polk and eastern Hillsborough Counties since 1978.

The subject wetlands were reclaimed primarily for natural system habitat value rather than for economic or man-dominated purposes. Long-term habitat considerations were incorporated in all phases of the reclamation process. Emphasis in design was given to diversity enhancement for topography, edge, flora, fauna and community interspersion. The focus of the planning process was to promote the establishment of wetlands that will be viable and self-perpetuating ecosystems. In each case, the reclaimed wetland habitat has been linked to existing natural systems to optimize biotic colonization and community integration.

As documented by ecological field assessments and detailed biotic inventories, each of these large-scale wetland creation projects is evolving toward a self-maintaining, natural system that is physically and hydrologically tied to existing, protected wetlands within the region. They are products of acquired reclamation techniques that offer the promise of continuing evolution toward valuable and complex ecosystems within the Florida landscape. Within this context, a holistic philosophy of functional wetland reclamation is discussed.

INTRODUCTION

A large body of work has been produced during the past decade by investigators studying the various aspects of wetland creation in general, and projects related to surface-mined sites in particular. Freshwater wetland creation as part of a mandated, post-mining reclamation process in Florida has been widely reported and summarized. In addition to the annual proceedings of the Conference on Wetlands Restoration and Creation, key sources in the literature regarding wetland reclamation studies associated with the Florida phosphate industry include Robertson (1983 & 1985), Barkuloo (1980), Clewell (1981), Dames and Moore (1983), Odum et al. (1983) and others.

Most prior studies of mine-site wetland reclamation have focused specifically on soil "mulching" feasibility, comparative planting techniques, plant species growth assessments, wildlife dynamics and other particular facets of a newly emerged field of investigation. This study does not take an in-depth approach to any of these reclamation subjects, but rather seeks to provide a synopsis of the important interrelationships of these factors via ecosystem assessments of three varied reclamation sites. The three reclamation projects selected for inclusion in this study have the common characteristics of being reclaimed for high natural system habitat values, long-term wildlife considerations, diversity enhancement and viable watershed function. Significantly, each project is directly associated with an existing, natural wetland system in order to enhance ecosystem integration and contributions, and each has been designed to function hydrologically on a self-sustaining basis.

Reclamation activities required for the creation of the three wetland systems have been carried out by the International Minerals and Chemical Corporation (a.k.a. IMC Fertilizer, Inc., and IMC) since 1978. The subject sites are "Tiger Bay," near Homeland in Polk County, "East of Peace River," to the southeast of Bartow in Polk County, and "West of K-16," in eastern Hillsborough County near Pinecrest. These sites are located in Figure 1. The biotic and abiotic conditions associated with each reclamation project were investigated in the field by ecologists Kevin Atkins and Phillip Sacco of Henigar & Ray Engineering Associates, Inc. in August 1986, and by Atkins in March and April 1987.

On-site reclamation tasks are complete for the Tiger Bay and West of K-6 projects, while reclamation of the East of Peace River site is on-going. The reclamation plans primarily were generated by IMC's reclamation engineers under the direction of Mr. Robert F. Goodrich, although the author co-authored a restoration guidelines report for West of K-6 (EcoImpact 1979) that was a basis for reclamation of that system. Comprehensive floral and faunal inventories recorded for each site are presented in the Discussion and Conclusions section as Table 1 and Table 2, respectively. Confirmation of the listed flora was provided by David W. Hall, Ph.D., at the University of Florida Herbarium. Animal occurrence data was collected via positive sightings and the opportunistic encountering of species-specific sign (tracks, scats, burrows, etc.).

STUDY SITES

Tiger Bay

The Tiger Bay wetland reclamation site, also known as South Tiger

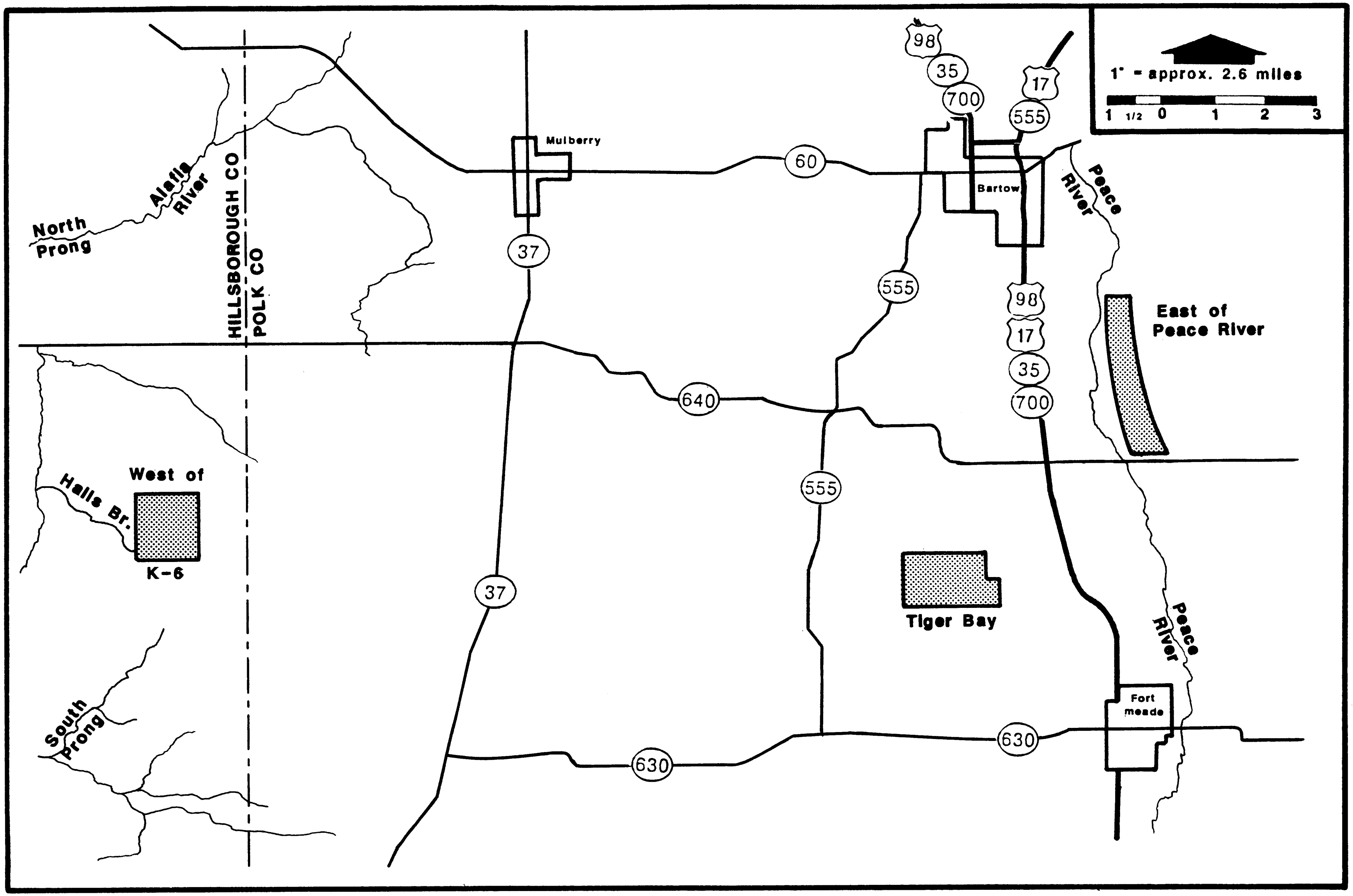

Figure 1. Wetland reclamation study site locations.

Bay, is a large tract located in portions of Sections 17 and 18 in Township 31 South, Range 25 East and Section 13, Township 31 South, Range 24 East. Appropriate mine reclamation activities included capping (overlaying phosphatic waste clays with sand tailings following product benefication), grading, seeding, mulching (a generalized phosphate industry and reclamation term for the spreading of organic soil from a donor wetland onto a reclamation site to inoculate the area with propagules, microbes and a richer soil) and fertilization. These were completed in 1982, but planting continued through 1986. The Tiger Bay study site occupies about 150 ha in a long and narrow, east-west configuration. It is surrounded almost entirely by recently mined land, although some natural vegetation occurs just off-site to the northeast, and it has been designed to be a self-contained drainage area. Outflow is directed eastward toward the Peace River via the historic Campmeeting Ground Branch drainageway and through a downstream control structure.

The wetlands and aquatic systems have peripheral slopes and dikes which contain water on-site. Much of the interior of the site has been designed to flood as wetlands on seasonal and semi-permanent bases, and some areas are permanent, aquatic habitat. Extensive grading of the entire tract was carried out upon the completion of mining in July 1978. The land reclamation was completed in June 1982, and most of the revegetation tasks were completed by 1984.

A botanical survey of Tiger Bay in August 1986 identified 123 plant species. Over 30 of these species were planted by IMC in order to increase the species richness of the plant community, accelerate succession and improve the habitat for wildlife. Planting techniques included hand-planting, direct-seeding and mulching. Seeds of some wetland edge species, such as bulrush (Scirpus validus and S. cyperinus), were dispersed in aquatic areas in an attempt to mimic natural processes (simulated hygrochory), and these efforts have been at least partially successful in establishing such plants.

According to IMC data submitted to the Florida Department of Natural Resources (DNR) in a 1985 post-mining reclamation report, about 60,000 plants have been planted in Tiger Bay, and nearly all were wetland or transitional species. A reclamation variance was sought by IMC and granted from DNR allowing a small, unreclaimed island in the middle of an aquatic system in the southeast quadrant of the parcel. The island has mature laurel oak (Quercus laurifolia) and black cherry (Prunus serotina) trees which will serve as seed sources and wildlife attractors. The sanctuary of the island is known to attract otters (Lutra canadensis) and many species of wading birds.

Cattail (Typha domingensis) is the dominant aquatic edge plant in the system, although many species of more ecologically desirable plants are established throughout and typically are increasing their areal cover. Arrowroot (Thalia geniculata) is thriving, and other common plants include spikerush (Eleocharis cellulosa), fringerush (Fimbristylis autumnalis and F. puberula), soft rush (Juncus effusus), maidencane

(Panicum hemitomon), arrowhead (Sagittaria lancifolia), bulrush, and pickerelweed (Pontederia cordata). Within the topographic depressions and created sloughs of the interior floodplain, plants such as smartweed (Polygonum hydropiperoides and P. punctatum), water-hyssop (Bacopa monnieri), arrowhead, rushes (Juncus dichotomus, J. marginatus & J. megacephalus), micranthemum (Micranthemum glomeratum) and bulrush form potentially high-quality, seasonally wet habitat with long-term significance. Southern naiad (Najas quadalupensis) is the most common submersed species. The surrounding uplands tend to be vegetated by a typical mineland reclamation assemblage of bahia grass (Paspalum notatum), hairy indigo (Indigofera hirsuta), aeschynomene (Aeschynomene americana), and various other ruderal and cultivated herbs and grasses.

The thousands of tree seedlings and saplings planted within the vast Tiger Bay site are most apparent when investigating on foot. The largest of the abundant Cypress trees are approaching 2.5m in height after 3-4 years, while the other tree species are somewhat smaller. Commonly encountered trees include pop ash (Fraxinus caroliniana), loblolly bay (Gordonia lasianthus), dahoon holly (Ilex cassine), sweetgum (Liquidambar styraciflua), southern magnolia (Magnolia cerifera), black gum (Nyssa sylvatica var. biflora), slash pine (Pinus elliottii), laurel oak, and water oak (Quercus nigra).

The wildlife community is relatively well represented for such a recently reclaimed site with a minor natural system connection. The vertebrate inventory includes 58 bird species, 9 mammals, 5 reptiles, 3 amphibians and 4 fishes. Various species of ants appear to be the most common terrestrial invertebrates. A useful aspect of the habitat restoration program on-site is the establishment of several artificial "roost trees" for wading birds, hawks and other, as well as Osprey (Pandion haliaetus) nesting platforms. A pair of young Ospreys was successfully fledged in a nest built in one of the platforms in the spring of 1987, and the adults were able to capture fish from the on-site aquatic systems. Some protruding woody "snags" in open water areas provide perching sites for Anhingas (Anhinga anhinga), Double-crested Cormorants (Phalacrocorax auritus), Kingfishers (Ceryle alcyon), and others.

Investigations on-site led to the discovery that many Red-winged Blackbirds (Agelaius phoeniceus) and some Common Yellowthroats (Geothlypis trichas) were nesting during spring in young trees planted for reclamation. Cypress was preferred in almost all cases, with dahoon holly selected occasionally. Another encouraging avian observation was the sighting of a Red-tailed Hawk (Buteo jamaicensis) capturing an apparent cotton rat on-site. Bobcat (Felis rufus) sign also is fairly common, suggesting that dynamic predator-prey relationships are being established on-site. Small mammal trails are numerous throughout the interior and shallow marsh areas, with attendant sign for otters, marsh rabbits (Sylvilagus palustris), raccoons (Procyon lotor) and rodents. In addition, the substrate in aquatic areas is satisfactory for bluegill (Lepomis macrochirus) and other centrarchid reproduction since active beds have been observed at densities greater than one per square meter in some littoral zone areas. Aquatic invertebrates and fresh-water mussels also were observed, providing further evidence of food chain establishment since reclamation.

East of Peace River

The East of Peace River wetland reclamation parcel, also known generally as the CS-11 Floodplain, lies in portions of Sections 27, 34 and 35, in Township 30 South, Range 25 East. Appropriate reclamation activities (capping, grading, seeding, mulching, planting) have rapidly followed the mining process northward from County Road 640 since 1979, and mining and reclamation of this region will continue through 1988. This active reclamation program is taking place on mined land immediately adjacent on the east to the 25-year floodplain of the Peace River forest. Within an approximately 250 ha watershed study area, the wetlands reviewed for this study occupy about 100 ha. Additional wetland acres are being reclaimed within months of the dragline leaving an area.

The East of Peace River site represents an interesting opportunity for successional vegetation studies over known time intervals. Aquatic, wetland and upland reclamation has been carried out by IMC in annual increments from south to north since 1979. The floral density and diversity clearly increase over time. The botanical survey of this site revealed an assemblage of at least 120 plant species. Of these, about 30 species have been planted in order to increase community structure and diversity and to accelerate natural vegetative succession. Wetland basin mulching (from recently mined, isolated wetlands sources) on this site also has been an effective method for achieving these goals. The early reclamation sites now supply wetland "donor" plants for continuing reclamation.

The variety of woody and herbaceous plants incorporated in the reclamation design should result in a mosaic of canopied and uncanopied systems throughout the site within a couple of decades. At a wetland parcel reclaimed in 1984, a single belt transect (30m x 10m) was established randomly along the shoreline such that the centerline was the landward edge of standing water on August 26, 1986. The intent was to ascertain a representative number of surviving tree transplants within the system's littoral and transition zones.

The results indicated 76 living, young trees of eight different species (cypress, dahoon holly, ash, black gum, red maple, sweetbay, laurel oak and loblolly bay). Assuming that this assessment is representative, a hectare of shoreline (10 m wide) may be expected to contain approximately 2,700 living trees. Even allowing for various mortality factors, the density and diversity of such a reclamation tree planting reasonably can be expected eventually to produce a heterogeneous swamp forest fringe around the open water wetland. The surrounding uplands also have been planted with oaks (*Quercus geminata*, *Q. laurifolia* & *Q. nigra*), slash pine (*Pinus elliottii*), dogwood (*Cornus* florida), persimmon (Diospyros virginiana), southern red cedar (Juniperus silicicola) and sweetgum.

The herbaceous plant assemblage associated with the East of Peace River wetlands, whether planted or naturalized, is vigorous and expanding. In the emergent zone, the Pontederia-Juncus-Sagittaria-Thalia association tends to occupy as much area as the typically common cattail, and many other sedges, rushes and grasses also occur within the wetland fringe. Southern naiad (Najas quadalupensis) is the most common submersed species. Also, the close proximity of the Peace River floodplain forest should be an increasingly important successional asset, as evidenced by seedlings from such trees as sugarberry (Celtis laevigata), elm (Ulmus americana), sparkleberry (Vaccinium arboreum), red maple (Acer rubrum), small viburnum (Viburnum obovatum), and sweetgum encroaching upon the reclamation site from the existing natural system.

The wildlife community clearly is responding to this newly created wetland and upland habitat. Fifty-nine bird species were identified on-site, including many wading and shore birds. Notable among these were the Greater and Lesser Yellowlegs (Tringa melanolenca & T. flavipes), Spotted Sandpiper (Actitis macularia), Red Knot (Calidris Canutus), Black-necked Stilt (Himantopus mexicaus), Wood Stork (Mycteria americana), Glossy Ibis (Plegadis falcinellus), Common Snipe (Gallinago gallinago), and all of the common herons and egrets. Other notable birds seen on-site were the Black-crowned Night Heron (Nycticorax nycticorax), Pied-billed Grebe (Podilymbus podiceps), Least Bittern (Ixobrychus exilis), Great Horned Owl (Bubo virginianus), Bobolink (Dolichonyx oryzivorus) and Wild Turkey (Meleagris gallopavo).

Mammals were noted primarily on the basis of diagnostic sign rather than sightings. The white-tail deer (Odocoileus virginiana), feral hog (Sus scrofa) and river otter are animals that visit the reclamation site from the adjacent floodplain. In addition to ten mammal species, five reptiles and three amphibians were encountered, all of which, except the green anole, are associated with wetlands (see Table 2 in the next section). No formal fish sampling study was conducted, but the mosquito fish (Gambusia affinis), bluegill, golden shiner (Notemigonus chrysoleucas) and largemouth bass (Micropterus salmoides) were observed. Various invertebrates encountered included crayfish, apple snails and several ant species.

Significantly, this reclamation project adjacent to the Peace River floodplain received the 1987 Environmental Achievement Award from the National Wildlife Federation's Corporate Conservation Council.

West of K-6

The West of K-6 reclamation unit of IMC's Kingsford Mine is located in eastern Hillsborough County in Section 11, Township 31 South, Range 22 East. Appropriate reclamation activities began in

April 1980 and were completed in 1984. The study site was mined between August 1979 and March 1980. For the purpose of this survey, the watershed area studied was about 55 ha and the associated aquatic and wetland area was about 14 ha. Other cover types on mined land within the site include reclaimed xeric scrub, planted pines, planted live oak hammock and an orange grove. A key feature of this program is the connection of the reclaimed wetland with the Hall's Branch tributary to the South Prong of the Alafia River. In effect, the West of K-6 wetland now forms the headwaters of the Hall's Branch system.

A West of K-6 plant regime for wetland species that was begun in 1982 was completed in 1984. In 1986, the upland plantings also were completed. Some wetland and upland tree species were direct-seeded at various locations within the tract. In addition to wetlands, the planting scheme included an 8 ha citrus grove (about 1300 trees) on the eastern side of the watershed, planted pines and upland hammock species on the south and sand pine and oak scrub on the northwest.

The connection of this wetland reclamation program with the Hall's Branch tributary system should produce long-term ecosystem restoration benefits. The dominant woody species in the existing, natural floodplain forest are sweetgum, sweetbay, laurel oak, water oak, willow (Salix caroliniana), live oak (quercus virginiana), red maple, elderberry (Sambucus simpsonii), black gum, red bay (Persea borbonia), primrose willow (Ludwigia octovalvis) and shining sumac (Rhus copallina). All of these indigenous plants either have been planted or have become established naturally at the West of K-6 reclamation site. An obvious benefit of the direct, natural system interface is the potential for plant materials exchange, and a few Hall's Branch plants, particularly sweetbay and red maple, are beginning to invade "upstream" into the created wetland system.

Many of the planted wetland species on-site are thriving, including cypress, dahoon holly, sweetgum, magnolia and loblolly bay. A planted and natural wax myrtle fringe in the upper transition zone is providing good cover and food (berries) for wildlife. The planted ground cover and emergent marsh species, such as soft rush, maidencane and cordgrass (Spartina bakeri) also display vigorous growth. The steeply sloped drainage basin on the north, east and west, and the more moderately sloped watershed to the south drain into the centralized wetland and aquatic system, which discharges into the Hall's Branch intermittent watercourse. The connection between the two systems already is relatively natural in appearance and it serves the intended purpose. On the south side of the wetland basin, seepage fingers and drainage sloughs have been designed into the downslope topography. These have been extensively planted with wetland trees and ground cover to control erosion and to ultimately form a wetland forest strand system.

The plant inventory for the site lists 141 species, which is about one-third more than similarly aged reclamation sites in this study because of the diversity afforded by the upland and xeric association plantings incorporated in the West of K-6 program. About 50 plant species have been planted to accelerate the vegetative succession process. The uplands have been planted with a variety of pines and oaks, along with such species as southern magnolia and red cedar. Experimental xeric mulching using sand pine scrub topsoil has been relatively successful. The wetlands and transitional zones are planted with an unusually diverse assemblage of woody species, including blue beech (Carpinus caroliniana), buttonbush (Cephalanthus occidentalis), fringetree (Chionanthus virginica), swamp dogwood (Cornus foemina), loblolly bay, American holly (Ilex opaca), sweetgum, sweetbay, red bay, black gum, swamp chestnut oak (Quercus michauxii), cabbage palm (Sabal palmetto) and cypress.

Several unusual, non-woody plants have become common within the wetland fringe and deserve special mention. Among these are: water-hyssop; yellow-white sedge (Carex albolutescens); water-spider orchid (Habenaria repens); red hibiscus (Hibiscus coccineus); winged lythrum (Lythrum alatum); micranthemum; milkwort (Polygala cruciata); tapering tri-vein fern (Thelypteris dentata); yellow-eyed grasses (Xyrus jupicai & J. platylepis); and sphagnum moss (Sphagnum sp.). The establishment of such desirable wetland ground cover plants is positive from the standpoint of heterogeneity, system function and aesthetics, and the fact that they have been "volunteer" colonizers is especially meaningful.

The wildlife community on the West of K-6 reclamation site is partially representative of some naturally occurring sites in the region. The wildlife inventory for the site includes 41 species of birds, 7 mammals, 6 reptiles, 6 amphibians, 2 fish species and many, unspecified macro-invertebrates. Notably, the wetland restoration site serves as a roost for Black-crowned Night Herons, which apparently spend their days in the cover of the shrubs and willow trees associated with the open water area. The only bird species observed at the West of K-6 site that was not observed at the other study sites was the eastern kingbird (Tyrannus tyrannus). Except for the armadillo (Dasypus novemcinctus), eastern cottontail (Sylvilagus floridanus) and cotton rat (Sigmodon hispidus), all of the mammals were identified by species-specific sign. Wetland-related reptiles included the Florida cottonmouth (Agkistrodon piscivorus conanti) and soft-shell turtle (Trionyx ferox). The amphibians included 6 frog species. The comprehensive faunal inventory is presented in Table 2 in the following section.

DISCUSSION AND CONCLUSIONS

The three sites reported in this study have been selected from many potential candidates because each has wetland reclamation attributes that represent rapid development toward viable natural system function and ecosystem integration. Among these characteristics are: watershed-based design; contiguity with existing, natural wetlands tied to the regional drainage network; topographic diversity; extensive and diverse native vegetation plantings; specific considerations regarding wildlife habitat and attractors; unmanipulated hydroperiod and stage level conditions; and maximized interspersion of aquatic areas, wetlands and uplands. Although the basic procedures for reclamation planning are dictated by the DNR through statutory authority under Sections 16C-16 and 16C-17 of the Florida Administrative Code, and by the Florida Reclamation Advisory Committee, there can be enough flexibility of approach to provide for techniques that can enhance projects with respect to ecological and watershed considerations.

The three IMC projects reviewed for this paper represent an advancement over most prior wetland reclamation on phosphate-mined lands in terms of the commitment to establishing short-term ecological complexity and re-establishing watershed function. The relative percent of created wetlands, the abundance and diversity of planted native vegetation and the enhancement of wildlife habitat values all exceeded any mandated reclamation criteria. Although the three study areas have been evolving only for six months to six years, this paper documents a combined floral inventory (both planted and successional) of 212 plant species (Table 1). During the limited fall 1986-spring 1987 faunal surveys (opportunistic identification only), 76 bird species, 10 mammals, 10 reptiles, 6 amphibians and 5 fishes were recorded between the three sites (Table 2)

The design of these large-scale reclamation projects involved a systems-oriented, watershed-based approach that went beyond achieving mere plant cover and program release by the DNR. Reclamation concepts must become more ecologically holistic in order to keep pace with the increasing demands by the public and the various regulatory agencies for greater species richness, systems interspersion and natural system values. Reclaimed wetlands must be self-maintaining and ultimately provide the natural system functions of water storage, water purification, groundwater recharge, flood attenuation, fish and wildlife habitat, and recreation, among others. A habitat-oriented approach should be incorporated, including optimized littoral zones, irregular shorelines and plant association edges, topographic diversity, variable water levels, a functional proportion of open water, emergent marsh and forested wetland (as applicable) and increased incorporation of wildlife attractors and corridors. Each of the three study sites had evidence of the effectiveness of attracting wildlife and creating features (natural and man-made) that support the life requirements of a diversity of classes and species of animals.

Finally, the economic and energy investments in large-scale, watershed-based, wetland reclamation projects must be protected over the long term by regulations in order to allow for the evolution of truly functional natural systems with wide-ranging importance in the regional ecosystem. Contiguity of reclaimed wetlands with jurisdictional, natural system wetlands can help to ensure the maximum level protection accorded by local, state and federal law.

Table 1. Recently reclaimed mineland plant species inventory.

Scientific Binomial	Common Name	Tiger Bay	E.of P.R.	W.of K-6
Acalypha gracilens	Three-seeded Mercury			
Acer rubrum (P)	Red Maple	X	X	X
Aeschynomene americana	Shy-leaves	X	X	X
Agalinis fasciculata	Cluster-leaf Gerardia	X		
Alternanthera philoxeroides	Alligator Weed	X		
Alysicarpus vaginalis	Alyce-clover		X	
Amaranthus australis	Giant Amaranth			X
Ambrosia aratemisiifolia	Common Ragweed	X	X	X
Ammannia coccinea	Purple Ammannia		X	
Ampelopsis arborea	Pepper Vine		X	
Andropogon virginicus	Broomsedge	X	X	X
Asclepias incarnata	Swamp Milkweed			X
Asimina reticulata (P)	Paw-Paw			X
Azolla caroliniana	Mosquito Fern		X	X
Baccharis halimifolia	Saltbush	X	X	X
Bacopa monnieri	Water-hyssop	X	X	X
Betula nigra (P)	River Birch			
Bidens bipinnata	Spanish Needles			
Bidens alba	Spanish Needles	X	X	
Brachiaria mutica	Paragrass		X	
Buchnera americana	Blue Heart	X		X
Bulbostylis barbata	Water-grass	X	X	X
Campsis radicans	Trumpet Vine		X	
Carex albolutescens	Yellow-white Sedge	X		X
Carex lupulina	Carex Sedge		X	
Carpinus caroliniana (P)	Blue Beech			X
Carya aquatica (P)	Water Hickory	X		X
Carya glabra	Pignut Hickory			X
Carya tomentosa (P)	Mockernut Hickory	X		X
Cassia chamaecrista	Partridge Pea	X	X	X
Cassia nictitans	Sensitive Plant		X	
Cassia obtusifolia	Sicklepid	X	X	
Celtis laevigata (P)	Sugarberry	X	X	
Centella asiatica	Coinwort			X
Cephalanthus occidentalis (P)	Buttonbush	X	X	X
Chamaesyce hyssopifolia	Hyssop Spurge	X	X	X
Chenopodium ambrosioides	Mexican Tea			X
Chionanthus virginica (P)	Fringetree			X
Chloris petraea	Fingergrass	X		
Cinnamomum camphora	Camphor Tree			
Clematis virginiana	Virgin's Bower			
Conyza canadensis	Dwarf Horseweed	X	X	X
Cornus florida (P)	Flowering Dogwood	X	X	X
Cornus foemina (P)	Swamp Dogwood		X	X
Crotalaria lanceolata	Lanceleaf Crotalaria		X	

(P) = Planted

Table 1. Recently reclaimed mineland plant species inventory (cont'd).

Scientific Binomial	Common Name	Tiger Bay	E.of P.R.	W.of K-6
Crotalaria pallida	Smooth Crotalaria		X	X
Crotalaria rotundifolia	Rabbit-bells	X	X	X
Crotalaria spectabilis	Rattle-box	X		X
Cynodon dactylon	Bermuda Grass	X	X	X
Cyperus compressus	Annual Sedge	X	X	X
Cyperus distinctus	Sedge		X	
Cyperus globulosus	Globe Sedge	X		
Cyperus iria	Rice Flat Sedge	X		X
Cyperus odoratus	Flat Sedge		X	X
Cyperus polystachyos	Texas Sedge	X	X	X
Cyperus retrorsus	Cylindric Sedge	X	X	X
Cyperus surinamensis	Surinam Sedge	X	X	X
Cyperus virens	Sedge		X	
Dactyloctenium aegyptium	Crowfoot Grass		X	
Dalea carnea	Three-corner Prairie-Clover		X	
Desmodium incanum	Creeping Beggarweed			X
Desmodium lineatum	Beggarweed	X	X	X
Desmodium paniculatum	Lance-leaved Beggarweed			X
Desmodium tortuosum	Annual Beggarweed		X	X
Desmodium triflorum	Sagotia Beggarweed		X	
Digitaria bicornis	Tropical Crabgrass	X	X	X
Digitaria ciliaris	Southern Crabgrass	X	X	
Diodia virginiana	Buttonweed	X		X
Diospyros virginiana	Persimmon	X	X	
Echinochloa colonum	Jungle Rice	X		
Echinochloa crusgalli (P)	Barnyard Grass	X		
Echinochloa walteri	Coast Cockspur	X	X	X
Eclipta prostrata	Eclipta	X	X	X
Egeria densa	Brazilian Elodea		X	
Eichhornia crassipes	Water Hyacinth	X		
Eleocharis cellulosa	Spikerush	X		
Eleusine indica	Goose Grass	X		X
Equisetum hyemale	Scouring Rush	X		
Eragrostis pilosa	Indian Love Grass	X		
Erechtites hieracifolia	Fireweed	X		X
Eupatorium capillifolium	Dogfennel	X	X	X
Eupatorium serotinum	Late Boneset	X	X	X
Euthamia minor	Flat-topped Goldenrod	X	X	X
Fimbristylis autumnalis	Slender Fringerush	X		
Fimbristylis dichotoma	Forked Fringerush		X	
Fimbristylis puberula	Fringerush	X		
Fraxinus caroliniana (P)	Pop Ash	X	X	X
Garberia fruticosa (P)	Garberia			X

(P) = Planted

Table 1. Recently reclaimed mineland plant species inventory (cont'd).

Scientific Binomial	Common Name	Tiger Bay	E.of P.R.	W.of K-6
Gnaphalium obtusifolium	Rabbits's Tobacco			X
Gordonia lasianthus (P)	Loblolly Bay	X	X	X
Habenaria repens	Water-Spiker Orchid			X
Heterotheca subaxillaris	Camphor Weed	X	X	X
Hibiscus coccineus	Red Hibiscus	X	X	X
Hydrilla verticillata	Hydrilla	X	X	
Hydrocotyle umbellata	Marsh Pennywort	X	X	X
Hypericum mutilum	Dwarf St. John's-Wort			X
Hyptis mutabilis	Bitter Mint	X		
Indigofera hirsuta	Hairy Indigo	X	X	X
Ilex cassine (P)	Dahoon Holly	X	X	X
Ilex glabra (P)	Gallberry	X		
Ilex opaca (P)	American Holly	X		X
Imperata Cylindrica	Satin-tail Grass	X		X
Ipomoea indica	Morning Glory	X		X
Juncus dichotomus	Two-parted Rush	X	X	X
Juncus effusus (P)	Soft Rush	X	X	X
Juncus marginatus	Shore Rush	X		
Juncus megacephalus	Large-headed Rush	X		
Juncus scirpoides	Rush	X		
Juniperus silicicola (P)	Red Cedar		X	X
Lantana camara	Lantana	X	X	
Leersia hexandra	Cutgrass		X	
Lemna obscura	Duckweed	X		X
Licania michauxii (P)	Gopher Apple			X
Limnobium spongia	Frog's Bit			
Lindernia grandiflora	Round-leaved False Pimpernel	X		
Liquidambar styraciflua (P)	Sweetgum	X	X	X
Ludwigia octovalvis	Long-fruited Primrose Willow	X	X	X
Ludwigia peruviana	Primrose-willow			X
Ludwigia repens	Red Ludwigia	X		X
Ludwigia suffruticosa	Headed Seed Box			X
Lythrum alatum	Winged Lythrum	X	X	X
Macroptilium lathyroides	Phasey Bean	X	X	X
Magnolia grandiflora (P)	Southern Magnolia	X		X
Magnolia virginiana (P)	Sweetbay	X	X	X
Micranthemum glomeratum	Micranthemum	X		X
Mikania scandens	Climbing Hempvine	X	X	X
Momordica charantia	Wild Balsam-apple		X	
Myrica cerifera (P)	Wax Myrtle	X	X	X
Najas guadalupensis	Southern Naiad	X		X
Nyssa sylvatica var. *biflora* (P)	Black Rum or Tupelo	X	X	X

(P) = Planted

Table 1. Recently reclaimed mineland plant species inventory (cont'd).

Scientific Binomial	Common Name	Tiger Bay	E.of P.R.	W.of K-6
Oxalis stricta	Yellow Wood Sorrel			X
Panicum dichotomiflorum var. *bartowense*	Hairy Fall Panicum	X	X	
Panicum hemitomon (P)	Maidencane	X	X	X
Panicum maximum	Guinea Grass	X	X	X
Panicum repens	Torpedo Grass		X	
Panicum virgatum (P)	Switch Grass		X	X
Parthenocissus quinquefolia	Virginia Creeper			
Paspalum notatum	Bahia Grass	X	X	X
Paspalum paspalodes	Knot Grass		X	
Paspalum urvillei	Vassey Grass	X	X	
Passiflora incarnata	Passion Flower Vine	X		
Peltandra virginica (P)	Arrow Arum		X	
Persea borbonia (P)	Red Bay		X	X
Persea humilis (P)	Silk or Scrub Bay			X
Phyla nodiflora	Matcheads	X	X	X
Phytolacca americana	Pokeweed	X		X
Pinus clausa (P)	Sand Pine			X
Pinus elliottii var. *densa* (P)	Southern Slash Pine			X
Pinus elliottii var. *elliottii* (P)	Northern Slash Pine	X	X	X
Pinus palustris (P)	Longleaf Pine			X
Pinus taeda (P)	Loblolly Pine			X
Pluchea odorata	Fleabane	X	X	X
Polygata cruciata	Milkwort			X
Polygonum hydropiperoides (P)	Mild Water-pepper	X	X	X
Polygonum punctatum (P)	Dotted Smartweed	X	X	X
Polypremum procumbens	Polypremum	X	X	X
Pontederia cordata (P)	Pickerelweed	X	X	X
Prunus serotina (P)	Black Cherry	X		X
Pterocaulon virgatum	Blackroot			X
Quercus chapmanii (P)	Chapman's Oak			X
Quercus geminata (P)	Scrub Live Oak		X	X
Quercus incana (P)	Bluejack Oak			X
Quercus inopina (P)	Scrub Oak			X
Quercus laevis (P)	Turkey Oak			X
Quercus laurifolia (P)	Laurel Oak	X	X	X
Quercus michauxii (P)	Basket or Swamp Chestnut Oak	X		X
Quercus nigra (P)	Water Oak	X	X	X
Quercus virginiana (P)	Live Oak		X	X
Rhododendron viscosum (P)	Wild Honeysuckle			X
Rhus copallina	Shining Sumac		X	

(P) = Planted

Table 1. Recently reclaimed mineland plant species inventory (cont'd).

Scientific Binomial	Common Name	Tiger Bay	E.of P.R.	W.of K-6
Rhynchelytrum roseum	Natal Grass	X	X	
Rhynchosia minima	Small Rhynchosia			X
Richardia scabra	Florida Purslane			X
Ricinus communis	Castorbean		X	
Rubus cuneifolius	Elderberry	X		
Sabal palmetto (P)	Cabbage Palm	X	X	X
Sacciolepis indica	India Cupscale Grass			X
Sacciolepis striata	American Cupscale Grass		X	
Sagittaria lancifolia (P)	Arrowhead or Wapato	X	X	X
Salix caroliniana	Carolina Willow	X	X	X
Salvinia rotundifolia	Water Spangles		X	
Sambucus simpsonii	Elderberry	X		X
Schinus terebinthifolius	Brazilian Pepper	X		
Scirpus cyperinus (P)	Wool-grass Bulrush	X	X	X
Scirpus validus	Soft-stem Bulrush	X	X	
Scoparia dulcis	Goat Weed	X	X	X
Serenoa repens (P)	Saw Palmetto			X
Sesbania macrocarpa	Hemp Sesbania		X	X
Sesbania vesicaria	Bagpod	X	X	
Setaria geniculata	Knotroot Foxtail	X	X	X
Sida acuta	Southern Sida			
Sida rhombifolia	Indian Hemp	X		X
Smilax bona-nox	Greenbriar		X	
Solidago chapmanii	Chapman's Goldenrod		X	X
Solidago gigantea	Giant Goldenrod		X	X
Spartina bakeri (P)	Baker's Cordgrass		X	X
Sphagnum sp.	Sphagnum Moss			X
Spirodela punctata	Duckweed		X	X
Sporobolus indicus	Smut Grass	X	X	X
Taxodium distichum (P)	Bald Cypress	X	X	X
Thalia geniculata (P)	Arrowroot or Fire Flag		X	
Thelypteris dentata	Tapering Tri-Vein Fern	X		X
Typha domingensis	Southern Cattail	X	X	
Ulmus americana (P & un-P)	Elm	X	X	
Urena lobata	Caesar's Weed	X	X	X
Vaccinium arboreum	Sparkleberry	X	X	X
Viburnum obovatum	Small Viburnum		X	
Wolffiella floridana	Bog-mat			X
Woodwardia virginica	Chain Fern		X	
Xyris jupicai	Common Yellow-eyed Grass			X
Xyris platylepis	Broad-scale Yellow-eyed Grass		X	
Yucca Filamentosa	Beargrass			X

(P) = Planted

Table 2. Vertebrate animal species inventory on recently reclaimed mineland.

Birds

Scientific Binomial	Common Name	Tiger Bay	E.of P.R.	W.of K-6
Actitis macularia	Spotted Sandpiper		X	
Agelaius phoeniceus	Red-winged Blackbird	X	X	X
Aix sponsa	Wood Duck		X	
Anas americana	American Wigeon	X	X	
Anas crecca	Green-winged Teal	X		
Anas discors	Blue-winged Teal	X		
Anas fulvigula	Florida Duck	X	X	X
Anhinga anhinga	Anhinga	X	X	X
Ardea herodias	Great Blue Heron	X	X	X
Aythya collaris	Ring-necked Duck	X		
Bubo virginianus	Great Horned Owl		X	
Bubulcus ibis	Cattle Egret	X	X	X
Buteo jamaicensis	Red-tailed Hawk	X	X	X
Buteo lineatus	Red-shouldered Hawk		X	
Butorides striatus	Green-backed Heron	X	X	X
Calidris canutus	Red Knot		X	
Calidris pusilla	Semipalmated Sandpiper	X		
Caprimulgus carotinensis	Chuck-will's Widow		X	
Cardinalis cardinalis	Northern Cardinal	X	X	X
Casmerodius albus	Great Egret	X	X	X
Cathartes aura	Black Vulture	X	X	X
Ceryle alcyon	Belted Kingfisher	X		
Charadrius vociferus	Killdeer	X	X	X
Chordeiles minor	Common Nighthawk		X	
Circus cyaneus	Marsh Harrier	X	X	X
Colinus virginianus	Bobwhite	X	X	X
Columbina passerina	Ground Dove	X	X	X
Coragyps atratus	Black Vulture		X	X
Corvus brachyrhynchos	Common Crow	X	X	X
Dendrocopus pubescens	Downy Woodpecker			X
Dendroica coronata	Yellow-rumped Warbler	X		X
Dendroica palmarum	Palm Warbler	X		X
Dendroica pinus	Pine Warbler	X		
Dolichonyx oryzivorus	Bobolink		X	
Egretta caerulea	Little Blue Heron	X	X	X
Egretta thula	Snowy Egret	X	X	X
Egretta tricolor	Louisiana Heron	X	X	X
Eudocimus albus	White Ibis	X	X	
Falco sparverius	Kestrel			X
Fulica americana	American Coot	X	X	
Gallinago gallinago	Common Snipe	X	X	X
Gallinula chloropus	Common Moorhen	X		
Geothlypis trichas	Common Yellowthroat	X	X	X
Haliaeetus leucocephalus	Bald Eagle	X	X	

Table 2. Vertebrate animal species inventory on recently reclaimed mineland (cont'd).

Birds (cont'd)

Scientific Binomial	Common Name	Tiger Bay	E.of P.R.	W.of K-6
Himantopus mexicanus	Black-necked Stilt	X	X	
Ixobrychus exilis	Least Bittern	X	X	
Lanius ludovicianus	Loggerhead Shrike	X	X	X
Larus atricilla	Laughing Gull	X		X
Larus delawarensis	Ring-billed Gull	X	X	X
Limnodromus scolopaceus	Long-billed Dowitcher	X		
Megaceryle alcyon	Belted Kingfisher		X	
Meleagris gallapavo	Wild Turkey		X	
Mimus polyglottos	Mockingbird	X	X	X
Mycteria americana	Wood Stork	X	X	
Myiarchus crinitus	Great-crested Flycatcher		X	
Nycticorax nycticorax	Black-crowned Night Heron		X	X
Oxyura jamaicensis	Ruddy Duck	X		
Pandion haliaetus	Osprey	X	X	
Passerculus sandwichensis	Savannah Sparrow	X	X	
Pelecanus erythrorhynchos	White Pelican	X		
Phalacrocorax auritus	Double-crested Cormorant	X	X	X
Pipilo erythrophthalmus	Rufous-sided Towhee		X	X
Plegadis falcinellus	Glossy Ibis	X	X	
Podilymbus podiceps	Pied-billed Grebe	X	X	X
Quiscalus major	Boat-tailed Grackle	X	X	X
Quiscalus quiscula	Common Grackle	X	X	X
Sterna caspia	Caspian Tern	X		X
Sterna hirundo	Common Tern	X	X	X
Strix varia	Barred Owl	X	X	
Sturnella magna	Eastern Meadowlark	X	X	X
Tachycineta bicolor	Tree Swallow	X		
Thryothorus ludovicianus	Carolina Wren		X	
Tringa flavipes	Lesser Yellowlegs	X	X	
Tringa melanoleuca	Greater Yellowlegs	X	X	
Tyrannus tyrannus	Eastern Kingbird			X
Zenaida macroura	Mourning Dove	X	X	

Mammals

Scientific Binomial	Common Name	Tiger Bay	E.of P.R.	W.of K-6
Dasypus novemcinctus	Nine-banded Armadillo	X	X	X
Didelphis virginiana	Opossum	X	X	X
Felis rufus floridanus	Bobcat	X	X	X

Table 2. Vertebrate animal species inventory on recently reclaimed mineland (cont'd).

Mammals (cont'd)

Scientific Binomial	Common Name	Tiger Bay	E.of P.R.	W.of K-6
Lutra canadensis	River Otter	X	X	
Odocoileus virginiana	White-tail Deer		X	
Procyon lotor	Racoon	X	X	X
Sigmodon hispidus	Cotton Rat	X	X	X
Sus scrofa x *S. vittatus*	Feral Hog	X	X	
Sylvilagus floridanus	Eastern Cottontail	X	X	X
Sylvilagus palustris	Marsh Rabbit	X	X	X

Reptiles

Scientific Binomial	Common Name	Tiger Bay	E.of P.R.	W.of K-6
Agkistrodon piscivorus conanti	Florida Cottonmouth			X
Alligator mississippiensis	American Alligator	X	X	
Anolis carolinensis	Green Anole		X	X
Chrysemys f. floridana	Florida Cooter	X	X	
Cnemidophorus s. sexlineatus	Racerunner			X
Coluber constrictor priapus	Southern Black Racer	X		
Nerodia cyclopion floridana	Florida Green Water Snake	X	X	X
Nerodia taxispilota	Brown Water Snake			
Trionyx ferox	Florida Softshell	X	X	X

Amphibians

Scientific Binomial	Common Name	Tiger Bay	E.of P.R.	W.of K-6
Acris gryllus dorsalis	Southern Cricket Frog	X	X	X
Gastrophryne carolinensis	Eastern Narrow-mouthed Frog			X
Hyla cinerea	Green Treefrog			X
Rana catesbeiana	Bull Frog			X
Rana grylio	Pig Frog	X	X	X
Rana sphenocephala	Southern Leopard Frog	X	X	X

Table 2. Vertebrate animal species inventory on recently reclaimed mineland (cont'd).

Fishes

Scientific Binomial	Common Name	Tiger Bay	E.of P.R.	W.of K-6
Gambusia affinis	Mosquito Fish	X	X	X
Lepisosteus platyrhynchos	Florida Gar	X		
Lepomis macrochirus	Blue Gill	X	X	X
Micropterus salmoides	Large mouth Bass	X	X	X
Notemigonus crysoleucas	Golden Shiner		X	

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MITIGATION MANAGEMENT OF AN IMPOUNDED BRACKISH WATER MARSH

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ABSTRACT

Sykes Creek marsh on Merritt Island, Brevard County, Florida, was chosen by the U.S. Fish and Wildlife Service and the U.S. Army Corps of Engineers as a site for mitigation of habitat losses occurring during a Federal deepening project in Canaveral Harbor. The 2,000 acre marsh had been isolated from the adjoining Banana River lagoon by construction of encircling levees in the mid-1950s, and has since been managed just for control of mosquitoes. As a result, the marsh has lost its historic functions as a spawning and nursery ground for marine invertebrates and fishes, and as a source of detritus that contributed to productivity in the adjacent estuarine ecosystem. The mitigation plan involves reconnecting the marsh and estuary by installing numerous gated culverts through the levees. A Management plan has been developed to allow exchange of water and fishes without compromising the ability to control mosquitoes. Another management objective is to enhance the area for wood stork feeding to mitigate for the loss of such habitat during harbor deepening. Preliminary results from preconstruction monitoring studies are discussed in terms of their future use to fine-tune the water management plan.

INTRODUCTION

As part of a U.S. Army Corps of Engineers' study of navigation improvement at Port Canaveral, Brevard County, Florida, the authors analyzed the unavoidable losses of significant fish and wildlife resources from construction of the project. These included the loss of a 6.1 ha man-made pond which is currently used as a feeding habitat by wood storks, the loss of 2.4 ha of intertidal mangroves, 3.6 ha of sheltered sandy shoreline used by shorebirds for resting and feeding, 4.9 ha of live oyster bar and sandy shoal, and 19 ha of productive shallow-water habitat. Initially, the authors searched for potential in-kind mitigation measures to offset these losses. Most of these turned out to be prohibitively expensive, or were unacceptable to the local sponsor because they involved Port lands that had been set aside for future expansion. Attention was ultimately focused on two impounded brackish-water marshes associated with Sykes Creek on Merritt Island, 5 km west of the Port (Figure 1).

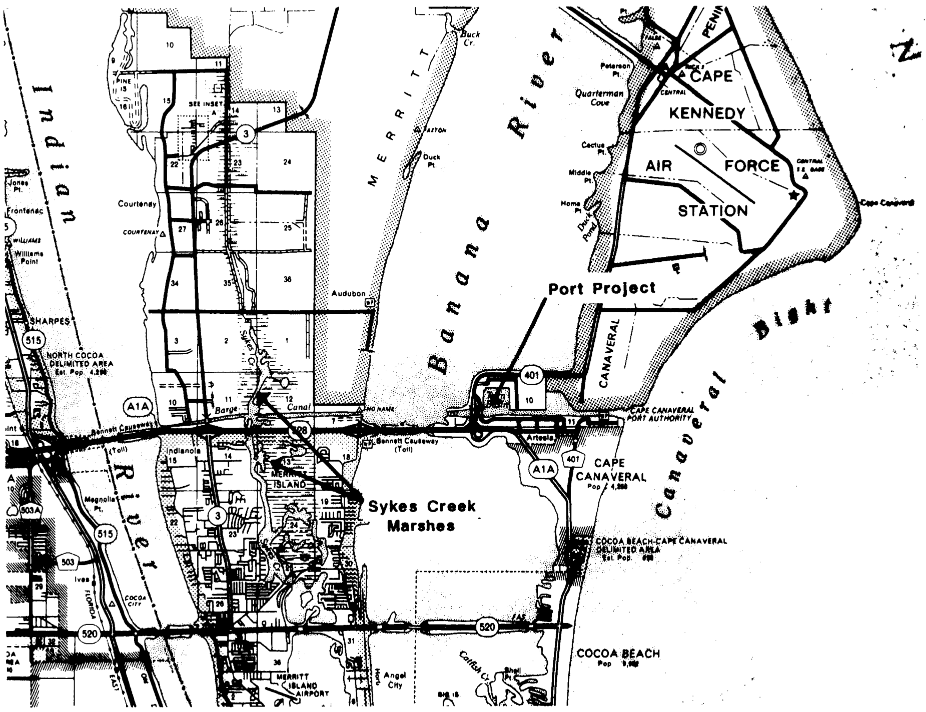

Figure 1. Site location.

STUDY SITE

Sykes Creek is a minor estuary, located between the Banana and Indian Rivers. The major extent of its marshes, about 600 ha, south of State Road 528, lies within a mosquito impoundment, east of the creek channel. North of this road, another 300 ha, including the natural channel and marshes, also have been impounded. Headwater runoff now is routed around the northern impoundment, and is discharged into the barge canal which runs from the Port to the Indian River. These marshes were leveed-off from the creek and adjacent estuaries in the mid-1950s, and have been managed solely for mosquito control purposes ever since. A plan was developed to reconnect the marshes to the surrounding estuary by installing a number of gated culverts at strategic locations in the levees to facilitate the movement of water and biota into and out of the impoundments under the influence of wind-driven tides and seasonal differences in water levels (Figure 2).

There were several objectives for the mitigation measures described above. One was to enhance feeding opportunities for wood storks, a Federally listed endangered species which would be affected by the Port project. A second objective was to increase access to these wetlands for estuarine fishes such as mullet, tarpon, snook, or redfish. Finally, reconnecting the tidal creek to its wetlands would restore the export of nutritive materials to the estuary. While the last two benefits can be claimed purely as a result of constructing a tidal connection through the levees, the benefit to wood storks is

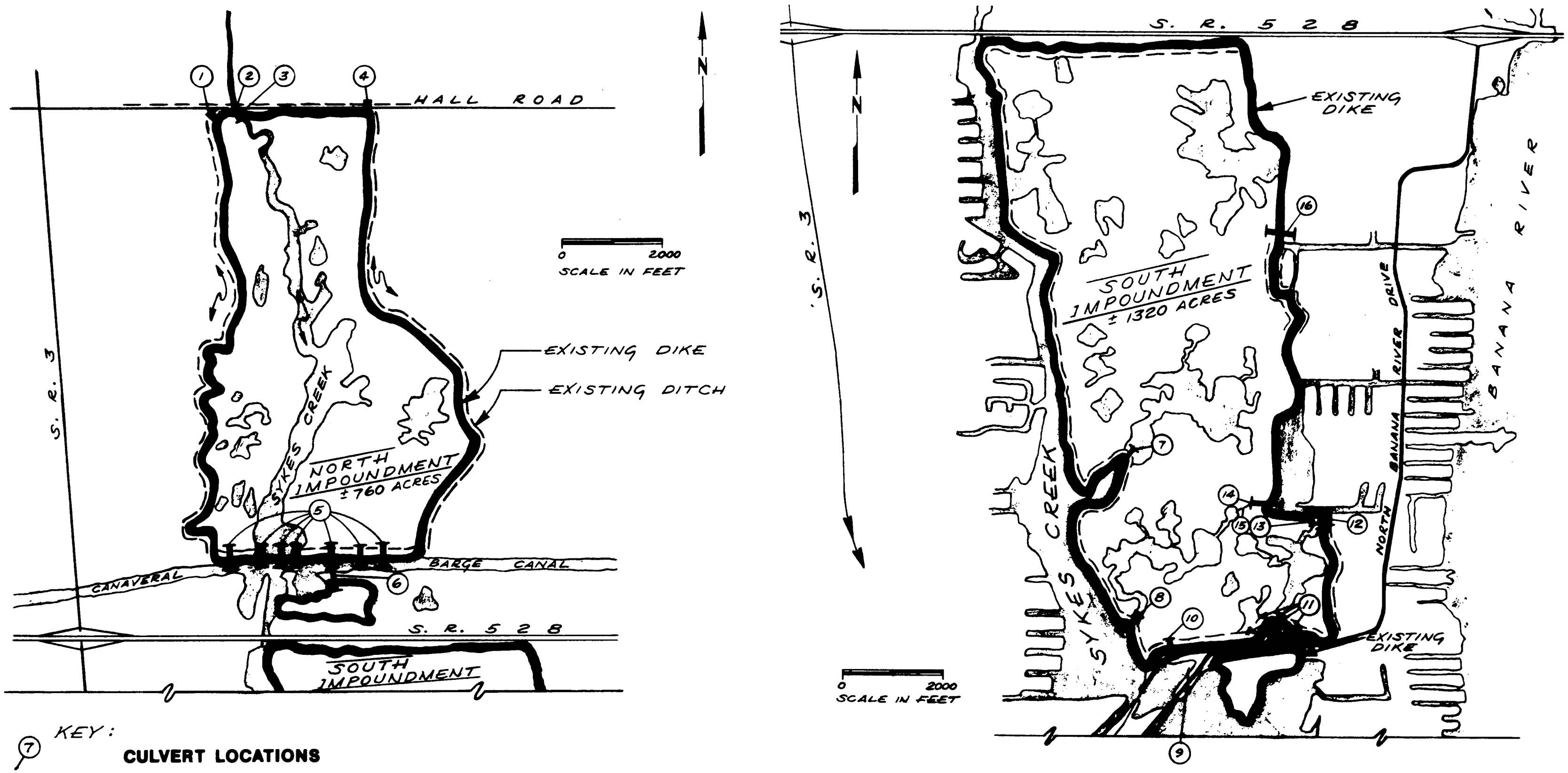

Figure 2. Sykes Creek Impoundments

Figure 2. Sykes Creek impoundments.

dependent upon active management of water levels in the impoundments. The usefulness of these marshes to wood storks would increase if there either were more forage organisms available to them, or if food organisms were available at more critical stages during the life cycle, such as when storks are rearing young.

Since the site is currently managed for mosquito control, management for wood storks had to accommodate the flooding of these areas to prevent oviposition during the mosquito breeding season (which is approximately June through September). The management program currently used for mosquito control impoundments has two phases. Culverts are opened to the estuary during the fall, winter, and spring months, allowing water exchange and migration of animals. During the second phase, all structures are closed and water is pumped in, covering the entire marsh surface during the summer. This process has been given the name Rotational Impoundment Management (RIM).

The management schemes which we will be testing at Sykes Creek will include the RIM (two phase) plan in the north impoundment, but a modification of this will be tested in the south impoundment (Figure 3). At this latitude, wood storks rear young between February and June, with the greatest numbers produced in March, April and May. In contrast, storks currently use these impoundments predominantly in the falls. Therefore, the authors propose to reflood the south impoundment in late winter to promote growth and reproduction of forage fishes. Water would be allowed to gradually decline, through seepage and evaporation to concentrate forage and make it available to the wood storks in the spring. While the amounts of forage available in an impoundment may not be sufficient to support an entire rookery, it is possible that it would supply critical food in years when the major source of food, the St. Johns River marshes, is less than optimal.

During summer, high temperatures and low dissolved oxygen levels are likely to stress estuarine fishes trapped within the impoundments. Thus, another action proposed is a brief (two week) opening in mid-summer to allow escape of fishes from the impoundment to Sykes Creek. The larger water pumps being installed as part of the project will allow rapid replacement of water losses, and so avoid buildup of mosquito populations. Overall, this management plan should provide a reasonable compromise between free and open exchange with the estuary, enhancement of stork feeding opportunities, and control of mosquitoes.

It was obvious that a method was required to evaluate the success of these proposed experimental management proposals. For this reason, the Corps and the Fish and Wildlife Service funded a three-year monitoring study of both impoundments. This first year assesses baseline conditions, prior to any modifications. Next, the standard rotational plan in the north impoundment will be tested along with the modified rotational plan in the south impoundment. In the third year the management plan will be fine tuned. The following are some initial results of this monitoring program.

MATERIALS AND METHODS

There are two general approaches to measuring the success or benefits derived from management measures. The first is through modeling, in which one monitors environmental variables which predict or indicate suitable conditions. The second method of measuring success is direct observation. In this approach, one actually observes or counts the "target organisms" as they use the habitat. Neither method is perfect. The modeling approach assumes that one has taken into account all potential limiting factors, while the other assumes that the period of observation is normal and that the method of observation is accurate. For these reasons, a combination of the two approaches was used. Measurements of salinity, water depth, and substrate elevation were used as indicators of habitat suitability and accessibility for fishes. Feeding conditions suitable for wood storks are estimated from water depths and the concentration of fishes over 25 mm in length. In addition, counts of feeding wood storks along transects between sampling stations are conducted, and fishes collected at each station are sorted by species. They then are further classified as being typical marsh residents (e.g., killifishes), or transients spending part of their life cycle outside the marsh (Gilmore, Cooke, & Donohoe 1982).

Other information is being acquired for habitat evaluation. This includes continuous recording of water levels in both impoundments and in Sykes Creek. Water level records are compared to the known ground elevations within the marsh to calculate water depths, stage duration, and proportion of any type of habitat or vegetation category flooded, at all times of the year.

Preliminary inspection indicated there are five general types of habitat. These include permanent water bodies which historically were linked or connected with the estuary by a channel. Other permanent water bodies were connected through an expanse of wetlands, and thus presumably were less accessible to estuarine organisms. A third category is seasonally flooded wetlands connected across the marsh surface. These are even less accessible, and would require recolonization annually. Two types of vegetated marsh are being sampled; the highest elevation marsh which is characterized by saltgrass and clumped cordgrass, and a longer hydroperiod marsh vegetated by needlerush and leather fern. Each of these five habitat types was sampled, induplicate, within both of the impoundments. In addition, there is one quantitative station in the estuary from which qualitative collections searching for transient fishes are made.

RESULTS

Figure 4 shows data from the initial monthly collections. Note that the sharp decline in number of storks appears correlated with a decrease in concentration of forage-sized fishes.

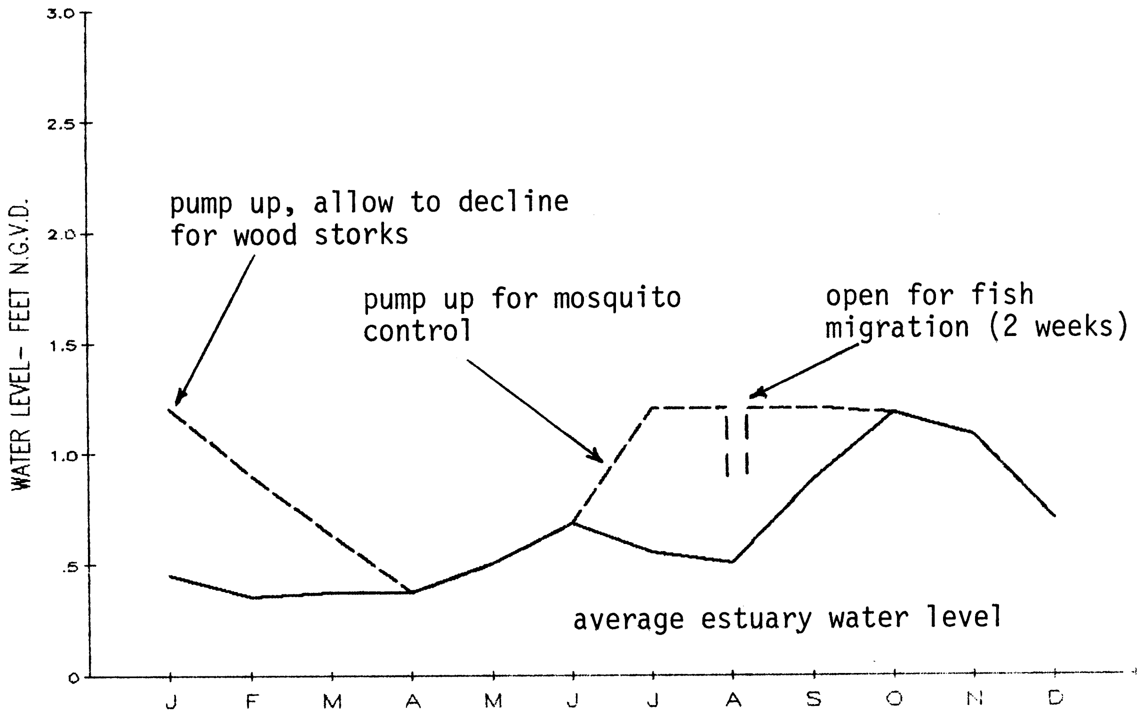

Figure 3. Management proposal for the Sykes Creek impoundments.

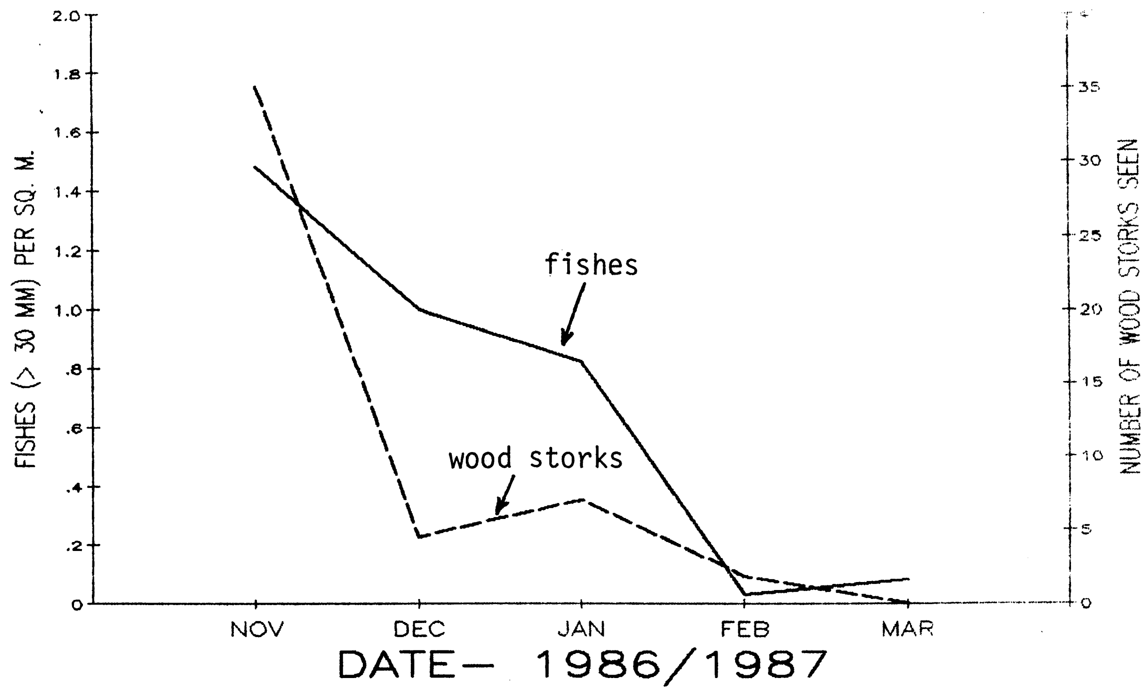

Figure 4. Abundance of forage fishes and wood storks - Sykes Creek impoundments.

Transient fishes (striped mullet and silversides) made up less than 5 percent of the total number of specimens collected (Figure 5). Essentially, the fish community consisted of an assemblage of killifishes. Although the south impoundment yielded greater numbers, the species distribution was the same as in the north. Water salinities of the two impoundments were similar, but both were less saline than Sykes Creek (Figure 6). Salinity is a useful indicator of water exchange. Also, more connections are likely to create a gradient across the marshes, and thus add to habitat diversity.

To this stage of the data collection, the two types of ponds are yielding similar results, as are the two types of emergent marsh. Data have been combined, allowing examination of just three categories: high marsh, seasonal ponds, and permanent ponds. Figure 7 shows the water surface elevations (averaged for the two impoundments) compared to characteristic substrate elevations for the three habitats. Wind tides and rainfall are responsible for the major short term fluctuations both in the creek and the impoundments. The seasonal change in sea level, a range of about 30 cm, results in a longer period fluctuation.

Fish populations appeared responsive to change in water stages. In the fall of the year (Figure 8), the greatest concentration of fish was found in the high marsh; the seasonal ponds had somewhat lower fish densities and the deeper bodies had the lowest. This was in part due to the fact that sampling was initiated late enough in the year that water levels were declining, concentrating fishes in depressions within the marsh. The subsequent samples showed rapid depletion of marsh populations, and a slight increase in fish densities in the ponds, probably individuals which escaped from the marsh. This pattern held throughout the spring. As the higher elevation wetlands dried up, the fishes were easily accessible to shore and wading birds. Later, concentrations in the ponds dropped again. This can be attributed to two factors: heavy predation by least terns and black skimmers, and minor, recurring fish kills. These impoundments have been isolated from free connection with the estuary for 20 years. During this time, silt has accumulated in the ponds and creek to a thickness of up to 60 cm. When the water layer over this silt dropped down to a few cm or so, we noticed fishes expiring any time the sediment was resuspended.

As the year progressed, the abundance of fishes declined, but the size distribution remained fairly constant (Figure 9, 10). This suggests that young were produced even during winter, and, thus, may be available to grow out in response to managed water level manipulation.

CONCLUSIONS

Even these early findings lead us to a few conclusions. The first is that the production of organic material in these impoundments must be considerable. Some may dispute the importance of marshes as significant sources of organic production to estuaries. However, the

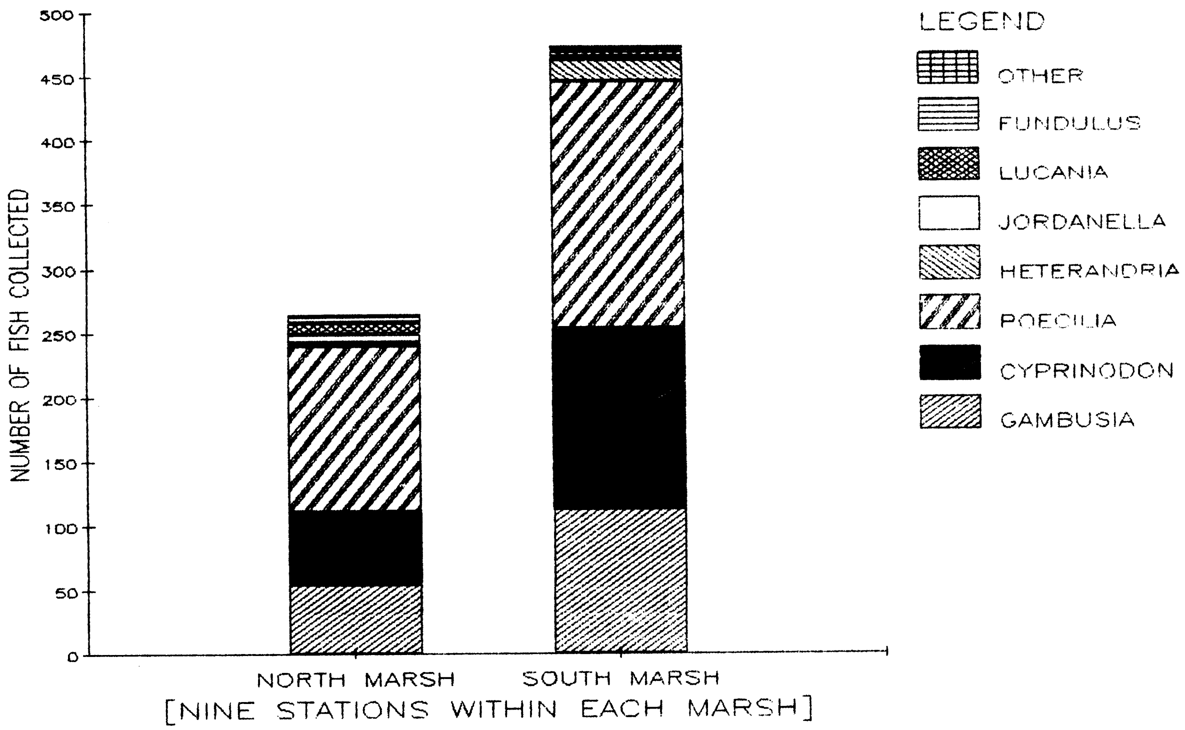

Figure 5. Totals of fish collections in Sykes Creek marshes.

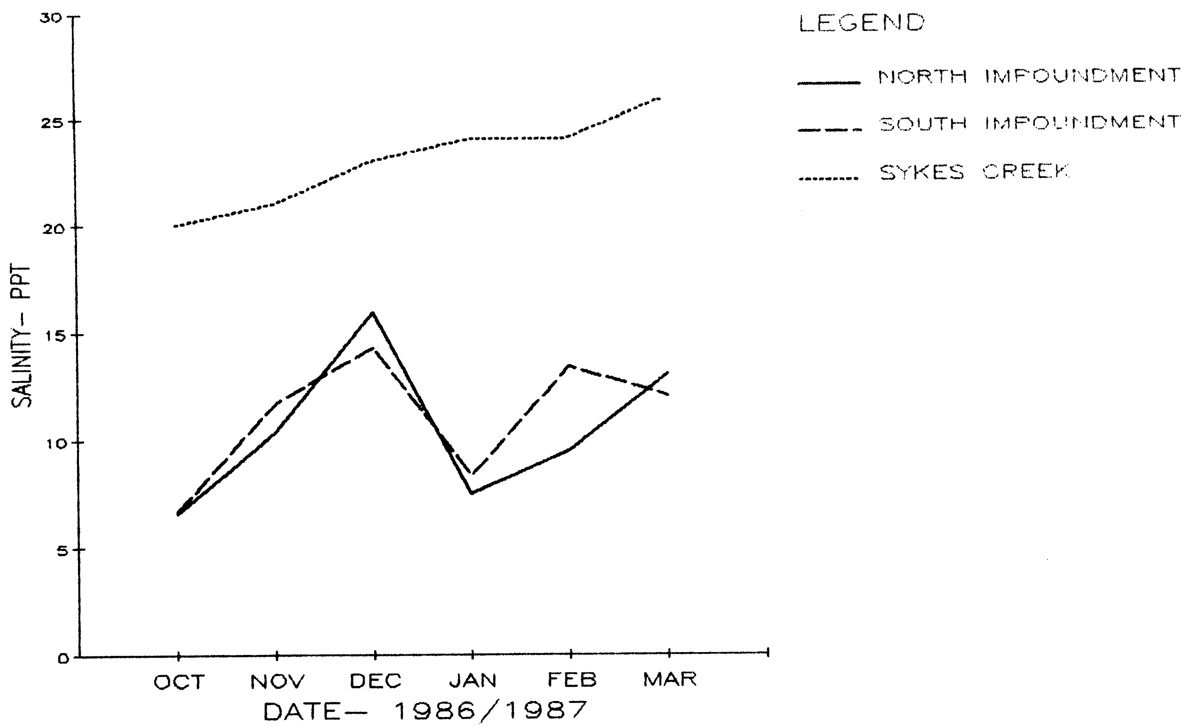

Figure 6. Salinity of the marsh and of Sykes Creek.

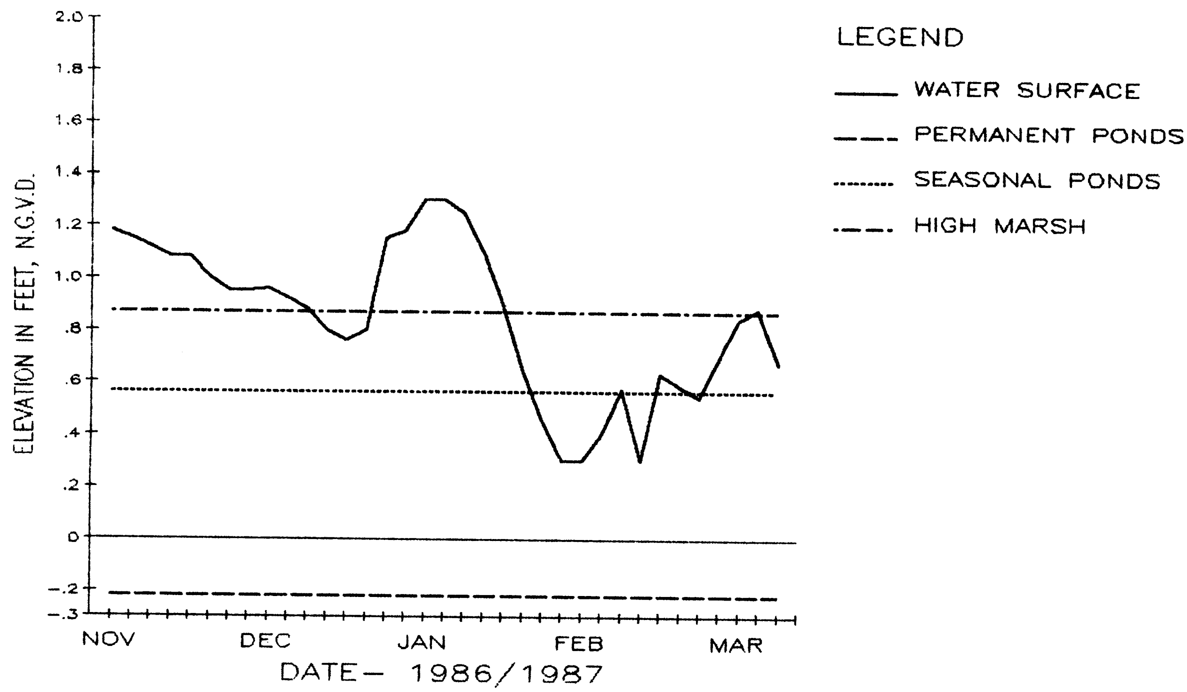

Figure 7. Water surface compared to substrate elevation for three marsh habitats.

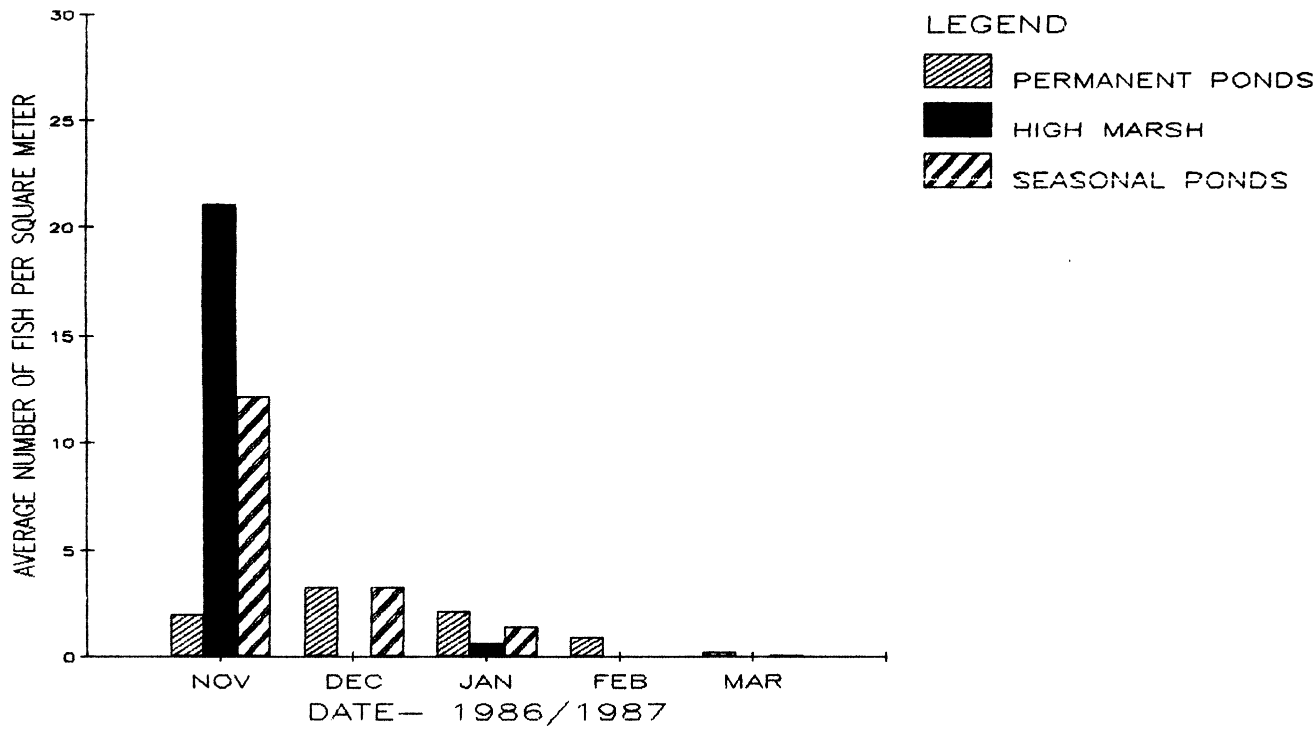

Figure 8. Fish density by marsh habitat type.

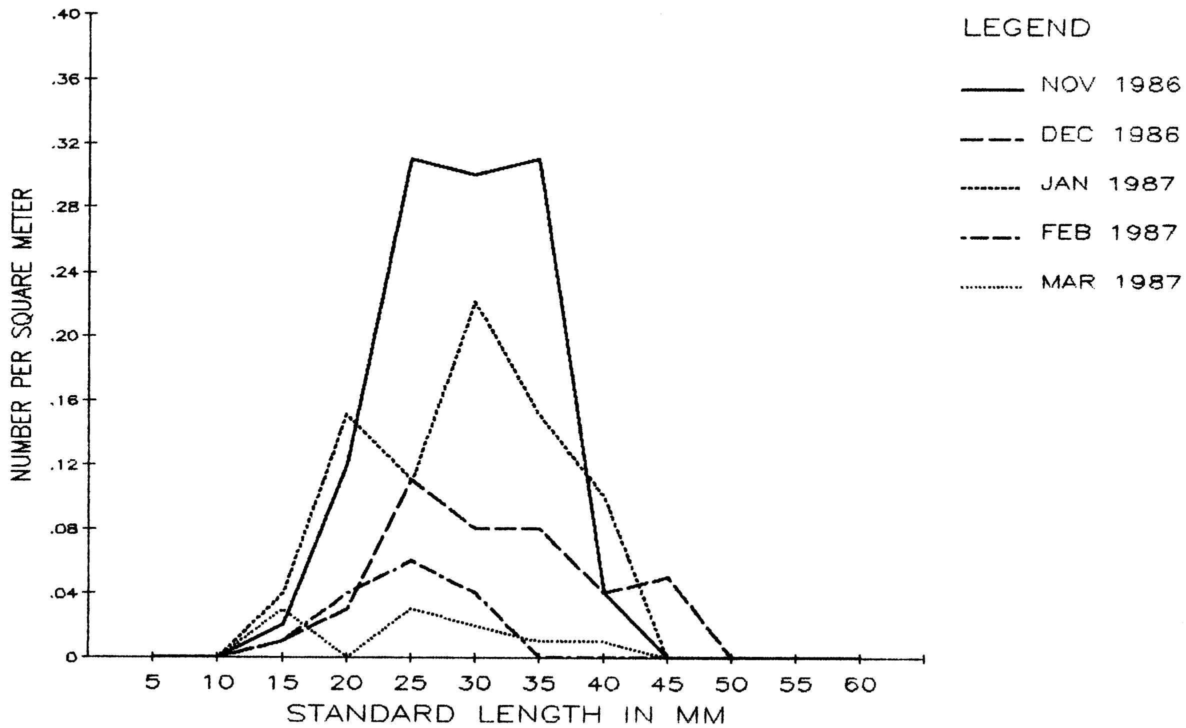

Figure 9. Size distribution of Syprinodon monthly collections.

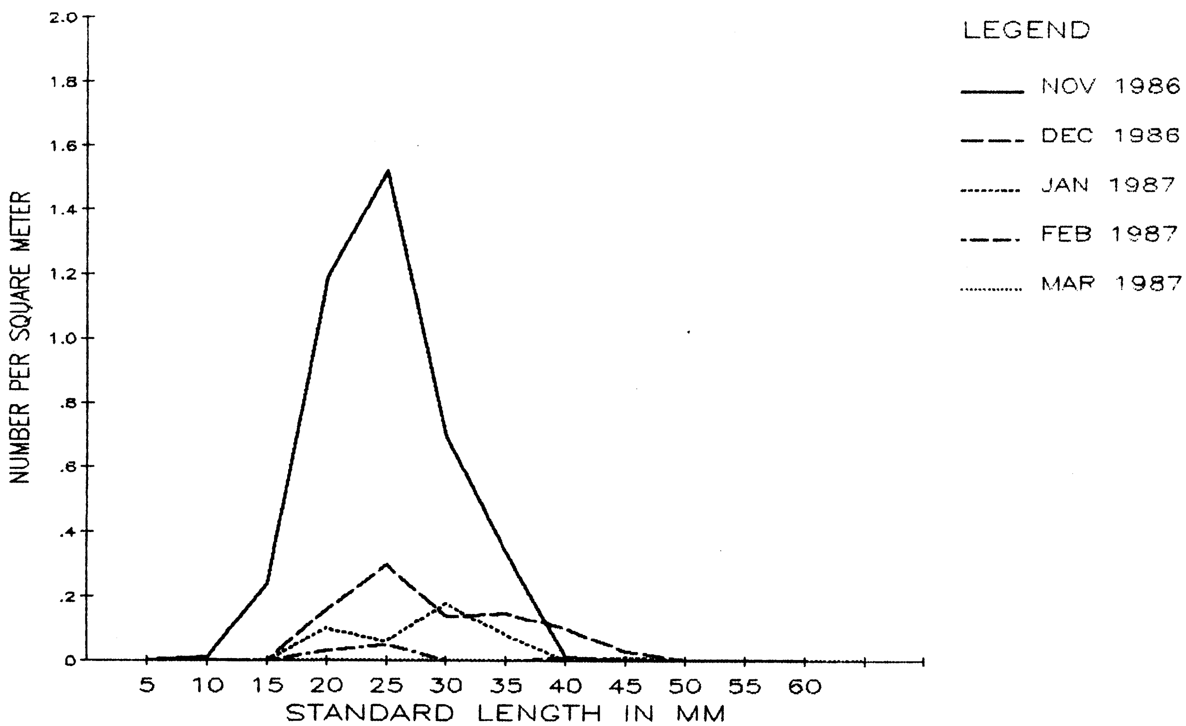

Figure 10. Size distribution of Poecilia monthly collections.

plant material which fills all depressions in these impoundments is generated internally, and once would have been flushed out to deeper, more open waters.

These impoundments appear to be a stressful environment because of high rates of organic production and accumulation. As water level declines, resuspension of fine material exerts a very heavy oxygen demand on the water column which is lethal even to killifishes. We expect that reconnecting the marshes to the estuary will ultimately flush out fine materials or allow them to oxidize because of more complete drying. Thus, the proposed management plan could result in improved habitat value by allowing access to estuarine fishes, and by allowing reconditioning of the waterways and improving the substrate.

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SEAGRASS ZONATION: TEST OF COMPETITION AND DISTURBANCE AT SEAHORSE KEY, FLORIDA

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ABSTRACT

The zonation of Halodule wrightii and Thalassia testudinum (which occur in adjacent, virtually non-overlapping monocultures in the intertidal and shallow subtidal respectively at Seahorse Key, Florida) was subjected to experimental manipulations. Transplantation was done above, within and between beds, and mean blade length (as a measure of plant health) was monitored. Upper limits appeared to be set by exposure (desiccation and/or heat) stress while the lower limit of Halodule was not set physiologically since transplants into clearings in the Thalassia bed flourished. Experimental evaluations of shading, leaf abrasion, short term allelopathy and root-root interactions were performed to isolate potential competitive mechanisms. Thalassia outcompeted Halodule via long-term root interactions. Herbivory were modeled by cropping at the border zone and was capable of decreasing the ability of Thalassia to exclude Halodule and enabling Halodule to invade. Other potential forces which could alter the outcome of the competition were discussed.

INTRODUCTION

Subtropical grassbeds along the coast of the Gulf of Mexico often exhibit zonation of the shallow, intertidal shoal grass, Halodule wrightii Aschers., and the deeper, subtidal turtle grass, Thalassia testudinum Banks ex Konig (Humm 1956; Phillips 1960, 1962; Moore 1963; Keller & Harris 1966; den Hartog 1970; Iverson & Bittaker 1986) though mixed beds are known to exist (Voss & Voss 1985; Humm 1956; Strawn 1961). Little experimental work has been done on the zonation of Halodule and Thalassia (but see Phillips 1960 as an early approach). The purpose of this research was to explore experimentally the causal mechanisms of the zonation of Halodule and Thalassia and to examine phenomena which could affect the zonation.

STUDY SITE

All experiments were performed on the south beach of Seahorse Key (83 04'West and 29 06'North), a small island 5 kilometers offshore from Cedar Key, Florida on the Gulf Coast. Stations were established at approximately 60 cm (Upper Beach), 30 cm (Middle Beach), 15 cm (Halodule bed) and -15 cm (Thalassia bed) from the mean tide level. These correspond to 2, 10, 30 and 80 meters respectively from the extreme high spring water (EHSW) mark. The Upper Beach station was located on a stretch of clean sand. Middle Beach was placed centrally on a muddy sandflat, and the grass bed stations were located 4 meters from their leading edges. Four stations were established along one transect in 1979 and another four stations were established along a parallel transect 10 m to the west in 1980. All grassbed stations were carefully raked to remove all plants and root systems, and the level of the sediment in the experimental plots (approximately 1 meter by 3 meters with the long axis of the plot paralleling the edges of the beds) was matched to the level of the sediment in the surrounding bed.

METHODS AND MATERIALS

The question of physiological limitations was addressed by transplanting grasses with a modified plug technique (Phillips 1976; Van Breedveld 1975). A steel sleeve (17.8 cm diameter by 30 cm long) was driven into the sediment at the transplantation site and the sediment within the sleeve was removed. A capped polyvinyl chloride (PVC) core (15.2 cm diameter by 30 cm long) with a lateral vent in the cap was placed over the grass to be transplanted, was pushed into the sediment 30 cm and the vent hole was closed. The core was extracted, carried to the sleeve and inserted. The sleeve was removed and sediment was packed around the core. The vent hole was opened and the core removed from the sediment. Care was taken to match the depth of the sleeve and the core so that the plug would be level with the surrounding sediment.

A method was required which would measure the relative health of the grasses and would not require destructive sampling of experimental plots. Techniques using the ratio of physiologically active to inactive chlorophyll, leaf stapling, photography or clip-and-recover methods were not applicable due to diatom contamination, the leaf morphology of *Halodule*, turbid water and the confounding effects of removing photosynthetic tissue, respectively. The only feasible measure of plant health was blade length (Phillips 1960). Both macrophytes have distinct morphologies which allow accurate measurements.

The first set of experiments addressed physiological limitations of the grasses and the potential impact of transplantation procedures. In 1979 and in 1980, 5 cores of each grass were moved to Upper Beach, and 10 cores of each grass were moved to the other three stations. In each year, all 70 cores were transplanted within one tidal cycle (August 1, 1979 and June 9, 1980). Qualitative data was gathered in both years and quantitative data was taken in 1980. In that year, between 10 to 15 plants were collected from each grass at each station and the approximately 30 blades from each grass and station were measured. Sampling periods increased from weekly, initially, to twice a month after one month.

The second group of experiments addressed potential competitive mechanisms. Relative shade tolerance was estimated by placing 2x and 4x neutral density platforms (1 meter square) 30 cm above plots of both grasses and monitoring the grass lengths. After one week, diatom growth had converted both filters into opaque surfaces. Grass lengths were monitored for 79 days (July 24 to October 11, 1980).

Leaf abrasion of *Halodule* by *Thalassia* was estimated by planting artificial *Thalassia* plants (3 strips of 4 mm clear polyethylene plastic 22 cm by 0.6 cm fastened to aluminum nails with rubber bands) at the normal density of *Thalassia* plants in an experimental plot in the *Halodule* bed. The monitoring period was the same as for the shading experiment. The artificial plants were allowed to remain for one year to qualitatively assess long-term effects.

To ascertain potential short-term allelopathic (e.g., toxic) interactions and to examine long-term root-root interactions, 10 cores of *Halodule* in 1979 and 10 more in 1980 were transplanted into the *Thalassia* bed. Except for damage caused by the transplantation procedure, *Thalassia* plants within the experimental plot were not disturbed. Five of the cores each year were enclosed in 30 cm long plastic sleeves (17.8 cm in diameter) and the other five were implanted without sleeves. The latter group was marked with small PVC stakes. The sleeves were set flush with the sediment, extended below the *Thalassia* rhizomes, and would presumably protect the enclosed plants from any root allelopathic effects while still exposing the plants to potential leaf interactions. Grass in the cores transplanted in 1980 were monitored for blade length for 75 days (July 18 to October 11).

The final effort addressed herbivory. Three 1 meter square plots were established on the border between the grassbeds (with the border bisecting each plot and with *Thalassia* to the south and *Halodule* to the north). The corners and the midpoints of all four sides of the three plots were marked with PVC stakes protruding 1 cm from the sediment. Each plot was effectively reduced to 80 cm on a side for experimental manipulations, leaving a 10 cm margin around the borders of the plots as a secondary control. The southeast quadrant (*Thalassia*) of the west plot, the west half (*Thalassia* and *Halodule*) of the center plot and the northwest quadrant (*Halodule*) of the east plot were cropped weekly from April 23 to August 20, 1981. Grass was clipped with garden shears 2 to 3 cm above the substrate mimicking the grazing activities of green turtles (Bjorndal 1979). After 119 days of simulated herbivory, the plots were divided into four quadrants. Twenty-five core samples (5 rows of 5 cores, each being 10 cm in diameter) were taken from each quadrant (300 cores total). The number of emergent shoots and the ash-free dry weight (weight difference before and after combustion at 550°C for 3 hours) of each grass in each core sample was determined.

RESULTS

The responses of *Halodule* and *Thalassia* to the transplantation treatments through time are presented in Figure 1. Covariance analyses

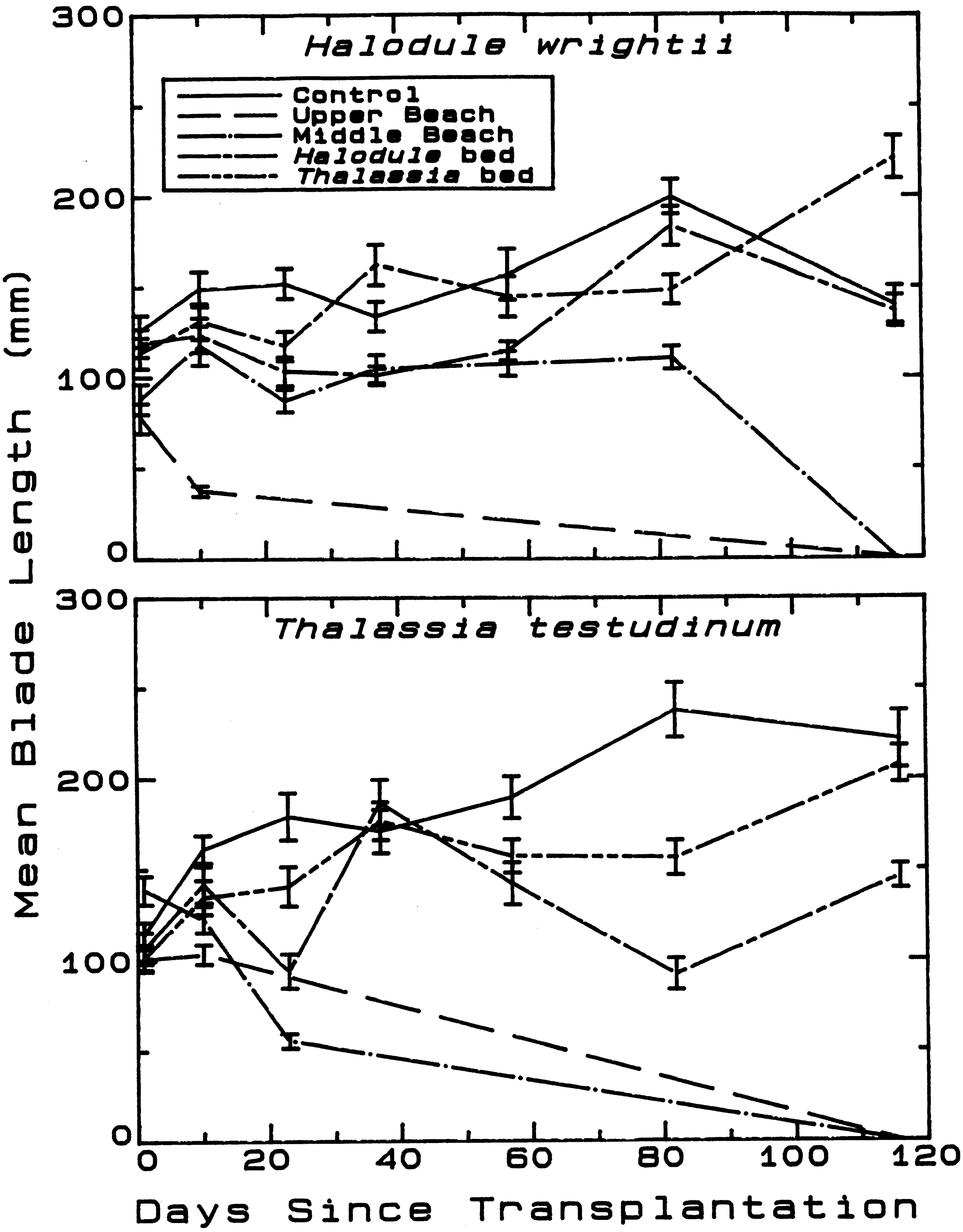

Figure 1. Response to Halodule wrightii (A) and Thalassia testudinum (B) to transplantation along a tidal gradient (Upper Beach > Middle Beach > Halodule bed > Thalassia bed in terms of tidal depth). Vertical bars represent two standard errors.

were performed between experimental and control treatments ($\alpha = 0.05$). Both grasses rapidly perished at the Upper Beach station and Thalassia perished after 14 more days at the Middle Beach station. Leaves of both grasses initially turned brown and became visually indistinguishable from detrital blades except that they were still attached to the rhizomes (mean blade length then underestimates the degradation of the plants and is, therefore, a conservative measure of plant health). Halodule at Middle Beach eventually perished but it did persist for an appreciable period of time. Transplants of Thalassia into the Halodule bed persisted for the duration of quantitative measurements but the blades were discolored by brown patches and eventually the plants disappeared. All Halodule transplants into either grassbed and Thalassia transplants into its own bed appeared normal throughout the experimental period.

The regressions of blade length through time for both grasses at the upper two stations were noncoincident with the control samples (mean length being 53% and 61% relative to the control for Halodule and Thalassia respectively). The regression of Halodule transplanted into the Halodule bed paralleled the control samples (mean length of 83% of the control) while the regression of Halodule transplanted into the Thalassia bed was noncoincident with the control (mean length of 97% but with a steeper positive slope). Thalassia transplants within the Thalassia bed paralleled the control (86% of mean length) while the regression for those in the Halodule bed were noncoincident (at 73% of mean length). The parallel (but lower) regressions of transplants compared to control regressions can be interpreted as being due to the initial impact of transplantation.

The results of the manipulative experiments were uniformly negative. While shading platforms did not completely block light, insolation was appreciably and equally reduced at both sites. After 79 days, there was no statistically demonstrable damage to either grass. Leaf abrasion by Thalassia was found to have no discernable effect on Halodule. The artificial plants were left in place for one year and even after that period, no perceivable damage had taken place. Transplants of Halodule which were not protected by plastic sleeves into the Thalassia bed were statistically indistinguishable from the control samples while those that were protected exhibited greater transplantation damage but no evidence of competitive interactions (all tests were covariance analyses at $\alpha = 0.05$). Thus, interactions by shading, abrasion, or short-term allelopathic effects were not found to occur.

In 1979 and in 1980, 10 plugs were tested for short-term allelopathic effects and the plugs were allowed to remain undisturbed for one year (20 total, 10 with and 10 without sleeves). After that year, a distinct and statistically valid difference did exist between treatments (Fishers Exact Probability test for two independent samples with $\alpha = 0.05$). Eight of 9 plugs protected by sleeves persisted (the tenth plug had been excavated by a blue crab and that plug was excluded from analysis). Of the cores without sleeves, 9 of the plugs perished. Since the location of each plug was marked, the sediment transplanted with the grass could be examined. In all cases in which the transplanted Halodule perished, rhizomes of Thalassia interdigitated the sediment. The unprotected plug which persisted was located in a small clearing and the sediment under it was free of Thalassia rhizomes. The competitive mechanism thus appears to require root-root interdigitation.

Two sets of disturbance experiments were performed (one inadvertently and one by design). The Thalassia bed stations were established 4 meters into the Thalassia bed from the border. Sampling was done on all visits to the site in 1979 and 1980 at high tide. In August of 1981, the site was visited during a spring low tide (0.06 m below mean tide level). A tongue of Halodule was found to extend from the old border zone 2 meters towards the 1979 station and 4 meters towards the 1980 station. Apparently, foot-traffic generated by sampling in 1979 and 1980 damaged either the rhizome system and/or the blades sufficiently to allow invasion by Halodule and to cause a noticeable decrease in the density of Thalassia.

The results of the herbivory (cropping) experiment are clear (Figure 2). Cropping resulted in an increase of Halodule in the Thalassia bed and softened the sharp decline at the border between the beds. Conversely, the extent of Thalassia in the Halodule bed declined and the border between the beds sharpened for Thalassia. Statistically (Newmans-Keuls Multiple Range Test at $\alpha = 0.05$), there was no difference between Halodule in control and cropped area in the Halodule bed while Halodule numbers in the cropped area were greater than the control samples in the Thalassia bed. For Thalassia, there was a significant difference (a decrease) between the control and the cropped areas in both beds. The patterns derived from analysis of biomass paralleled all of the above and are not presented here.

DISCUSSION

Seagrass distributions have long been of interest. One prominent feature of subtropical seagrass beds is the zonation of Halodule wrightii and Thalassia testudinum. Just as in algal systems where physiological limitations were first postulated to control zonation and only later were the roles of biological interactions appreciated (Lubchenco 1980), seagrass zonation patterns have been attributed to physiological limitations (Strawn 1961). Suggestions have been made that the relationship might be competitive (Phillips 1960; den Hartog 1970). As Dayton (1973) has strongly asserted, assuming the operation of an ecological mechanism (competition, predation, etc.) solely on the basis of descriptive and correlative data can be highly misleading and experimentation is often required for verification.

Desiccation has often been identified as the factor controlling the upper limit of seagrasses (den Hartog 1970; Humm 1956; Keller & Harris 1966; Moore 1963; Phillips 1960, 1962). At Seahorse Key, desiccation (drying) and head stress are typically paired and all heat

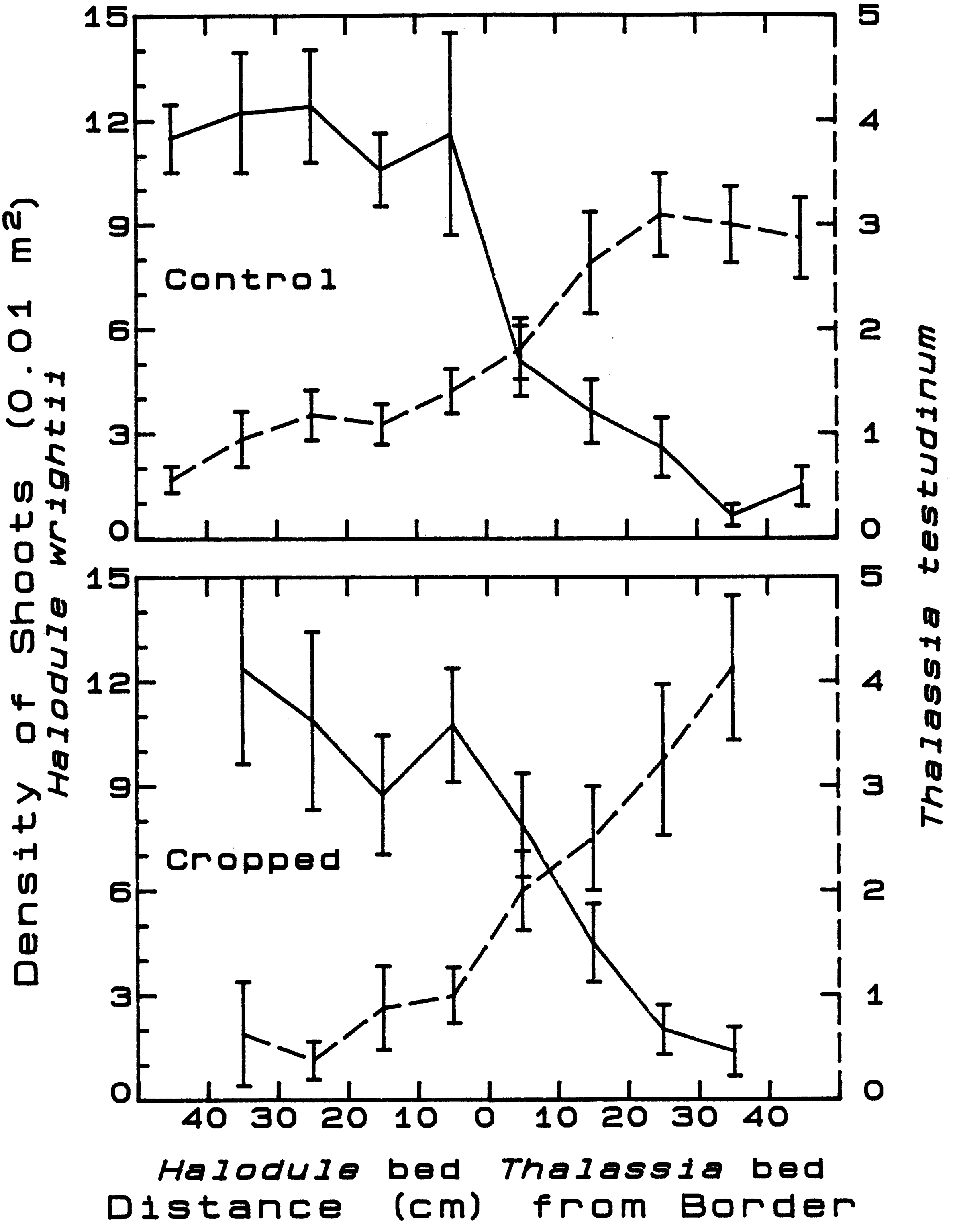

Figure 2. The pooled result of weekly cropping of Halodule and Thalassia in three experimental 1 meter square plots centered on the border between the grassbeds for a period of 119 days. Vertical bars represent two standard errors.

effects are subsumed within the term "desiccation" in this paper. However, see McMillan (1984) for systems in which the two effects can be separated. *Halodule*, relative to *Thalassia*, is characterized by a high volume-to-surface ratio due to narrower blades (0.5-1.5 mm versus 4-12 mm respectively) (den Hartog 1970) and a greater ability to form moisture-retaining mats due to a greater density of emergent shoots and the ability of those shoots to lie closer to the substrate (Phillips 1960). Thus, the morphologies of the grasses lend credence to the concept that *Halodule* is more desiccation-resistant and is able to penetrate further into the intertidal than *Thalassia*. The responses and the rapidity of those responses of *Halodule* and *Thalassia* to transplantation into water shallower than their respective grassbeds fully support the contention that the upper limit of the grasses is set by desiccation stress. *Thalassia* was more rapidly eliminated at either Upper Beach or Middle Beach than was *Halodule* and was demonstrably harmed when existing in clearings in the *Halodule* bed, but was able to overcome transplantation damage when placed in its own bed. *Halodule* was able to tolerate conditions at Middle Beach longer than *Thalassia* and was able to overcome transplantation damage in both grassbeds.

Based on quantitative transplantation data, the appearance of the grasses, and the distributional features of the beds (the leading edges trace depth contours), desiccation stress sets the upper limits of both beds. The presence of the *Halodule* bed with its moisture-retaining ability, however, does allow the *Thalassia* to intrude somewhat into the *Halodule* bed (Figure 2).

While this study did not address the factor(s) which control the penetration of *Thalassia* into deeper water (although light intensity and its correlate, water turbidity, have often been identified as causal) (den Hartog 1970; Phillips 1960), it has unequivocally demonstrated that *Halodule* can persist and thrive in clearings in the *Thalassia* bed at greater depths than the border zone of the zonation (Figure 1 & 2). Physiological limitations do not limit *Halodule* from extending its range from its observed limit at Seahorse Key to deeper waters. Causal sediment characteristics are highly unlikely given the invasion of *Halodule* into the *Thalassia* bed along paths to the *Thalassia* bed stations or within the herbivory experiments. If physiological limitations and sediment distributions can be eliminated, biological interactions must be considered.

Of the two major biological interactions which are known to control organismal distributions, predation and competition, predation does not appear to be of importance at Seahorse Key. There are relatively few herbivores which directly consume seagrass (Moore 1963; Randall 1965; den Hartog 1970; Ogden 1976) although herbivores are known to affect seagrass distributions (Camp et al. 1973) and the vigor of cropped plants (Phillips 1960). Of the major herbivores, reef fishes do not occur at Seahorse Key, turtles and manatees are rare, and while sea urchins are common in adjacent channels, they have not been observed in the grassbeds south of Seahorse Key. Furthermore, there has not been any evidence of cropping, i.e. clipped blades (Greenway

1976), grazed areas (Camp et al. 1973; Randall 1965), bitemarks (Greenway 1976) or evidence of differential epiphyte grazing (Howard & Short 1986). Since herbivory does not appear to be occurring to any noticeable extent, competition is implicated.

Competitive mechanisms between plants fall into several categories: (1) competition for light (Dayton 1975); (2) physical interactions via abrasion or whiplash (Dayton 1975); (3) allelopathic effects (Krebs 1972); and (4) root-root interactions including nutrient competition and root crowding (Richlefs 1976). Shading has been identified as a potential intraspecific competitive mechanism (McRoy & McMillian 1977) and thus could be an interspecific mechanism. Since Thalassia has a broader blade and, due to the structure of the emergent shoot, stands higher than Halodule, Thalassia has the potential of outcompeting Halodule for light. The results of the shading experiment indicated that the potential is not realized. Given the reality of shifting currents and high turbidity, it is highly unlikely that Thalassia could outshade Halodule.

Leaf interactions by abrasion were quantitatively assessed for 79 days and qualitatively assessed for over a year. In neither instance was any evidence generated to support leaf abrasion as an effective competitive mechanism.

Short-term allelopathic effects could be due to soluble, dispersing compounds or to contact-toxins, and Halodule is known to exude significant amounts of fixed organic carbon into the sediment (Moriarty et al. 1986). The presence of soluble compounds was tested by exposing one cohort of Halodule to Thalassia root-systems and protecting another cohort from contact. The results were unambiguous. Halodule experienced no deleterious effect due to exposure to Thalassia over the experimental period. This result was not unexpected since mixed beds and some interdigitation at the border zone do exist. If there were soluble compounds, a bare zone would be predicted.

Long-term effects did exist. Plugs of Halodule exposed to Thalassia foot-systems for one year disappeared while those which were protected persisted. In all cases where Halodule vanished, Thalassia root systems interdigitated the sediment of the Halodule plug. This effect could be due to a number of mechanisms which are not distinguishable by the experimentation used here, but all involve slow root-growth of Thalassia and the close proximity of the root-systems of the two grasses.

The distinct zonation observable at Seahorse Key appears to be a result of Thalassia being unable to penetrate into the intertidal due to physiological limitations (desiccation resistance) and the inability of Halodule, which can survive in the intertidal, to penetrate into the subtidal due to being outcompeted by Thalassia. Mixed beds, however, are known to exist (see Introduction for references) and a discussion of zonation would be incomplete without consideration of the mechanisms such as tidal gradients and disturbance, which can obscure the zonation pattern.

One potential generator of mixed beds is the slope of the substrate. Given the desiccation-ameliorating effects of the Halodule bed, there should be an extension of Thalassia into the Halodule bed effectively creating a zone-of-confusion along the border extending over a few centimeters of tidal height and a mixed bed over some horizontal distance. If the slope would double, the horizontal extent would be halved, or if the slope became twice as shallow, the horizontal extent would double. At Seahorse Key, a raised bank approximately 15 m wide parallels and borders the main channel. The broad flat bank occurs at the same tidal height as the grassbed border zone and is covered by a mixture of both grasses.

Disturbance can also cause mixed beds or tongues of Halodule extending into the Thalassia bed by decreasing the competitive ability of Thalassia and thereby allowing Halodule to invade and coexist in the Thalassia bed. Sources of such disturbance include physical disruption due to propeller damage or waders, defoliation due to storms or environmental fluctuations, or reduction in photosynthetic tissue by herbivory. Zieman (1976) has discussed the effects of propeller damage on grassbeds. At Seahorse Key at the border zone, new scars are devoid of vegetation, scars of moderate age often have Halodule extending up the scar into the Thalassia bed, and old, deep scars occasionally have Thalassia extending into the Halodule bed along the bottom of the trough made by the propeller. Similarly, the observation that weekly wading along the transects across the grassbeds allowed tongues of Halodule to extend from the border zone to the Thalassia bed stations indicates that low-level but repeated physical disruption can affect the ability of Thalassia to exclude Halodule.

Storm damage can also result in mixed beds. The degree of damage is obviously a function of the propensity of the blades to separate from the rhizomes, and the storm intensity. Thalassia blades are easily broken from the rhizome (Tomlinson 1972) and this grass can be expected to be more severely impacted by the storms than Halodule. While the extent of storm damage has ranged from slight (Oppenheimer 1963) to reports of whole plants being ripped from the bottom (Moore 1963; Reid 1954), the normal impact of storms is defoliation (den Hargot 1970). Reid (1954) described the hurricane of 1950 in the Cedar Key area as having winds greater than 100 mph, being accompanied by 24 inches of rain (causing a salinity drop from 23.5 to 9.7 0/00 for 4 days), and being responsible for substantial seagrass disruption. A survey of seagrass distributions was carried out in the same area after only a few months (Strawn 1961) and documented that Halodule extended through the Thalassia bed. The same area was examined qualitatively in 1980 and zonation of the grasses was visible. Strawn's (1961) observations could be due to defoliation of the grassbeds in 1950 due to the hurricane, followed by a rapid colonization of the disrupted zone by Halodule (which has a faster growth rate and greater dispersal abilities than Thalassia (den Hartog 1970). After the Halodule became established throughout the Thalassia bed, only gradually and over a period of years would *Thalassia* outcompete *Halodule*. Large scale mixed beds on tidal gradients where zonation would normally be expected may well be due to historical events such as storm damage or, conceivably, extreme salinity fluctuations which act to depress *Thalassia* sufficiently to allow the invasion of *Halodule*. Given the slow recovery rates of *Thalassia* (Zieman 1976), years may pass until the zonation is reestablished.

The final source of disturbance which could obscure the zonation pattern is herbivory. While relatively few organisms directly consume seagrass (see above for references), herbivores can exert a disproportionate influence on plant distributions (Vance 1979; Duggins 1980) and are known to affect seagrass beds (Camp et al. 1973; Randall 1965). The procedure of weekly clipping the seagrass a few centimeters above the substrate mimics the feeding activities of green turtles (Bjorndahl 1979). Turtles repeatedly mow a given area and consume the regenerating blades. While the maximum population densities of turtles in the grassbeds of the Florida Gulf Coast are unclear (Carr & Ingle 1959), turtles may have had a major impact on the organization of seagrass beds (Randall 1965), and their form of herbivory is far less deleterious to the grass than the disruptive activities of birds (McMahan 1968) or manatees (Rathbun pers. comm.), which rip rhizomes from the substrate, or of sea urchins which can obliterate beds if they occur in dense aggregations (Camp et al. 1973). This form of herbivory was chosen to minimize damage to the grasses while simulating herbivory, and the results are clear. Whether measured by number of emergent shoots or total plant biomass, cropping of *Thalassia* reduced the penetration of *Thalassia* into the *Halodule* bed and permits the invasion of *Halodule* into the *Thalassia* bed. Since *Thalassia* plants on the border or plants which occur in the *Halodule* bed are stressed due to exposure, additional stress due to cropping could overwhelm the plants and result in their decline or disappearance. Plants not removed by the cropping must divert energy from root growth and competitive interactions into the regeneration of photosynthetic tissue and must contend with less energy production during the regeneration phase. *Halodule* with its faster growth rate should be able to tolerate moderate cropping and invade the *Thalassia* bed when *Thalassia* is no longer able effectively to exclude *Halodule*.

The phenomenon of zonation due to competition being broken or obscured by selective predation on the superior competitor is well established and well documented (see Dayton 1975 for an extended list of examples, and Mann 1973 and Vadas 1968 for marine plant examples). The effect of herbivory on seagrass zonation follows the same pattern but does not require selective predation. If the superior competitor, in this case *Thalassia*, is differentially harmed by herbivory due to a slower recovery rate, equal predation can occur on both grasses with the net effect of decreasing the competitive ability of *Thalassia*. Mixed beds could thus be maintained by herbivores which simply feed on the mixture of grasses without selecting one or the other.

The suggestion exists that *Halodule* acts as a classical pioneer or early successional species and conditions the sediment (Phillips 1974) which then allows *Thalassia* to invade (Van Breedveld 1975). It is also possible, however, that seagrass succession is an example of tolerance succession rather than facilitation succession (see Connell & Slatyer 1977). *Halodule* may simply appear as an initial colonizer due to its high dispersal ability and rapid growth rate. If *Halodule* were present or absent, *Thalassia* might well follow the same schedule of invasion and grassbed establishment. Transplantation experiments can separate these possibilities.

In summary, the obvious zonation of *Halodule* and *Thalassia* in intertidal and shallow subtidal habitats is due to *Thalassia* being limited by desiccation/heat-stress to the subtidal or deep intertidal. *Halodule*, being more desiccation resistant, can extend farther into the intertidal and forms a monoculture there. Under conditions of little or no disruptive stress, *Thalassia* is able to outcompete *Halodule* via long-term root-interactions and to restrict *Halodule* to the intertidal. The steeper the slope, the more pronounced is the border zone between the grasses. If small-scale disruptions (propellers or waders), large-scale disruptions (storms or extreme environmental fluctuations) or herbivory occur, mixed beds may result and may persist for years. If such disturbances occur more frequently than the recovery period, mixed beds may persist indefinitely.

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PRELIMINARY REPORT ON TRANSPLANTING OF THE BENTHIC GREEN ALGA *CAULERPA PROLIFERA*

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ABSTRACT

The benthic green alga *Caulerpa prolifera* (Chlorophyta: Siphonales) is frequently found in Tampa Bay seagrass beds, and forms extensive monospecific stands in some parts of the Bay. Per the requirements of the Florida Department of Environmental Regulation, an experimental transplanting project was designed and implemented in conjunction with the maintenance dredging of the MacDill Air Force Base marina entrance channel. Three plots of transplanted material were established using 20 cm square mats of *Caulerpa* installed on centers of approximately 0.3 m, 0.6 m and 1.0 m. Persistence and survival of transplanted material was 85-90% (estimated) ten days following transplanting. Five months later, survival was 100% of material installed on 0.3 m centers, 79% of material on 0.6 m centers, and 72% of material on 1.0 m centers. Results suggested that transplantation of this alga is both feasible and cost-effective. The ease with which transplantation can be accomplished has two principal applications: (1) this plant can be used to vegetate submerged areas quickly and in a cost-effective manner; and (2) transplantation of this alga can be employed to test hypotheses regarding the role that it may play as an early successional precursor to seagrass colonization or its possible competition with seagrasses.

INTRODUCTION

Siphonaceous green algae are benthic, rhizophytic algae which are important components of subtropical and tropical submerged macrophyte communities. However, their ecological roles are poorly characterized. Their level of primary production may be comparable to that of seagrasses (Zieman & Wetzel 1980). In tropical areas, preliminary data suggest they may behave as pioneer species, initially colonizing bare substrate and preparing the area for subsequent colonization by seagrasses (Zieman 1982). *Caulerpa prolifera* (Chlorophyta: Siphonales) is a common component of Tampa Bay seagrass beds, and forms some large monospecific stands in parts of the Bay. In particular, a large stand exists along the southern shore of the Interbay peninsula in the vicinity of MacDill Air Force Base.

In accordance with a Florida Department of Environmental Regulation permit issued for the maintenance dredging of the MacDill Air

Force Base marina channel, a total of 0.1 hectare of *Caulerpa prolifera* was removed from within the project limits (Figure 1) and transplanted to an appropriate off-site location. This was done both to minimize potential adverse environmental impacts associated with the dredging of the channel, and as an experiment to investigate the feasibility of *Caulerpa* transplantation. The planting area was located approximately 1.2 km west of the donor site, just offshore of existing *Caulerpa* beds.

MATERIALS AND METHODS

Three experimental plots (1-3) were established in the field as shown in Figure 1, and planted on designated centers according to the following: Plot 1 was 16 m square and received approximately 700 units of *Caulerpa* on 0.6 m centers; Plot 2 was 22 m square and received approximately 700 units on 1.0 m centers; and Plot 3 was 8 m square and received approximately 600 units on 0.3 m centers. An average planting unit consisted of a 20 cm square mat of *Caulerpa* affixed to the substrate with two 15 cm long steel staples (see Figure 3). The initially proposed scope of work specified the use of two sizes of planting material, 10 cm plugs and 30 cm mats, but the vegetative nature of this plant (i.e. vine-like branched growth form, and very shallowly rooted) made it difficult to produce these two types of planting unit.

Salvage and transplanting operations were conducted in August 1986. Sampling of each plot was conducted at time zero (immediately following completion of planting) and at five months post-planting in order to assess *Caulerpa* survival. A qualitative site inspection was made ten days after planting to assess short term plant survival. Monitoring consisted of replicate sampling in each plot with haphazardly placed 0.25 m^2 quadrats. Within each quadrat, the number of planting units were counted and the percent cover by *Caulerpa* estimated. In addition, photographs were taken to document growing conditions, and other pertinent observations were recorded.

RESULTS

Monitoring results obtained in August 1986 (time zero) and January 1987 (five months post-planting) are presented in Figure 4. Plot 3 clearly exhibited the highest survival rate and increase in vegetative cover, compared with Plots 1 and 2. The initial 20 cm *Caulerpa* planting units placed on 0.3 m centers had completely coalesced to form a dense bed of algae (see Figures 5 and 6). Therefore, survival in this plot was estimated to be 100%.

Although plots 1 and 2 achieved somewhat lower survival rates, 79% and 72% respectively, many of the original plantings remained largely intact (Figures 7-10. Some of the units appeared to be viable and to have expanded several centimeters in diameter, while others appeared to be dormant or to have exhibited a slight decrease in biomass. An

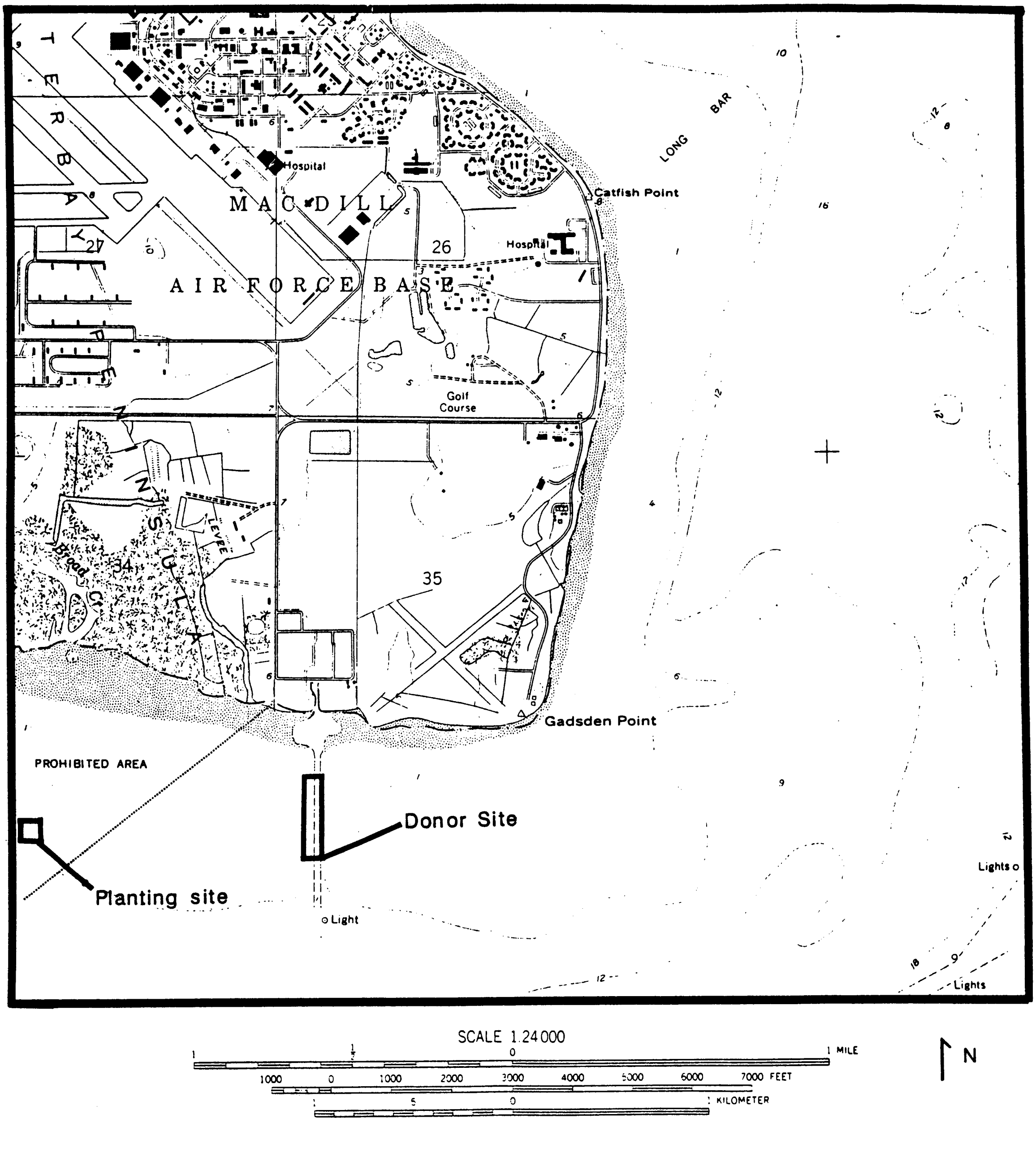


Figure 1. Location of the donor site and planting site for the transplantation of Caulerpa prolifera.

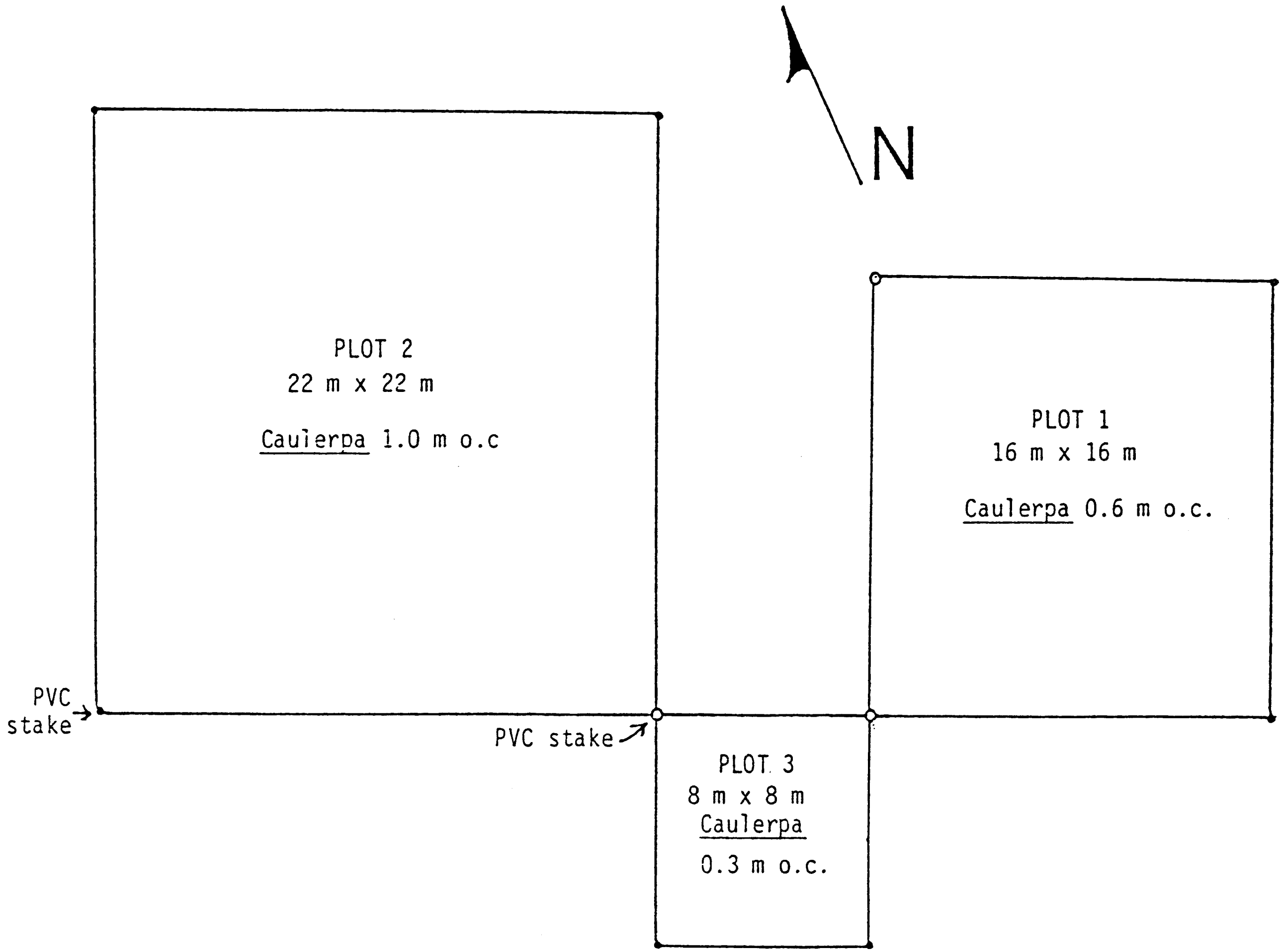


Figure 2. Diagram of the planting site plot configuration.

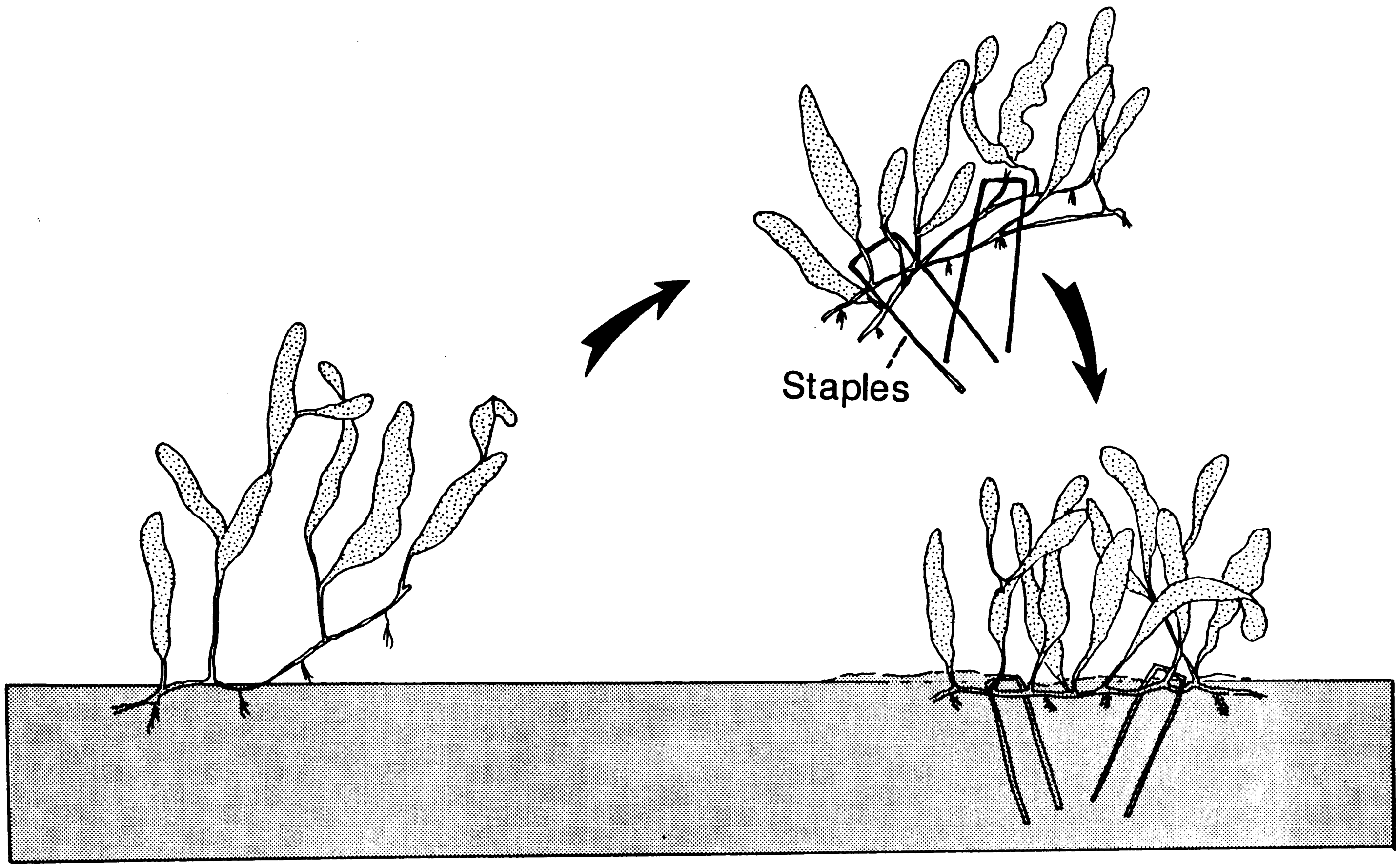
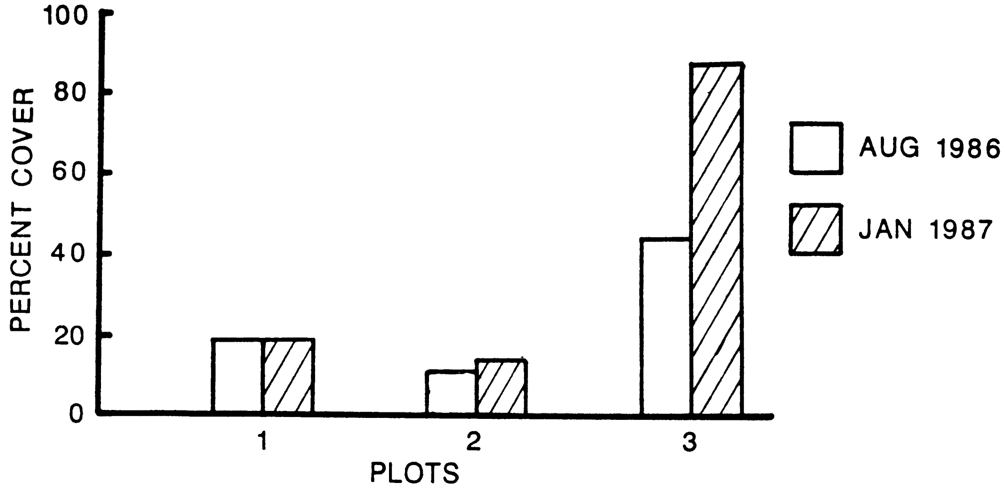


Figure 3. Diagram of the harvest and installation techniques used in transplanting *Caulerpa prolifera*.



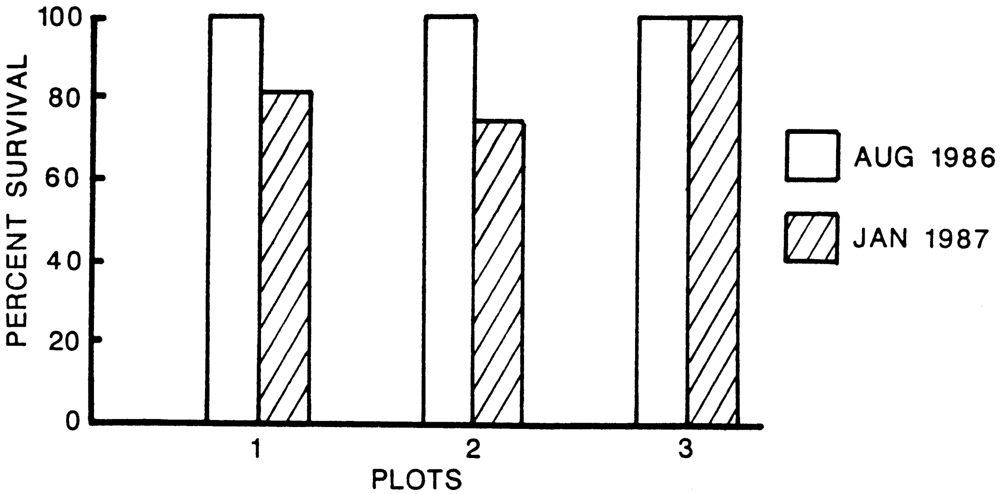


Figure 4. Mean percent cover and survival rates of *Caulerpa* in the three plots when planted (August 1986) and five months later (January 1987).

Figure 5. Plot 3 (units planted on 0.3 m centers) at the time of initial *Caulerpa prolifera* installation, August 1986. Staked line in center was utilized to maintain uniform plant spacing.

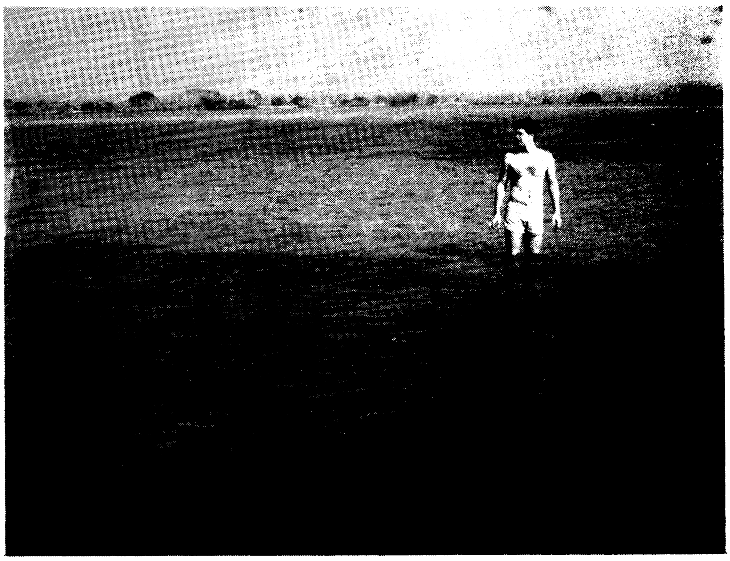


Figure 6. Plot 3, January 1987 (five months post-planting). Note the complete coalescence of *Caulerpa prolifera* planting units within the plot.

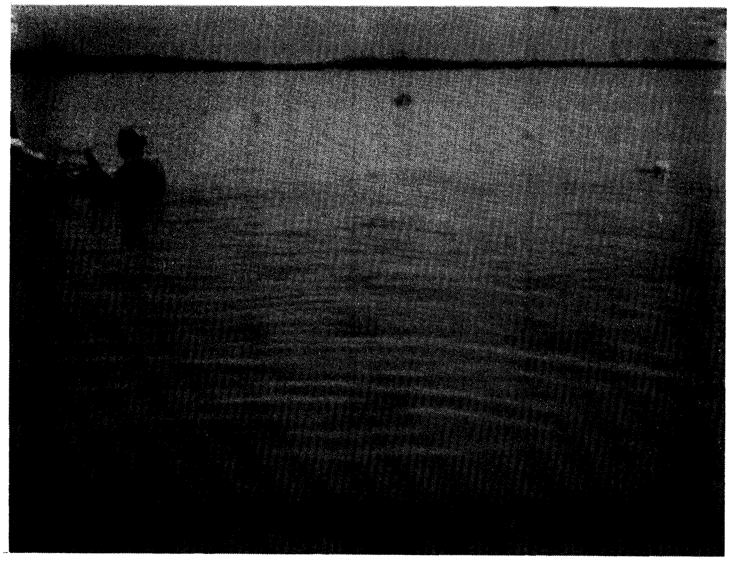


Figure 7. Plot 1 (units planted on 0.6 m centers) at the time of installation, August 1986.

Figure 8. Plot 1, January 1987. Many of the Caulerpa prolifera units have remained and expanded vegetatively within the plot.

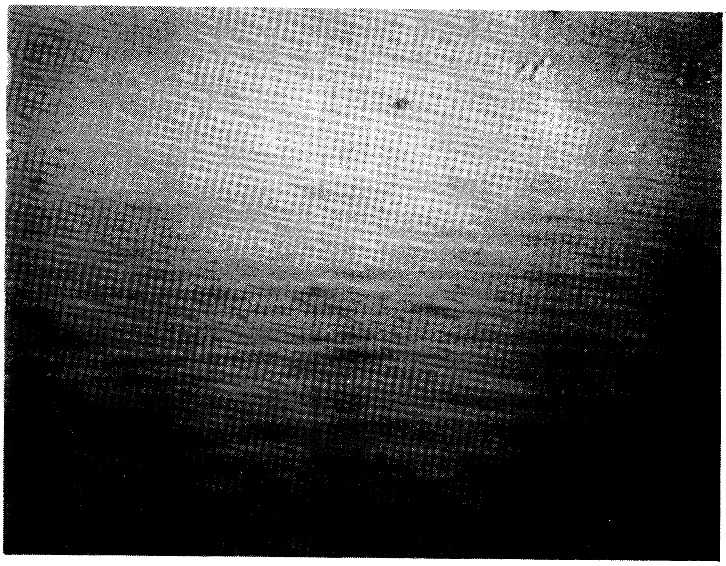


Figure 9. Plot 2 (units planted on 1.0 m centers) at the time of installation, August 1986.

Figure 10. Plot 2, January 1987. Although many *Caulerpa prolifera* units remain and had grown to some extent, the wide spacing in this plot prevented unit coalescence.

undetermined number of units in both plots had been uprooted, leaving bare areas. However, *Caulerpa* vegetative percent cover in these plots remained essentially *unchanged*, due to growth of the surviving units.

Qualitative sampling of epibenthic fauna in the Plot 3 *Caulerpa* bed was conducted using a 2 mm mesh dipnet. A number of invertebrates and fishes were collected. A preliminary list of identified species is presented in Table 1. Similar sampling of adjacent unvegetated bay bottom produced relatively few organisms.

Table 1. List of epibenthis organisms collected by dipnet in Plot 3 *Caulerpa* bed and adjacent unvegetated substrate in January *1987*.

CRUSTACEANS
- *Gammarus* sp.
- *Tubiculous* amphipods (2 spp.)*
- *Erichsonella filiformis*
- *Cymodoce faxoni**
- *Paracerceis* sp.*
- *Hippolyte zostericola**
- *Latreutes parvelus**
- *Palaemonetes* sp.
- *Neopanope texana**
- *Pagurus longicarpus**
- *Pagurus pollicaris**

MOLLUSCS
- *Melongena corona*
- *Mitrella lunata**
- *Oliva sayana*
- *Nassarius vibex*

FISHES
- Blennies (2 spp.)*
- *Gobiosoma* sp.*

*Collected only in *Caulerpa* bed.

DISCUSSION

Three factors directly associated with plant density are proposed to account for the more successful establishment of *Caulerpa* in Plot 3 versus Plots 1 and 2. Higher initial plant density *may have* provided for increased stabilization of (1) reducing current velocity in the vicinity of the individual planting units, which in turn prevented currents and shifting sediments from displacing the material; (2) decreasing biotic disturbance by discouraging ray foraging activities within the plot, as rays generally prefer more open, less vegetated, substrate (several rays were observed uprooting planting units in the other experimental plots); and (3) reducing the time required for adjacent units to coalesce, thus providing a larger, more stable rhizome matrix.

A review of seagrass planting literature (Continental Shelf Associates 1982; Fonseca et al. 1985; Fonseca, unpublished; Lewis & Phillips 1981; Mangrove Systems 1985) disclose harvest rates for seagrass planting units of from 25 to 500 units per manhour of effort, and planting rates of 20 to 75 units per manhour. *Caulerpa* planting units in this project were harvested at a rate of 675 *units* per manhour and planted at the rate of 75 per manhour. Thus, the use of Caulerpa on a unit effort basis is comparable to the most cost-effective methods of seagrass restoration, with survival rates equivalent or superior to rates of seagrass planting unit survival (Continental Shelf Associates 1982; Fonseca et al. 1985; Mangrove Systems 1985).

Caulerpa has been postulated to act as an ecological "pioneer species" in a successional sequence leading to development of a seagrass community (Williams 1981, in Zeiman 1982), invading unvegetated substrate and enhancing conditions for subsequent seagrass colonization. This hypothesis may be tested experimentally using Caulerpa transplantation.

It has also been suggested (Durako, pers. comm.) that Caulerpa may in some instances compete with seagrasses for growing space. This may also be tested using manipulative experiments employing Caulerpa transplantation/removal.

SUMMARY

The major conclusions of this study are:

1. Survival after five months ranged from 72% to 100%, with the highest rate occurring in the plot planted on 0.3 m centers.

2. The vegetative cover in the plot planted on 0.3 m centers had increased noticeably, while the cover in the plots planted on 0.6 and 1.0 m centers remained essentially unchanged.

3. Some units in the plots planted on 0.6 m and 1.0 m centers had been uprooted by shifting sediments and/or by ray foraging activities.

4. The transplanted Caulerpa (in Plot 3) is providing suitable habitat for a variety of epibenthic invertebrates and fishes.

5. The ease and apparent effectiveness of Caulerpa transplantation will aid investigation and research into aspects of its hypothesized ecological roles in submergent macrophyte communities.

ACKNOWLEDGEMENTS

This presentation is dedicated to the memory of Mr. Bill Ackerman, who generated the initial ideas for the project and assisted with site selection and preliminary scope of work preparation before his untimely death. The project was funded by MacDill Air Force Base and Mangrove Systems, Inc. The manuscript was prepared for publication by TEXT, a technical writing service.

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FLOODED FALLOW RICEFIELDS AND THE STRUCTURE OF BIRD COMMUNITIES

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ABSTRACT

The interaction of abiotic and biotic factors, their intensity, and the mechanisms by which these factors affect population dynamics are crucial data for understanding how ecological communities work. Tropical grasslands with standing water have a higher bird species diversity (BSD) than those without; habitat characteristics thought responsible for this effect were evaluated in four fallow ricefields. A three-year-old flooded fallow field had the highest BSD, while an unflooded three-year-old had the same BSD as a six-month-old flooded field. The other six-month unflooded field had the lowest BSD. Aquatic birds accounted for 76% and 83% of the diversity in the three-year and six-month-old flooded fields, respectively. In contrast, water birds accounted for 0% (three-year) and 27% (six-month) of the diversity in the two unflooded fields. There were significant linear relationships between the dependent variables (BSD) and on water depth, light intensity, plant height, and vegetation density, respectively. The strength of the relationships accrued in descending order. Although vegetation density was the best predictor of community BSD, aquatic bird diversity was predicted by percent of area flooded, and water depth. The prediction that the presence of water increases BSD was corroborated.

INTRODUCTION

Community ecologists focus on deciphering patterns that characterize natural assemblages of species, elucidating causal agents of these patterns, and demonstrating their generality in nature (Wiens 1983). Bird community structure is a reflection of habitat selection (Lack 1937; Hilden 1965) because most birds have specific requirements for feeding, courting, mating, reproduction, and other important survival activities. Many avian species do so with such specificity that their presence in a particular habitat makes them useful indicators of environmental change. The juxtaposition of various habitat types is dictated by varying combinations of overlapping gradients of abiotic factors. These factors include topography, soil mineral content, pH, soil type, and moisture regimes, which in turn affect the patchy

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distribution of vegetation. Vegetation, standing and running water, and topography produce the kind of habitat patchiness which is most apparent to us. They add dimension and variety to the landscape. These landscape components are the substrate on which terrestrial and aquatic birds act out their ecological roles.

Habitat structure has long been shown to be a major determinant of bird community structure because bird species diversity (BSD) tends to increase as a function of vegetation complexity (MacArthur & MacArthur 1961; Karr & Roth 1971; Wiens 1974; Willson 1974; Roth 1979; Wiens 1983). However, variability in BSD/habitat relationships cannot be explained by vegetation complexity alone (Roth 1977; Karr 1983; Osborne et al. 1983; Wiens 1983). I attempted to elucidate other factors influencing avian community structure in flooded and unflooded fallow ricefields in Guyana, South America, from January to March 1984. The bleak economic conditions gripping third world countries made it possible to conduct this comparative study because marginal agricultural lands were removed and continue to be removed from production. I was especially interested in testing Karr's (1968) prediction that the presence of water increases bird diversity. I also wanted to demonstrate that ecological and recreational values are enhanced by allowing succession to proceed on marginal, inundated agricultural lands.

STUDY SITES

Field work was conducted at four 2 ha locations on coastal Guyana at Turkeyen (6° 49'N, 58° 8"W) and neighboring Cumming's Lodge about 6.4 km east of Georgetown, the capital (Figure 1). These sites are on

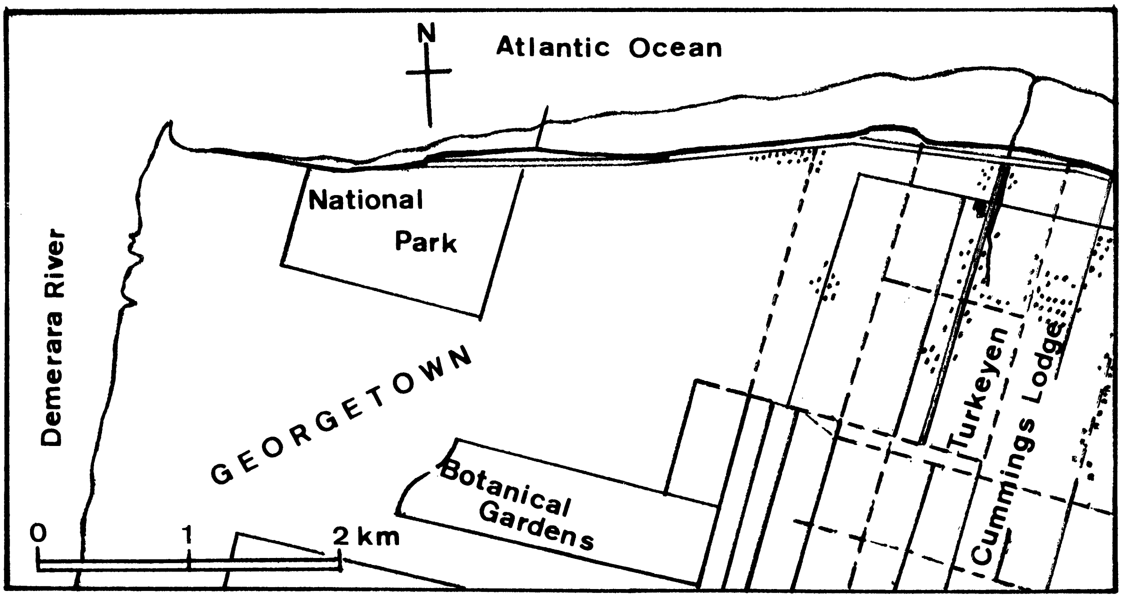

Figure 1. Northern Buyana showing the locations of Turkeyen and Cumming's Lodge. After the Directorate of Overseas Surveys, Fifth Edition, sheets (20/NE, 21/NW), 1965).

the eroded coastal flood plain with several drainage canals dissecting the landscape. These canals discharge water into the Atlantic Ocean to the north. Annual precipitation exceeds 2,250 mm (Giglioli 1959), but during my sampling period only 201 mm of rainfall fell during the three months. Mean yearly temperature is 27° C, while mean annual relative humidity is 82% (Giglioli 1959). Detailed descriptions of geology, flora and fauna of Guyana's coast are available in Harrison et al. (1913), Pain (1950), Snyder (1966), and Poonai (1970).

MATERIAL AND METHODS

Habitat Sampling

Two three-year, and two six-month-old 2 ha fallow fields were sampled to obtain quantitative estimates of five habitat variables. One of each field age class was flooded and one was unflooded. Each 2 ha sample plot was part of the interior of larger fields. This was done to minimize the influence of edge effect. A modification of the point quadrat random sampling method (Wiens 1968) was utilized along transects established at 50 m intervals. Twenty-five samples were taken in each field.

Three random numbers were used to locate the sample units (Wiens 1969). The first random number indicated how far (in meters) to travel along a transect. The second was used to determine the side of the transect to sample. Thus, when this number ended with an odd digit, I worked on the left of the transect, and when it ended in an even digit, I sampled the right side. The third random number dictated the distance to travel perpendicular to the transect to locate the sample unit (Wiens 1969).

After sample units were located, a 1 m^2 quadrat was lowered into place. A 2.5 m long metal rod 4.5 mm in diameter, marked at 100 mm intervals was placed vertically into the vegetation, 30 mm in from each corner of the quadrat. At each point, I recorded the presence or absence of each vegetation physiognomic type, i.e. (1) gramoid or narrow leaf herbaceous plants; (2) forb or broad leaf herbaceous plants; (3) woody vegetation; and (4) no vegetation present. The number of contacts of vegetation at each 100 mm height increment was recorded to quantify vegetation density (complexity) (Wiens 1969). Plant heights and water depths were also recorded at each point.

Vertical light penetration (footcandles) were recorded as an index of vegetation density. Sampling was done between 1030 and 1330 hours (Guyana Standard Time), to ensure some verticality in light penetration (Wiens 1969). Readings were made above the vegetation (open sky) and at 200 mm above ground level, or just above the water surface when the 100 mm marker was submerged. These data were converted to percent open-sky intensity.

Bird Census, Foraging Guild Construction and Diversity

In order to evaluate the influence of habitat structure on avian community composition, I conducted transect counts of all diurnal year-round resident birds utilizing the fields between sunrise and 0930 hours. Counts were made twice weekly from January to March 1984.

Each bird species was assigned to a foraging guild consisting of a three digit number (ABC). This is a modification of Willson's (1974) scheme to better represent tropical conditions as follows: A. PRIMARY FOOD HABITS: 1. frugivore, 2. granivore, 3. insectivore, 4. omnivore, 5. nectivore, 6. carnivore, 7. scavenger. B. FORAGING SUBSTRATE: 1. ground, 2. low plant elevation (0-1 m), 3. middle elevation (1-6 m), 4. high elevation (> 6 m), 5. bark, 6. flower, 7. termitorium, 8. water, 9. air. C. FORAGING BEHAVIOR: 1. ground peck, 2. foliage glean, 3. flower probe, 4. mud probe, 5. bark drill, 6. dabble, 7. sally, 8. dive, 9. strike.

The Shannon-Wiener function (Shannon & Weaver 1949) was calculated to estimate BSD (H'_S), and foraging guild diversity (H'_{FG}):

$$H'_S = - \sum_{i=1}^{S} p_i \log_n P_i$$

where H'_S is the amount of diversity in a group of S species of H'_{FG} is the diversity of a sample of foraging guild types; S is the number of species; p_i is the fraction of the whole sample composed of species i, n is the base of logarithm used (here it is the natural logarithm or base e).

A separate diversity index was calculated for all birds that used water as their foraging substrate (water birds), i.e., birds with eight as the middle digit of their foraging guild number (see Appendix). The diversity of these water birds (H'_{WB}) was compared to community BSD to evaluate the importance of water in structuring avian communities in fallow ricefields (see Osborne et al. 1983).

RESULTS

Habitat Structure

Both flooded fields were dominated by gramoid vegetation which comprised 69% cover in the three-year-old, and 78% in the six-month-old field. Forbs contributed 17% and 12%, while woody vegetation accounted for 6% and 2%, respectively. Eight percent of each field was unvegetated. Unflooded fields were also dominated by gramoids, which contributed 62% cover in the three-year-old and 78% in the six-month-old fields. Forbs provided 12% and 10%, woody plants covered 23% and 4%, and 3% and 8% was without vegetation in these two unflooded fields, respectively. Gramoid forms appeared to become less dominant in the three-year-old fields, being replaced by forbs and woody vegetation, especially in the unflooded field.

Table 1. Measurements (mean ± one standard deviation) of habitat characteristics for four fallow 2 ha ricefields on coastal Guyana, January-March 1984.

Parameter	Sites			
	3-YR F[1]	3-YR D[2]	6-MTH F[1]	6-MTH D[2]
Water Depth (mm)	200±173	5±12	109±70	4±10
Plant Height (mm)	760±378	530±286	309±210	206±278
Vegetation Density (contacts/100 mm)	6±2	4±3	3±3	1±2
Light Intensity (% open sky)	57±24	54±22	74±16	84±16
Percent Flooded	80±5	15±2	86±3	14±1

[1]Flooded
[2]Unflooded

Qualitatively, all of the sites looked different, and these differences were reflected in quantitative measurements (Table 1). Comparisons of mean water depth indicated significant differences among sites ($P<0.01$, Mann-Whitney U Test), except for the comparison between the three-year- and six-month-old unflooded fields [$P= 0.89$, Mann-Whitney U Test (Table 1)]. Comparisons of plant height at all sites (Table 1) showed significant differences ($P<0.01$, Mann-Whitney U Test). Light intensity (Table 1) comparisons indicated differences among the three-year-old flooded, and six-month-old flooded and unflooded fields, and three-year-old unflooded and six-month-old flooded and unflooded fields ($P\leq0.05$, Mann-Whitney U Test). There were no differences between the two three-year-old fields [$P= 0.18$, Mann-Whitney U Test (Table 1)]. Comparisons of vegetation density (Table 1) indicated no significant differences except for that between the three-year-old flooded and six-month-old dry fields ($P<0.05$, Mann-Whitney U Test). Finally, comparisons of percent area flooded (Table 1) indicated significant differences for the three-year- and six-month-old flooded fields contrasted with the two unflooded fields ($P<0.01$, Mann-Whitney U Test).

Bird Community Structure and Organization

The qualitative and quantitative habitat differences in the four fields (Table 1) were reflected in differences in bird community structure and organization (Table 2).

Table 2. Bird community characteristics of four fallow 2 ha ricefields on coastal Guyana, January-March 1984.

Parameter	Sites			
	3-YR F[1]	3-YR D[2]	6-MTH F[1]	6-MTH D[2]
Species Richness	33	27	20	11
Guilds	18	16	13	7
Individuals	273	180	180	159
H'_S	3.1	2.8	2.8	2.2
H'_{FG}	3.8	3.6	3.1	2.2
H'_{WB}	2.4	0	2.3	0.6

[1]Flooded
[2]Unflooded

There were significant statistical differences in avian community composition for all sites in terms of species richness and number of individuals [$P<0.05$, Mann-Whitney U Test (Table 2)], except for the comparison between three-year-old dry and six-month flooded fields. Also, there were no differences in the number of guilds (Table 2) for comparisons between three-year-old flooded and unflooded fields, three-year- and six-month-old flooded fields, and three-year-old dry and six-month-old flooded fields ($P>0.05$, Mann-Whitney U Test). However, significant differences existed between three-year-old flooded and six-month-old dry fields, three-year- and six-month-old dry fields, and between the two six-month-old fields ($P<0.05$, Mann-Whitney U Test).

All sites had relatively high BSDs and foraging guild diversities with both three-year-old fields having the highest indices (Table 2). It is notable that the six-month-old flooded field had the same BSD as the three-year-old unflooded field, but a lower foraging guild diversity index (Table 2). Birds that used water as their foraging substrate accounted for most of the diversity of the two flooded fields (Table 2). Water birds contributed 76% and 83% of the diversity for the three-year- and six-month-old flooded fields respectively, but contributed 0% and 27% to the community diversity of the three-year- and six-month-old unflooded fields (Table 2).

Analysis of the ability of habitat parameters to predict three bird species diversity indices indicated that water depth, light intensity, plant height, and especially vegetation density were good predictors of community BSD (Table 3). The best predictors of bird foraging guild diversity were plant height, vegetation density, and light intensity, while water depth and percent area flooded predicted water bird diversity (Table 3).

Table 3. Regressions of three bird species diversity indices on habitat characteristics on coastal Guyana, January-March 1984.

Diversity Index	on Habitat Parameters	r^2	y=ax+b
$H_S{}^1$	Water Depth	0.55*	y=0.003·x+2.47
	Plant Height	0.85*	y=0.001·x+2.07
	Vegetation Density	0.96*	y=0.177·x+2.08
	Light Intensity	0.76*	y=-0.023·x+4.23
	Percent Flooded	0.36	y=0.006·x+2.43
$H^1{}_{FG}$	Water Depth	0.33	y=0.004·x+2.83
	Plant Height	0.83*	y=0.003·x+2.00
	Vegetation Density	0.90*	y=0.327·x+2.03
	Light Intensity	0.90*	y=0.048·x+6.38
	Percent Flooded	0.19	y=0.008·x+2.80
$H^1{}_{WB}$	Water Depth	0.83*	y=0.012·x+0.40
	Plant Height	0.09	y=0.002·x+0.66
	Vegetation Density	0.20	y=0.258·x+0.42
	Light Intensity	0.0004	y=0.002·x+1.22
	Percent Flooded	0.94*	y=0.030·x+0.12

*Significant at P<0.05

DISCUSSION

In this study birds preferred the inundated fallow fields irrespective of the time elapsed since last cultivation. The three-year-old flooded field had 18% more species, 22% more guilds, and 34% more individuals than the three-year-old dry field. Similarly, the six-month-old flooded field was occupied by 45% more species, supported 46% more guilds, and 12% more individuals than its unflooded counterpart. Furthermore, the three-year-old unflooded field with its significantly greater vegetational complexity supported 26% more species and 7% more guilds, but exhibited no differences in the number of individuals or community BSD, compared to the six-month-old inundated field. Thus, Karr's (1968) prediction that the presence of water increases avian species diversity is supported.

Only one BSD data set is known for tropical wetlands. BSD in the three-year-old flooded field was 3.1 and is similar to the 3.2 recorded for a 1.3 ha two-year-old fallow ricefield censused by me during 1974 and reported in Osborne et al. (1983). However, the latter index included a southern migrant, the Forked-tailed Flycatcher (Muscivora tyrannus), while the present index consists of data on year-round residents. Although Wiens (1983) cautions about the pitfalls of comparing BSDs from different geographical localities, it is interesting to note that the unflooded three-year-old field had a similar BSD (2.6) as African grasslands (Karr 1976). The general pattern of biotic diversity being higher in the tropics holds since BSDs measured during this study exceed those (1.5-1.6) reported for temperate North (Wiens 1969) and South American (0.7-1.3) grasslands (Cody 1966). Tropical moist forest have the highest avian species diversity indices, but they are not much higher than the estimates for tropical fallow wetlands in this study. For example, Howell (1971) recorded a BSD of 3.6 in Nicaragua, Karr (1971) 3.7, and Karr and Roth (1971) 3.5 in Panama, and Lovejoy (1974) recorded 3.9 in Belem, Brazil.

There were qualitative differences in guild structure of the aquatic bird communities occupying the three-year- and six-month-old flooded fields. Open habitat shorebirds (Scolopacidae) occupied the six-month field and were not found in the three-year-old field. Both wet habitats shared several species of herons (Ardeidae), Snail Kites (Rostrhamus sociabilis) and Limpkins (Aramus guarauna) among others. Herons have food preferences that include fish, frogs and tadpoles, while kites and Limpkins specialize on apple snails (Pomacea dolioides). These food habits suggest that standing water enhances avian community diversity by providing access to additional aquatic food resources not available in the two unflooded habitats.

Other species found mostly in the three-year-old fields prefer to build their nests over water (Haverschmidt 1968). These species include the Pale-breasted Spinetail (Synallaxis albescens), Yellow-throated Spinetail (Certhiaxis cinnamomea), Pied Water-Tyrant (Fluvicola pica), and White-headed Marsh-Tyrant (Arundicola leucocephala). A total of 22 individuals of the aforementioned species occupied the three-year-old field, and only five Pied Water-Tyrants lived in the six-month-old wet field. These findings suggest that all of the above species except Pied Water-Tyrants require some vegetational complexity. It also seems reasonable to conclude that water may provide relatively safer nesting sites not available in similarly structured but dry habitats.

Wetlands provide considerable amounts of animal protein for human consumption, particularly in Guyana. I have observed hundreds of people harvesting birds, including herons, whistling ducks, Limpkins, Purple Gallinules, Wattled Jacanas, shore birds, and Red-breasted

Blackbirds, in addition to several fish species, apple snails, and fresh water shrimp from inundated fallow ricefields. This role is critical because increasing numbers of Guyanese do not have the economic resources to obtain protein from the marketplace. By allowing marginal agricultural lands on coastal Guyana to undergo ecological succession, human survival, ecological and recreational values can be enhanced.

In North America and elsewhere, modifications of the methods described in this study could prove useful for obtaining quick evaluations of the functioning of the relatively few remaining but imperiled wetlands.

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APPENDIX

Bird Species, their foraging guild numbers and sites on coastal Guyana, January-March 1984.

Species		Foraging Guild	Site			
Striated Heron	*Butorides striatus*	689	3-YR F[2]		6-MTH F[2]	
Little Blue Heron	*Florida caerula*	689	3-YR F			
Cattle Egret	*Bubulcus ibis*	329	3-YR F	6-MTH D[3]	6-MTH F	6-MTH D[3]
Great Egret	*Casmerodius albus*	689	3-YR F		6-MTH F	
Snowy Egret	*Egretta thula*	689	3-YR F		6-MTH F	
Tricolored Heron	*Hydranassa tricolor*	689	3-YR F		6-MTH F	
Bl.-cr. Night-Heron	*Nycticorax nycticorax*	689	3-YR F			
B.B. Whist. Duck	*D. autumnalis*	286	3-YR F		6-MTH F	6-MTH D
Black Vulture	*Coragyps atratus*	711		3-YR D		
Ylw.-hd. Vulture	*Cathartes Burrovianus*	711		3-YR D		
Snail Kite	*Rostrhamus sociabilis*	688	3-YR F		6-MTH F	
White-Tailed Hawk	*Buteo albicaudatus*	618			6-MTH F	
Gray Hawk	*Buteo nitidus*	618		3-YR D		
Savana Hawk	*H. meridionalis*	618		3-YR D		
Long-wing. Harrier	*Circus buffoni*	618	3-YR F			
Ylw.-hd. Caracara	*Milvago chimachima*	618		3-YR D		
Crested Caracara	*Polyborus plancus*	711	3-YR F	3-YR D	6-MTH F	6-MTH D
Limpkin	*Aramus guarauna*	484	3-YR F		6-MTH F	
Purple Gallinule	*Porphyrula martinica*	482	3-YR F			
Wattle Jacana	*Jacana jacana*	482	3-YR F	3-YR D	6-MTH F	6-MTH D
Southern Lapwing	*Vanellus chilensis*	481			6-MTH F	
Solitary Sandpiper	*Tringa solitaria*	484			6-MTH F	
Greater Yellowlegs	*Tringa melandleuca*	484			6-MTH F	
Spotted Sandpiper	*Actitis macularia*	481			6-MTH F	
Common Stilt	*H. himantopus*	384	3-HR F		6-MTH F	

Eared Dove	*Zenaida auriculata*	211				6-MTH D
Ruddy Grnd.-Dove	*Columbina talpacoti*	211				6-MTH D
Br.-thr. Parakeet	*Aratinga pertinax*	132		3-YR D		
Smooth-Bil.Ani	*crotophage ani*	312	3-YR F	3-YR D		
Striped Cuckoo	*Tapera naevia*	322	3-YR F	3-YR D		
Bl.-throated Mango	*A. nigricollis*	523		3-YR D		
Wh.-tail. Gldthroat	*Polytmus guainumbi*	523	3-YR F	3-YR D		
Pale-br. Spinetail	*Synallaxis albescens*	322	3-YR F	3-YR D		
Ylw.-thr. Spinetail	*Certhaixis cinnamomea*	312	3-YR F	3-YR D		
Pied Water-Tyrant	*Fluvicola pica*	322	3-YR F	3-YR D	6-MTH F	
Wh.-hd. Marsh-Tyr.	*A. leucocephala*	382	3-YR F			
Tropical Kingbird	*T. melancholicus*	437	3-YR F	3-YR D		
Rust-mgd. Flcatcher	*Myiozetetes cayanensis*	322	3-YR F	3-YR D		
Great Kiskadee	*Pitangus sulphuratus*	432	3-YR F	3-YR D		
Cm. Tody-Flcatcher	*Todirostrum cinereum*	432	3-YR F	3-YR D		
Ylw.-bellied Elaenia	*Elaenia Flavogaster*	337	3-YR F	3-YR D		
Shiny Cowbird	*Molothrus bonariensis*	411	3-YR F			
Carib Grackle	*Quiscalus lugubris*	411	3-YR F	3-YR D	6-MTH F	6-MTH D
Ylw.-hooded Blbird	*Agelaius icterocephalus*	411	3-YR F			
Yellow Oriole	*Icterus nigrogularis*	432		3-YR D		
Red-br. Blackbird	*Leistes militaris*	411	3-YR F	3-YR D		6-MTH D
Red-capped Cardinal	*Paroaria gularis*	222		3-YR D		
Blue-bl. Grassquit	*Volatina jacarina*	222	3-YR F	3-YR D	6-MTH F	6-MTH D
Variable Seedeater	*Sporphila americana*	222				6-MTH D
Ruddy-br. Seedeater	*S. minuta*	222	3-YR F	3-YR D	6-MTH F	6-MTH D

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[1]English and scientific names taken from Meyer de Schauensee (1966)
[2]Flooded
[3]Dry

CONDITION AND MANAGEMENT OF TAMPA BAY TIDAL TRIBUTARIES

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ABSTRACT

The status of 34 tidal creeks or minor tributaries in the Tampa Bay basin has been reviewed and the tidal creeks have been classified into their stressed, restorable or natural condition. For three of the tidal creeks, a series of public workshops were held to develop management and/or restoration plans. The plans were then applied to other tributaries as a test for consistency. The end product provides a guideline for protective management and suggested habitat and tributary restoration techniques.

INTRODUCTION

Many of the tidal tributaries entering Tampa Bay have been filled, diverted, hardened, channelized, or otherwise modified by point and non-point source discharges. This habitat loss has resulted in declining populations of commercially valuable fish and shellfish, including a complete collapse of such fisheries as those for scallops and oysters, and major declines for bait shrimp, red drum, and spotted sea trout. In addition, the provision of adequate quantities of freshwater to Tampa Bay is critical to its function as a productive estuary. The freshwater must be provided at ecologically relevant times, and be relatively free of contaminants. At present, every river and many minor tributaries flowing to Tampa Bay are either dammed, tapped for cooling water, or have modified drainage patterns. Development pressures and demands for potable water are immense and increasing, meaning that the basic estuarine character of Tampa Bay is endangered.

Minor tributaries, or tidal creeks, flowing into Tampa Bay vary greatly in condition. Historical and anecdotal evidence exist to show that these streams were immensely productive estuarine zones. Modern data on relatively pristine tidal creeks support this view. Although little is known regarding the ecological condition of the majority of the minor tributaries entering Tampa Bay, the following conclusions are relevant to the study and management of these systems:

- Tidal creeks provide critically important habitat for the majority of economically important species of fish found in the Gulf coastal waters.

- A comprehensive study or summary statement has never been accomplished for the condition of rivers and creeks flowing into Tampa Bay, or of their individual management problems.

- The various tributaries of Tampa Bay are naturally and culturally different, and each has unique problems as well as problems common to other streams.

- Eventual management of each tidal creek as an ecological unit will have to involve several levels of government and authority.

- Although several streams among those considered are highly stressed, more are natural or are still restorable.

- Population growth threatens all bay tributaries and unless actions are taken before 1990 more streams will be irrevocably stressed by the year 2000.

The information provided in this report has been condensed from Ecological Assessment, Classification and Management of Tampa Bay Tidal Creeks (TBRPC, 1986a) to include only the minor tidal tributaries distributing into Tampa Bay. The report prepared by TBRPC (1986a) is an outgrowth of, and is consistent with The Future of Tampa Bay (TBRPC, 1984), a comprehensive management plan for Tampa Bay, and the recently completed Habitat Restoration Study for the Tampa Bay Region (TBRPC, 1986b).

STUDY SITE

The study area included 34 minor tidal tributaries flowing into Tampa Bay. Figure 1 identifies the classified minor tidal creeks and relative location within the Tampa Bay Region. Analysis of each tributary included a brief survey of conditions within the watershed and creek alignment, with special emphasis on the tidally-influenced portion of the tributary.

METHODS AND MATERIALS

Rivers and creeks flowing into Tampa Bay vary greatly in condition. While basic information on tidal tributaries is lacking, enough exists to allow the creeks to be classified by their overall condition from a management point of view. Tidal creeks were classified into natural, restorable and stressed condition depending upon the extent of alterations within the tributary. "Creeks" were defined as small streams of the Pamlico Terrace in which tidal prisms are equal to or larger than average discharge. All classifications were based on conditions within the tidal segment of each stream. For the purposes of this paper, the extent of tidal influences was determined by the transition of brackish to freshwater vegetation communities. Land use adjacent to each creek was identified to further characterize conditions and potential impacts.

Pinellas County
1) Salt Creek
2) Booker Creek
3) Tinney Creek
4) Grassy Creek
5) Long Branch Creek
6) Allen Creek
7) Alligator Creek
8) Mullet Creek
9) Bishop Creek
10) Moccasin Creek

Hillsborough County
11) Double Branch Creek
12) Channel A
13) Rocky Creek
14) Brushy Creek
15) Dick Creek
16) Woods Creek
17) Peppermound Creek
18) Sweetwater Creek
19) Fish Creek
20) Coon Hammock Creek
21) Broad Creek
22) Delaney Creek
23) Archie Creek
24) Bullfrog Creek
25) Little Bullfrog Creek
26) Newman Branch
27) Wolf Branch
28) Cockroach Creek
29) Piney Point Creek

Manatee County
30) Little Redfish Creek
31) Frog Creek
32) Cabbage Slough
33) McMullen Creek
34) Wares Creek

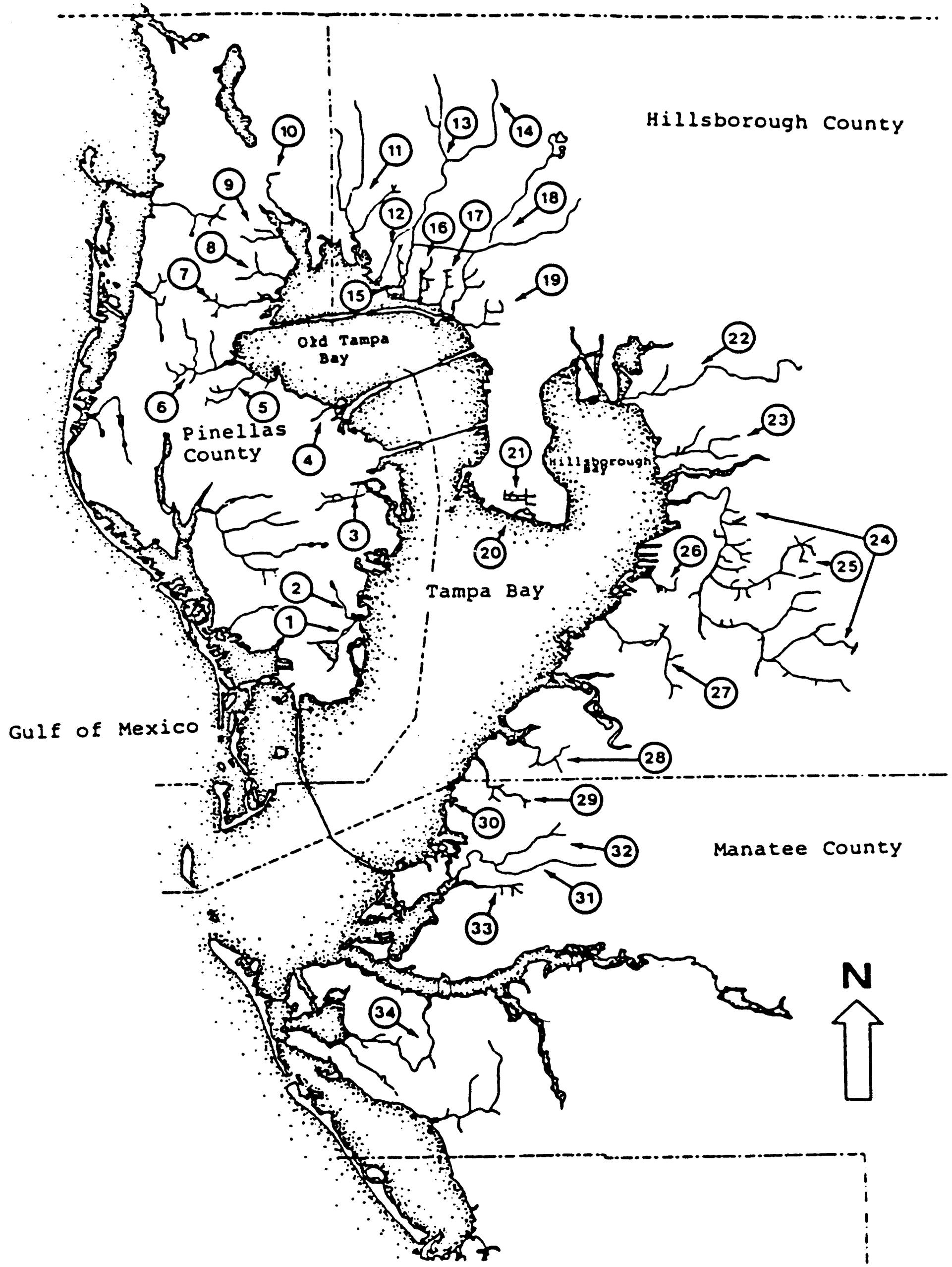

Figure 1. Classified tidal creeks and relative location in the Tampa Bay region.

Linear creek length was calculated using a Charvoz planimeter averaging three replicate measurements. Creek characterization was based upon historical literature, aerial photography and 1:24,000 USGS quadrangles.

A series of three public workshops were held to develop a management/restoration plan for three selected minor tributaries. The information collected from the workshops was then applied to each of the tributary classifications (natural, stressed and restorable condition) as a test for consistency.

RESULTS

Table 1 identifies the length, land use and condition of each tidal tributary reviewed in this report. Table 2 represents the breakdown of classified creek conditions for each of the three counties surrounding Tampa Bay.

One half of the minor tidal tributaries surveyed in Pinellas County are classified in stressed condition. This observation is primarily due to the intensive development that has occurred within the county. There are two and three restorable and natural tidal tributaries, respectively, in Pinellas County which discharge into Tampa Bay.

Hillsborough County contains a total of nineteen minor tidal tributaries. The majority of the tributaries remain in restorable (eight) or natural (three) condition. Extensive agricultural areas remain in the county, and urban expansion is expected to continue.

One-half of the surveyed tidal creeks in Manatee County are classified as natural in condition. One creek system is considered stressed with one in restorable condition. Within the county the urban areas are located in the western half with agricultural and mining areas in the eastern portion.

Due to the variety of governing organizations responsible for tidal creek and watershed management, it is necessary to acquire input from as many viewpoints as possible. A public workshop to develop management/restoration plans for each selected tidal tributary was held to facilitate local involvement. Local governments, environmental organizations and concerned citizens were invited to attend the public workshops. Results were tabulated and organized into general policies to support the management objectives. The framework was then reviewed by the Natural Resource Committee of the Tampa Bay Regional Planning Council Agency on Bay Management.

The framework is used as a general plan for all tidal tributaries and is illustrated in Table 3. The application for the recommended plan is expected to vary significantly depending upon the tidal creek condition and existing authority involvement.

Table 1. Tidal creek summary.

Creek/County	Approx. Length Kilometers (Miles)		Land Use	Condition
Pinellas County				
1) Salt Creek	2.4	(1.5)	Res.	Stressed
2) Booker Creek	3.1	(1.9)	Comm./Ind./Res.	Stressed
3) Tinney Creek	4.3	(2.7)	Res./Comm.	Stressed
4) Grassy Creek	1.3	(0.8)	Open Space/Comm.	Natural
5) Long Branch Creek	5.5	(3.4)	Res./Comm.	Restorable
6) Allen Creek	9.7	(6.0)	Res./Comm.	Stressed
7) Alligator Creek	7.1	(4.4)	Res./Agr.	Stressed
8) Mullet Creek	3.7	(2.3)	Res./Comm.	Restorable
9) Bishop Creek	2.9	(1.8)	Res.	Natural
10) Moccasin Creek	2.4	(1.5)	Res./Agr.	Natural
Hillsborough County				
11) Double Branch Creek	10.9	(6.8)	Agr.	Natural
12) Channel A	6.6	(4.1)	Res./Agr.	Man-made
13) Rocky Creek	17.5	(10.9)	Res./Agr.	Stressed
14) Brushy Creek	10.3	(6.4)	Res./Agr.	Non-tidal
15) Dick Creek	2.6	(1.6)	Res./Open	Restorable
16) Woods Creek	2.6	(1.6)	Res./Ind.	Stressed
17) Peppermound Creek	2.6	(1.6)	Res.	Restorable
18) Sweetwater Creek	16.7	(10.4)	Res./Agr.	Stressed
19) Fish Creek	3.7	(2.3)	Comm.	Restorable
20) Coon Hammock Creek	0.8	(0.5)	Open Space	Restorable
21) Broad Creek	4.0	(2.5)	Comm.	Restorable
22) Delaney Creek	17.4	(10.8)	Ind./Agr.	Stressed
23) Archie Creek	7.9	(4.9)	Ind./Res.	Restorable
24) Bullfrog Creek	28.2	(17.5)	Agr.	Restorable
25) Little Bullfrog Creek	9.7	(6.0)	Agr.	Non-tidal
26) Newman Branch	4.0	(2.5)	Ind./Res.	Stressed
27) Wolf Branch	10.5	(6.5)	Agr.	Restorable
28) Cockroach Creek	4.0	(2.5)	Agr./Res.	Natural
29) Piney Point Creek	4.3	(2.7)	Agr./Ind.	Natural
Manatee County				
30) Little Redfish Creek	0.8	(0.5)	Open/Ind.	Restorable
31) Frog Creek	18.5	(11.5)	Open/Agr.	Natural
32) Cabbage Slough	6.3	(3.9)	Agr.	Non-tidal
33) McMullen Creek	6.1	(3.8)	Res./Agr.	Natural
34) Wares Creek	15.1	(9.4)	Res./Comm./Inc.	Stressed

NOTE: Res. = Residential
Comm. = Commercial
Ind. = Industrial
Agr. = Agricultural

Table 2. County classification summary.

County	Stressed	Restorable	Natural	Other	Total
Pinellas	5	2	3	0	10
Hillsborough	5	8	3	3	19
Manatee	1	1	2	1	5
TOTAL	11	11	8	4	34

DISCUSSION

In general, the majority of the tidal tributaries surveyed remain in natural or restorable condition. The tidal tributaries to the Tampa Bay estuary add to the quality of life the residents of the Tampa Bay Region have grown to appreciate. With the population growth expected to continue within the area, it is essential that management considerations protect the value of tidal creek systems to promote the quality of life in our region.

Natural minor tidal tributaries to Tampa Bay have undergone little or no structural modification and receive relatively small volumes of stormwater runoff or point source discharges. Frog Creek, in Manatee County, retains a naturally meandering alignment through an extensive mangrove forest into Terra Ceia Bay. Double Branch Creek is located in Hillsborough County and additionally has preserved the tidally influenced vegetational communities and channel orientation to Upper (Old) Tampa Bay. Both of these tributaries are representative of a natural tidal creek.

Restorable tidal tributaries provide the opportunity for improvement (i.e., habitat function, water quality). As an example, Bullfrog Creek has moderate habitat loss through piecemeal development, receives sanitary wastes and is used for stormwater drainage. Fish Creek is an extensive drainage system around Tampa International Airport and the highway interchange of State Road 60, Eisenhower Boulevard and airport access roads.

Stressed tributaries are characterized by intensive development and modifications that prevent restoration. Wares, Allen, Salt and Booker Creeks are highly urbanized and affected by stormwater runoff. Sweetwater and Rocky Creeks are controlled and located in rapidly urbanizing basins. Alligator Creek has lost the tidal connection to Tampa Bay by the construction of an elevated weir.

Table 3. Framework for management/restoration of Tampa Bay tidal creeks.

Objective: Maintenance/Restoration of Natural Function

CONSIDERATION: Water Quality and Quantity

Policy: Water Quality Improvement through control of non-point source pollutant loadings
a. Identify problem areas
b. Prioritize improvements
c. Coordination of agencies for improvements

Policy: Minimize point-source pollutants
a. Develop ecological criteria for all discharges
b. Promote water recycling
c. Promote effluent disposal alternatives for problematic septic tank and package plant systems

Policy: Protect natural freshwater inputs
a. Groundwater
b. Surface water

Policy: Develop consistent tidal creek monitoring and enforcement program
a. Water quality
b. Habitat and species utilization

CONSIDERATION: Habitat Utilization

Policy: Protect or improve natural channel alignment and elevation requirements for maintenance of productivity

Policy: Preserve natural vegetation and fish and wildlife resources
a. Removal of exotic species
b. Encourage wetland creation
c. Restore impacted areas

Policy: Protection of archaeological sites
a. Identification of sites in all areas before development
b. Preservation or excavation prior to destruction

Objective: Develop consistent and compatible land use standards

Policy: Promote public land acquisition and conservation easements for environmentally sensitive lands

Policy: Encourage compatible low density development on adjacent upland areas
a. Minimize development within the 25 year floodplain

Table 3. Framework for management/restoration of Tampa Bay tidal creeks (cont'd).

Policy:	Encourage clustering of water oriented land uses
Objective:	Management of tidal creeks as an important public asset
Policy:	Promote public education a. Value of tidal tributaries b. Prevent public degradation c. Minimize user conflicts
Policy:	Promote compatible public access

Management of Tampa Bay Tidal Creeks involves consideration of the following broad objectives:

- Maintenance/Restoration of natural function
  - water quality and quantity
  - habitat utilization
- Develop consistent and compatible land use standards
- Management of tidal creeks as an important public asset

Using the framework identified in Table 3, general guidelines for each tributary condition (natural, restorable and stressed) can be accomplished.

Objective: Maintenance/Restoration of Natural Function

Policy: Water Quality Improvements through Control of Non-Point Source Pollutants

Urban and agricultural stormwater runoff have been identified as the major sources of water pollution in Tampa Bay, with the former apparently predominating (TBRPC, 1978 & 1984). All tidal tributaries draining into Tampa Bay are affected to varying degrees by non-point source pollutants.

Reductions in the stormwater pollutant loadings to Tampa Bay can occur through stormwater legislation, such as House Bill 242 (1985). Specific recommendations for future legislation must include:

- the establishment of priorities and time frames for all developed areas, and
- the inclusion of agricultural areas in legislation and the permitting process.

Non-point source pollution loadings have impacted the Tampa Bay estuary by historic development practices, wetland draining, tributary channelization, impervious surfaces, etc. Many sources will require retrofitting to improve water quality conditions. Stormwater pollution abatement will benefit all tidal tributaries in the Tampa Bay Region.

Policy: Minimize Point-Source Pollutants

Stressed tidal tributaries to Tampa Bay are often affected by industrial and municipal discharges to the creek systems, examples include Allen, Rocky, Delaney, and Wares Creeks. Management considerations for stressed tributaries shall be oriented toward minimizing water quality impacts to the downstream systems. Recommendations include:

- develop ecological criteria for all discharges

- promote effluent disposal alternatives

- promote water recycling.

Restorable tidal tributaries offer the potential for improvement. All measures should be taken to improve or eliminate point source discharge quantities. Further protective measures could include prevention of any new surface water discharge within restorable creek watershed.

Natural tributaries within the Tampa Bay Region receive point source discharges while retaining the ecological character of the natural system (examples include Piney Point & Frog Creeks). Further degradation of natural conditions must be prevented. Effluent discharge alternatives for point source discharge to natural creek systems are recommended to be implemented. All new surface water discharge to natural tidal tributaries should be prohibited.

Policy: Protect Natural Freshwater Inputs

Many tributaries to Tampa Bay are in a stressed condition due to disruption of natural freshwater flows. Alligator Creek has lost the natural connection to Old Tampa Bay by the installation of an elevated weir. In addition, Tinney Creek has been bypassed using a large open drainage ditch to Tampa Bay. Alteration of freshwater flow down the tidal tributary can eliminate the creek's estuarine system (Alligator Creek) or disrupt the natural movement of the saltwater/freshwater interface and associated environmental systems.

Maintenance or restoration of natural freshwater inputs are vital to estuarine systems. Stressed systems should be evaluated with respect to the importance of limiting freshwater (water supply, residential lake, etc.) as opposed to the value of downstream ecosystems. Restoration of flows is recommended where practical and beneficial results can be identified.

Restorable creek systems can be improved through regulation of freshwater flows. Areas containing large quantities of impervious surfaces will benefit from stormwater retrofitting. Dammed or rerouted systems can be designed to follow natural drainage features and acquire typical runoff volumes. Channel "A," for example, has circumvented freshwater flows down Rocky Creek and isolated adjacent wetland systems. Natural freshwater sheetflow through tidal marsh systems can be restored by lowering portions or all of the berm along Channel "A," to allow freshwater/tidal inundation.

Tampa Bay tributaries in natural condition should retain freshwater inputs though preservation. Disruption of freshwater flows can potentially degrade the natural ecosystems, and protective measures should be taken to:

- prevent large surface water withdrawals
- maintain natural base flow quantities, and
- prevent salinity barriers, dams or other flow impediments.

Policy: Develop Consistent Tidal Creek Monitoring Program

The value of tidal tributaries to estuarine systems is readily apparent but often overlooked. Historic research activities have focused upon larger rivers and tributaries. Little consistent information has been accumulated for the conditions within the smaller tributaries feeding the Tampa Bay estuary. Tidal creek monitoring programs should include water quality and biological analysis.

Tidal creek monitoring programs are required for stressed tributaries to prevent further degradation to the creek and bay systems. Programs developed for restorable tributaries can monitor and identify improvements to the system that can then be applied to other tributaries. Monitoring and enforcement programs for natural systems can prevent alterations and provide baseline information for creek management objectives.

Policy: Protect or Improve Natural Channel Alignment and Elevation Requirements for Maintenance of Productivity

The environmental assessment provided in TBRPC (1986a) identified that stressed tidal tributaries to Tampa Bay continue to provide habitat for fish and wildlife usage. Maintenance of existing pockets of natural communities and improvement where possible will continue to maintain and/or increase the potential for wildlife to utilize stressed tributaries.

Restorable tributaries provide the greatest potential for improvement through channel configuration and elevation alterations. Fish Creek has been channelized in an extensive drainage system around Tampa International Airport. The lower segment of Broad Creek retains a natural tidal marsh system while the middle and upper segments have been channelized for drainage from MacDill Air Force Base. Both tidal creek systems can be improved by realignment or lowering of the berms for additional wetland acreage creation while maintaining drainage for the airports.

Bullfrog Creek currently has moderate habitat loss through piecemeal development. The Future of Tampa Bay (TBRPC, 1984) recommended that Hillsborough County amend its comprehensive plan to tighten control of shoreline uses and establish incentives for private landowners to restore the shoreline.

Little Redfish Creek has been impacted by illegal filling activities during the development of Port Manatee. Currently FDER is applying monies from the Pollution Recovery Trust Fund for restoration in the area. One course of restoration under consideration is removing silt from the creek bottom and reestablishing a tidal connection with adjacent isolated ponds. The program has the potential to restore habitat available for fish and wildlife uses.

Natural tributaries retain the requirements for habitat environs. Often, small areas exist for restoration within the creek system. However, the focus of attention within natural systems is oriented toward preservation.

Policy: Preserve Natural Vegetation and Fish and Wildlife Resources

Stressed creek systems normally retain isolated areas of natural vegetation utilized by local fish and wildlife populations. If productivity is to be maintained in stressed tributaries, it is imperative to protect the natural areas from continued developmental encroachment.

Restorable tributaries can be improved to provide conditions that are advantageous to fish and wildlife usage. The addition of natural vegetation and habitat will help to buffer cultural shocks to the estuarine system. Local and regional programs are necessary to restore the impacted areas and create additional habitat.

The natural ecosystems within tidal tributaries should be protected to provide natural habitat for fish and wildlife. In addition, wildlife corridors are recommended to tie natural habitats together for a more effective and diverse system. The proximity of Cockroach and Piney Point Creeks, two tributaries classified as natural, to each other allows wildlife populations to intermix and form a more productive ecosystem. Protection of marsh and open green space is necessary to maintain a wildlife corridor between the tidal tributaries.

Policy: Protection of Archaeological Sites

The provision for protection of archaeological sites is applicable to all tributaries and is independent of current creek condition. Archaeological surveys are currently required before development. Identified sites are evaluated against the State of Florida or federal criteria for significance to determine eligibility for listing in the National Register of Historic Places. Sites meeting the criteria must either be preserved or excavated prior to destruction. Additional recommendations include a survey of wetlands prior to development.

Objective: Develop Consistent and Compatible Land Use Standards

Policy: Promote Public Land Acquisition and Conservation Easements for Environmentally Sensitive Lands

Tributaries in stressed condition around Tampa Bay are often encroached upon by adjacent development (e.g., Allen Creek). Public acquisition of available sensitive lands can be accomplished to:

- preserve the remaining natural system
- promote habitat creation, and
- increase public utilization for recreation.

Creek systems currently classified as restorable may require the transfer of ownership to the public to allow restoration. Areas along Archie Creek are currently within private ownership (Gardinier, Inc.). The restoration of channel alignment and bank configuration can improve conditions within the creek system. Acquisition of adjacent areas into public ownership can facilitate restoration efforts and prevent further encroachment.

Public land acquisition and implementation of conservation easements can protect environmentally sensitive systems within natural tidal tributaries. Undeveloped areas can be set aside for future generations of people and wildlife to utilize. Buffer easements established before development can provide public access, prevent developmental encroachments, and moderate the impacts of a rise in sea-level.

The purchase of Terra Ceia Isles by the CARL Program can prevent unsuitable development in an environmentally sensitive area between Frog Creek, Terra Ceia Bay and Bishops Harbor. In addition, the acquisition of upland areas between Cockroach and Piney Point Creeks can:

- maintain a wildlife corridor
- preserve the uplands between two natural tributaries

- provide area for passive recreation

- allow reclamation of agricultural dikes, and

- maintain natural zonation of wetlands during sea level rise.

Policy: Encourage Compatible Low Density Development in Adjacent Upland Areas

The stressed creeks of Tampa Bay have historical encroachment that may limit future management of adjacent upland areas. Creeks impacted by water quality degradation should be considered for setbacks or buffer zones to allow wetland creation that will help buffer impacts to the estuarine system. New development on stressed tributaries should be limited or prohibited within the 25-year floodplain.

Restorable tributaries should be prohibited from development within the 25-year floodplain to accomplish necessary improvements to the creek. In addition, low density development adjacent to the creek will prevent encroachment into the tributary after restoration activities have been completed.

Natural systems must be preserved or protected from intensive development. Only eight natural tributaries are identified in the three county region. Protection of the remaining unique systems through low intensity zoning or preservation is required.

Policy: Encourage Clustering of Water-Oriented Land Uses

Clustering of water dependent land uses within stressed creek systems is often after-the-fact planning. For restorable and stressed tributaries of Tampa Bay, new development should utilize existing alterations during design. Examples include:

- marina sitting is encouraged along existing channels with good circulation and sufficient natural depth,

- overhead crossings (roads, infrastructure, etc.) should be clustered or follow existing routes, and

- industrial development utilizing surface waters must prevent environmental degradation and long term impacts.

Natural systems can promote development of more stringent preventative management measures and include:

- no new development in environmentally sensitive areas

- clustered overhead crossings, and

- infrastructure located under the creek to provide long-term aesthetic qualities.

Objective: Management of Tidal Creeks as an Important Public Asset

Policy: Promote Public Education

The focus of education for the general public should include:

- the intrinsic value of tidal tributaries

- prevention of public degradation, and

- minimization of user conflicts.

Due to the developmental pressures occurring within stressed creek systems, all three recommendations apply. Generally, education will help prevent unnecessary impacts to downstream systems.

Restorable tributaries differ by providing increased awareness of ways man can improve conditions within tributaries and their affect on the Tampa Bay estuary. Restoration can improve the quality of life by:

- improving water quality for water contact sports, fish and shellfish harvesting and scenic aesthetics, and

- additional wetland creation potentially can provide:
    - utilization by fish and wildlife
    - buffering of water quality impacts
    - prevention of erosion, and
    - scenic amenity.

Natural tidal tributaries can be assessed for identification of unaltered conditions. Baseline information and education must have a control for comparison. Creek systems in natural condition will provide the model for restoration of impacted systems.

Policy: Promote Compatible Public Access

Public access is necessary for all conditions of tidal creeks but is limited by proximity to urban areas and available resources. Stressed tributaries often have the greatest access available, due to their close proximity to urban areas. However, the stressed creeks are adversely affected by the increase in usage and continued public degradation. Management of restorable and natural tidal tributaries can control type and volume of public usage within the watershed. Low intensity access should be provided to restorable tributaries for education of the public toward restoration and the benefits derived from improved conditions.

Passive recreation is also recommended in natural systems for people to identify with the high productivity that pristine environments provide. The natural system provides the highest quality of aesthetic resources available.

CONCLUSION

Developing general management/restoration recommendations for tidal creek ecosystems is difficult due to the great diversity of the tributary systems involved; their particular condition and management needs; and the regulatory, economic and other facets of each problem. Emphasis should be placed on the restorable tributaries since restoration can potentially prevent them from becoming a stressed system. Second, priority should then be given to protection of the natural tributary, followed by preventing additional impacts to the estuary from stressed tidal tributaries.

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IMPLICATIONS OF SEA LEVEL RISE FOR WETLANDS CREATION AND MANAGEMENT IN FLORIDA

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ABSTRACT

A 60 to 300 centimeter rise in sea level is forecast for the next century due to global warming caused by elevated CO_2 levels from increased energy consumption. Recent sea level has risen at a much slower rate, and wetland effects of the relatively gradual rise can be found along open coastlines, bay shores, and tidal rivers. The impact of a higher sea level depends on the rate of change and its final stand; geomorphic considerations; and biotic responses. Key geomorphic factors are sediment source, subsidence rate and upland slope. Key biotic responses are vertical accretion, dispersal, and salt tolerance. Tidal freshwater wetlands are particularly vulnerable to projected rates of sea level rise because of salt intolerance and limited substratum availability. Overall, wetlands in proximity to developed uplands may be seriously threatened if rates of sea level rise exceed mid-range estimates (about 180 cm per 100 years). Wetland managers need to decide on meaningful time horizons for preservation, conservation and restoration. Consideration of time horizons and various sea level rise scenarios should be a part of routine wetland management practice, especially creation, restoration, and mitigation. Aspects of wetland creation affected by sea level rise include species selection, site selection and preparation, monitoring, and design of compatible upland activities. Dedication of low-lying upland as permanent wetland easements could be an extremely meaningful form of mitigation if mid-range to high rates of sea level rise can be forecast or measured with greater certainty.

INTRODUCTION

The average and extreme levels of the sea affect marine, intertidal and riverine geology, chemistry and biology, as well as a number of economic factors. Sea levels have been studied throughout history and recently by scientists and engineers. An aspect of sea level central to all of these interests has been the rate of sea level rise or fall. This paper reviews data on sea level, the record and effects of sea level rise in Florida, and implications of projected rates for wetland management and science.

Virtually every aspect of Florida's biogeochemistry is the result of past sea level stands. The present distribution and abundance of tidal marshes and mangroves in the state are the consequence of low tidal range, a subtropical climate, and especially the fact that sea level has been rising at a low rate, relative to earlier times. About 7000 years ago sea level was 4.0 meters below modern sea level; 4000 years ago it was approximately 20 meters lower than present; and for the last 2000 years the sea has risen by about 1.0 meter (Scholl, Craighead, & Stuiver 1963).

Oceanographic measurements made during the past century reveal that sea level continues to rise along most shores. During 1940-1975 the average rate of sea level rise along the coast of the United States was 1.5 mm/yr (Hicks 1978), which is about half the rate measured during a 30-50 year period before 1940 when global temperatures were higher (Donn & Shaw 1967). For the period 1898-1980, sea level along the Florida coast has risen between 1.7-2.4 mm/yr (Hicks, Debaugh, & Hickman 1983), as shown in Table 1.

Table 1. Trends and variability of sea level in Florida, adapted from Hicks, Debaugh, and Hickman (1983).

	Record		Value, mm/yr	
Station	Began	Missing	Trend	± Std. Error
Fernandina Beach	1898	1924-38	1.7	0.4
Mayport	1929	0	2.3	0.3
Miami Beach	1932	1979	2.3	0.2
Key West	1913	0	2.2	0.2
Cedar Key	1915	1926-38	2.0	0.2
Pensacola	1924	0	2.4	0.3

FLORIDA'S TIDAL ENVIRONMENTS

The effect of tides occurs along the entire Florida coastline and includes a larger area than is affected by saltwater. Truly marine areas affected by the tide are those nearshore parts of the Gulf of Mexico and Atlantic Ocean where tidal currents are modified by bathymetry, and where wave climates, water chemistry, or other factors vary on a tidal basis. Sandy beaches are the most common boundary between marine areas and uplands.

Salt marshes and mangrove forests are exposed directly to the Gulf of Mexico along much of the west coast and usually adjoin waters of estuarine salinity. Submerged aquatic vegetation grows in shallow waters seaward of the marsh or mangrove coast, which also supports subtidal and intertidal oyster reefs, algae beds, or unconsolidated sediments. Coastal wetlands of the Big Bend area are primarily black needlerush (*Juncus roemerianus*) marshes, whereas the Everglades wetlands open to the sea are comprised mostly of red mangrove (*Rhizophora mangle*) or black mangrove (*Avicennia germinans*).

The upland side of coastal marshes and mangroves is vegetated either by freshwater grass marshes or by meadows of succulent halophytes. Large areas behind the Everglades mangrove zone are sawgrass (*Cladium jamaicensis*), bulrush (*Scirpus* spp.) or cattail marsh, usually with extensive intermixing by herbaceous freshwater vegetation. Where fringing forests or marshes are backed more closely by uplands, but separated by level ground affected by tides, large areas of cordgrass (*Spartina* spp.) or meadows of glasswort (*Salicornia virginica*), saltwort (*Batis martima*), sea purslane (*Sesuvium portulacastrum*) and other halophytic ground cover may grow. Salterns develop in the same areas when local conditions allow spring tides to flood poorly drained areas and salt accumulates by evaporation. Marshes and mangrove forests may also be separated from uplands by scarps of low relief, or there may be a rapid transition to upland plant communities where relief is even greater.

Tides (and salinity) affect the mouths and downstream ends of nearly all rivers in Florida, and tides alone affect an even greater reach of most rivers by causing current reversals and tidal cycles of shoreline submergence and exposure. River shorelines exhibit a wide range of physical conditions and vegetation as a result of the strong gradients established by river flow and tidal action. Saltwater marshes (or mangroves in southern areas) grow along the shoreline or as islands in lower to middle river reaches, but are replaced by brackish or freshwater marshes farther upstream if suitable substratum is available. Tidal freshwater marshes occur in Florida rivers but are not extensive. Instead, brackish marshes end abruptly at the downstream side of floodplain forests. Current reversals and shoreline submergence caused by tidal action can occur for several kilometers upriver, through the floodplain forest.

EFFECTS OF RECENT SEA LEVEL RISE ON FLORIDA WETLANDS

The effects of a gradually rising sea level have been read by geologists in the evolution and modern appearance of coastal landforms in the Everglades and marsh-dominated shorelines of the Big Bend region. Studies of comparable depth are presently unavailable for bays and estuaries or tidal rivers of the state.

Peats of mangrove origin occur under modern mangrove forests in the Everglades and below sands in Florida Bay and the Gulf of Mexico. Freshwater muds occur below the peat which indicate that mangrove forests expanded inland as sea level rose during the past several thousand years. The interface between peat and mud and samples of these materials have been used to date their periods of deposition, which represent the rates of sea level rise, deposition, and lateral movement. In general, submergence rates have equalled the rate of coastal sedimentation, about 3.4 cm/100 years for several centuries (Scholl 1964a).

The near equivalence of these rates does not imply that all forests have survived. First, mangrove peats are buried offshore where no modern forests stand. Second, the thickness of peat deposits thin landward. Third, mangrove peats eroded from other forests have probably settled in depositional environments along the coast. Finally, there is ample evidence today of forest erosion along the open coast; dissection of the forest by channels and intra-forest bays; and inland expansion of forests in tidal brackish and freshwater areas (Scholl 1964b; Spackman, Dolsen, & Riegel 1966). Altogether, the geologic and botanic record depicts "tracking" by mangrove forests of a steadily transgressing sea, with seaward losses and landward gains resulting in a dynamic stability for the wetland system.

A similar pattern of lateral translation by wetlands has been found toward the southern end of the marsh system in the Big Bend region, a sand-starved coastline (Hine & Belknap 1986). Low uplands have been drowned by the sea, isolating hammocks of terrestrial plants over highpoints in the underlying bedrock. The hammocks are surrounded by *Juncus* marshes growing on autochthonous peats (produced by the marshes). Interior marsh areas are dissected by small creeks aligned with fractures in the underlying limestone. Seaward marshes have larger creeks and more open water and fewer hammocks. The bedrock highs exposed by tidal action provide attachment sites for oysters, which are developed as coalesced reefs aligned perpendicular to east-west tidal currents. As in the Everglades example, the Big Bend marsh system is eroding on its seaward face, expanding inland, and accreting vertically at rates controlled by local conditions. In both the Big Bend and Everglades areas, major sediment redistribution probably occurs during hurricanes, and the extent of their effect is preconditioned by the accumulated action of sea level rise during antecedent periods of calm (Davis 1940; Hine & Belknap 1986).

Far less complete but supportive evidence is available for bays, estuaries, and tidal rivers. Trend analyses of wetland vegetation have been conducted based on aerial photographs, for the period 1950-1980 in the Tampa Bay region (Tampa Bay Regional Planning Council 1986) and for the period 1945-1982 in the Charlotte Harbor region (Harris, Haddad, Steidinger, & Huff 1983). In both cases, examples of inland encroachment of marshes and mangroves were found, especially in Charlotte Harbor. Although marsh expansion was less evident in the Tampa Bay area, it was noted where fringing uplands were low and level. Moreover, there probably has been an overall decrease in saltern area, at least in Hillsborough County, which cannot be attributed to upland development.

Scant data exist at present in Florida for tidal river effects attributable to sea level rise. Wharton (1985) reconstructed wetland vegetation in the tidal portion of the Myakka River (Sarasota & Charlotte Counties) from surveyors' notes in the 1840s and determined that mangroves presently occur some 7.4 river kilometers upstream of their historic location in downstream areas. This difference could be interpreted as a response to rising sea level but may also be attributable to freeze damage, recruitment during droughts, or other natural factors.

PROJECTED RATES OF SEA LEVEL RISE

An increase of carbon dioxide and other "greenhouse gases" has occurred during the past century due to deforestation, industrialization, and population growth. These gases promote atmospheric warming. The National Academy of Sciences (1983) predicted a 1.5-4.5 C warming if greenhouse gases double, a process which is expected with reasonable certainty during the next 100-300 years.

These predictions have stimulated a number of estimates regarding the effect of atmospheric warming and climatic feedbacks on sea level. As part of the NAS report, Revelle (1983) forecasted a 70 cm rise in sea level by 2085, in response to a global warming of 2.7 C. The U.S. Environmental Protection Agency estimated in 1983 that sea level would rise from a low of 56 cm to a high of 368 cm by 2100, and provided several intermediate estimates as well (Hoffman et al. 1983). Following refinement of glacial and other studies, EPA revised their estimate in 1986 to a 57-368 cm range by 2100 (Hoffman et al. 1986). Thomas (1986) also has estimated the 2100 sea level to be 64-230 cm above the present stand of the sea.

These calculations are based on a complex series of measurements, assumptions, and relationships between atmospheric, oceanic, and biologic processes, and are under continuous refinement. For purposes of this paper, however, it is instructive to note:

(a) Sea level has been rising in Florida at 1.7-2.4 mm/yr (Hicks et al. 1983).

(b) Marshes have accumulated sediments at rates one order of magnitude greater (Hatton et al. 1983); upper limits of sedimentation are not known, but 10 mm/yr submergence is considered "catastrophic" (Orson, Pangestou, & Leatherman 1985).

(c) Low estimates for future sea level rise fall between 5-6 mm/yr (Hoffman et al 1986; Thomas 1986).

IMPACTS OF RAPID SEA LEVEL RISE ON FLORIDA WETLANDS

A number of adverse impacts will result if sea level rises more rapidly than Florida wetlands can accumulate sediment. Tidal marshes and possibly mangroves will drown when submerged so often that waterlogging occurs, or when soil chemistry is adversely affected. These effects will be more pronounced along the seaward edge of wetlands and along creeks. Creek widths will increase as dissection of forests or marshes proceed, resulting in an increase in relative water surface area and also in marsh or forest edge (Hine & Belknap 1986).

Other changes will occur as sediment-related processes are affected. A reduction of plant biomass may lead to reduced rates of *in situ* organic accumulation, as well as a reduced ability of marshes or forest root-zones to trap water-borne sediments. Erosion of destabilized wetland borders may occur. On balance, some wetlands could benefit because eroded sediments become available for deposition elsewhere (Orson et al. 1985). Also, scarps may be relocated which could create additional marsh or forest substratum from upland areas and add additional sediments to down-gradient wetlands.

The areal effect of rapid sea level rise is likely to be an accelerated version of existing processes, e.g., heightened erosion of seaward wetlands and increased wetland expansion, landward (where low uplands are available). Species replacement within a particular wetland is probable (Redfield 1972). Where increased tidal access brings saltwater into fresh water marshes or cypress domes, replacement by salt tolerant species can be expected. Tidal freshwater marshes in rivers are likely to be eliminated if *Juncus* or other salt marshes migrate upriver in response to increased salinities because the tidal freshwater marshes will have no suitable, intertidal substratum farther upstream (Estevez 1988), at least not until the lower reaches of floodplain forests are killed and their sediments are redistributed.

The effects of sea level rise on Florida's tidal wetlands will be aggravated by human-caused changes to coastal environments. Chief among these are:

Shoreline Protection

Seawalls, upland fill, or other protective measures on the upland side of coastal wetlands will prevent their migration in response to rising sea level. The result, in light of heightened erosion on the seaward side, will be a net loss of wetlands as they are "pinched out" (Titus, Henderson, & Teal 1984). The current practice of regulatory agencies to permit conversion of salterns to storm water basins--or even uplands--is particularly retrogressive in this respect.

Channelization, Spoiling, and Dikes

These structures accelerate the rate of local subsidence by compressing marsh sediments; starving marshes of water-borne sediment by routing water away from marshes; promoting saltwater intrusion; and changing patterns of tidal inundation (Craig, Turner, & Day 1980). Spoils and dikes, as well as clearing operations on the upland sides of wetlands, promote invasion by exotic species such as Brazilian pepper (*Schinus terebinthifolius*) and Australian pine (*Casuarina equisetifolia*), which crowd out native wetland species.

Diversions of River Flow

The distribution, abundance, and condition of wetlands along the tidal reach of Florida rivers are controlled by the physical and chemical interaction of freshwater discharge and tidal action. River discharges provide nutrients and sediments, affect water levels, and establish salinity gradients in tidal rivers (Mahmud 1985). Flow reductions mimic sea level rise by shifting salinity patterns upstream, dislocating stationary and dynamic estuarine environments. Where flow reductions occur *in addition to* sea level rise, rapid and catastrophic wetland impacts *can be expected* in the tidal reaches of coastal plain rivers, especially for spring-fed systems.

The special case of seagrasses has not as yet been considered in relation to the sea level rise issue. Flooding, waterlogging, erosion and may other impacts to wetlands are not relevant in the case of these plants, but increased light attenuation by a longer light-path may be significant for seagrasses located near their compensation depths. Sediment redistribution caused by erosion may increase local turbidity, which can also result when the mean depth of lagoons and bays increases to the point that fetch, wave climates, current structure, or other physical features are changed. For example, the loss of offshore bars along "estuarine shelves" (Lewis, Durako, Moffler, & Phillips 1985) could expose inshore grassbeds to higher wave energy and turbidity. Estuarine grassbeds and submerged aquatic vegetation would also be affected as salinity patterns changed in tidal rivers.

IMPLICATIONS FOR WETLAND MANAGEMENT

Florida shares equally with the rest of the nation the task of choosing appropriate responses to the issue of sea level rise, but the state has proportionately more wetlands to lose if responses are inappropriate or are implemented too late. If mid-range to high rates of sea level rise can be forecast with greater certainty, Florida wetland managers should be ready to implement well-reasoned programs, rather than to start their design in a crisis atmosphere. In light of the wetland impacts likely to occur even under historical rates of sea level rise, a few recommendations can already be tendered.

1. Use Relevant Vertical Reference Data

Neither mean sea level of 1929 nor the National Geodetic Vertical Datum are intrinsically meaningful with respect to modern wetland elevations. Sea level has risen in Florida since 1929 and so have its wetlands. The National Ocean Survey redetermined local tidal datum planes in the 1970s, and these data should be consulted when planning tidal wetland projects. Elevations for new projects can also be established by surveying nearby natural marshes.

2. Establish Useful Life as a Design Criterion

When time horizons for wetland projects are discussed at all, the usual sense is that the system will be expected to persist indefinitely. This is a desirable goal even though hurricanes, freezes, and other natural forces set upper limits to the longevity of a specific wetland. It may be useful to intentionally design "utility wetlands" with shorter useful lives than "wilderness wetlands" (Clark 1986). Also, the time lines set for created wetlands may not need to be as long as ones set for restoration or mitigation wetlands.

3. Take Advantage of Upland and Inland Sites

Wetland creation, restoration or mitigation projects in areas where sea level rise impacts will not be felt first include tidal rivers; the upper ends of bays and estuaries; blind ends of lagoons; and creeks and streams flowing to tidal waters. In some cases it may be sufficient to prepare low uplands for natural wetland recruitment, through removal of ditches, spoils, or other barriers (see below).

4. Prevent and Remove Upland Barriers to Wetland Migration

Florida's extensive lowlands near tidal wetlands are important as incipient wetlands. Barriers include roads, fill, seawalls, ditches and buildings. These structures could be removed; removed once depreciated; or never built in order to allow for wetland migration. Whether or not sea level rise accelerates, one meaningful measure would be protection of salterns. These tidal landforms are being converted into uplands, stormwater catchment basins, or other uses which will prevent wetland migration from occurring.

5. Dedicate Low-lying Uplands

Governments and developers of large coastal properties should inventory the actual location and extent of lowlands adjacent to tidal wetlands and consider their long-term preservation as a land-use and planning tool. Property can be conveyed fully as part of site planning or perpetual easements could be dedicated. Such lowlands (but not salterns) could be used as freshwater wetlands for stormwater management, until they are encroached upon by tidal wetlands.

6. Establish Long-term Monitoring Programs

Sea level *per se* is actively monitored but there is a need for data on current wetland dynamics. Much more information is needed in Florida on elevations of specific wetlands; rates of sediment accumulation and loss; historic trends in unstudied wetlands; effects of exotic species on wetland movement onto lowlands; saltern geomorphology and habitat value; and wetland dynamics in tidal rivers.

CONCLUSION

Sea level is likely to rise at rates which are significantly greater than have occurred in the recent past, but even if coastal wetlands can accumulate sediment at comparable rates, Florida will probably experience net wetland losses due to sea level rise because of the extensive amount of protected shoreline already in place; ditches, channels, and levees created for mosquito control, land-fill and navigation; existing hydrological alterations; and increasing demands by a growing population for flood control and potable water.

The moderate to high rates forecast for sea level rise should be detectable by measurements made during the next twenty years. Two decades would not be an excessive period to wait for more definitive data except that Florida is experiencing tremendous population growth, especially along its coastlines, and large areas of the state near tidal wetlands will be developed in twenty years. Management decisions regarding sea level rise, tipped in favor of wetlands, would only put off more intensive land uses for a few years if predicted rates do not materialize, but would significantly protect wetlands if higher rates do occur.

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GUIDELINES FOR CREATION OF SMALL STREAM FLOODPLAIN ECOSYSTEMS IN NORTH AND CENTRAL FLORIDA

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ABSTRACT

Twelve first order streams in north and central Florida were characterized by species composition, vegetation type and general stream morphology. The streams studied are located in seven different river basins in or near the phosphate mining regions of north and central Florida. Two basic stream types with different cross sectional relief were studied; deeply incised streams with narrow floodplains and wide streams with broad, flat floodplains. Basin parameters of slope, stream length, percent hydric soils and watershed area were measured and differences between stream types were indicated. Vegetation types were found to be distributed differently with respect to stream type and stream reach. Some vegetation types such as *Gordonia lasianthus* and *Taxodium ascendens* were found only in the headwater locations, while most other types were distributed throughout headwaters, midreach and lower reach. Similarly, some vegetation types such as *Nyssa sylvatica* var. *biflora* occurred more frequently in broad, flat stream types while vegetation type *Liquidambar styraciflua*-*Quercus laurifolia*-*Quercus nigra*-*Pinus elliottii* occurred more frequently in incised stream types. This study indicates that general stream morphology, basin characteristics and vegetation types can be used as guidelines for restoration and creation of small streams.

INTRODUCTION

The overall objective of this research was to document the physical and biological organization of relatively undisturbed small stream watersheds and their floodplain ecosystems in central Florida for use as guidelines for reclamation of phosphate mined lands.

Stream floodplain ecosystems are wetland forests that border streams. There may be a variety of types of floodplain ecosystems along one stream from headwaters to mouth. The type of wetland that develops depends on the energy of the stream, topography, water quality, and sediment carried by the stream (Wharton et al. 1977).

Studies describing the relationship between plant species distribution and hydrologic regimes have concluded that frequency and duration of inundation exert a controlling influence on the composition, structure, and distribution of wetland plant communities (Shelford 1954; Sigafoos 1964; Bedinger 1971; Bell 1974; Johnson & Bell 1975; Bell & Johnson 1975; Robertson et al. 1978; Frye & Quinn 1979). Most of these studies have been on high relief streams in northern areas of the United States.

Few studies have related vegetation characteristics to watershed basin parameters in Florida. Central Florida stream landscapes are unique in their character of non-alluvial, low relief watersheds. Studies which have examined floodplain ecosystems of Florida have generally investigated larger stream systems, for example higher order streams, than the first order streams of this report. The most notable of these studies are Monk's study of successional patterns in major floodplain forest types in north and central Florida (Monk 1965; Monk 1966); Leitman's study on the relationship between tree communities and soil composition, elevation and water levels on the Apalachicola River (Leitman 1978; Leitman et al. 1983); and Clewell's study of the floodplain communities in west central Florida riverine forests (Clewell et al. 1982).

Factors affecting tree species distributions and the structure and development of forest communities on river floodplains include flooding frequency (Bell 1974; Johnson & Bell 1975), flood duration (Huffman 1976), period of inundation and flood-generated physical damage (Sigafoos 1964), edaphic parameters (Shelford 1954; Robertson et al. 1978); Frye & Quinn 1979), and seedling flood tolerance (Bell 1974; Bell & Johnson 1975; Theriot & Sanders 1986).

Bedinger (1971, 1978) found plant species are distributed along a flooding gradient according to physiological response to flooding. Other research indicates a strong influence of hydrologic environment early in plant development (Hosner 1958). Tolerance of seeds to flooding, degree of root zone saturation and inundation of seedlings are the main factors influencing survival and growth (Demaree 1932; Hosner & Boyce 1962; Dickson et al. 1965). These tolerances largely control the distribution of species in relation to flooding. Adult species distributions are also influenced by flooding regimes (Teskey & Hinckley 1977; Harms et al. 1980; Malecki et al. 1983). Flood frequency and duration constantly change during the geomorphic history of the floodplain altering environmental conditions throughout the floodplain and favoring different species at different times.

Factors which should be taken into account when restoring landscapes on a watershed scale include defining watershed boundaries, their shape, size and slope; the stream channel slope, sinuosity and channel dimensions; and magnitude and length of streams in the Florida landscape. Data which can be used in estimating basin runoff in storm events include many of these same factors (USDA 1986). Hydrologic models for runoff hydrographs generally require information on watershed size, slope, proposed land uses, soil characteristics for retention and infiltration, amount of surface retention in lakes and wetland areas, travel time of flows including slope of basin, length and depth of flow path and roughness of flow surfaces, and type and extent of vegetative cover (USDA 1986). A number of these parameters were measured in this study.

STUDY SITES

Twelve first order streams in north and central Florida were characterized by vegetation type, species composition, and general stream morphology. Streams studied were selected from seven different river basins in or near the phosphate mining regions of north and central Florida. Two basic stream types with different cross sectional relief were studied; deeply incised streams with narrow floodplains and wide streams with broad, flat floodplains.

Incised streams were concentrated in the St. Mary's, Suwannee, Peace, and Little Manatee river basins, while broad, flat streams were distributed in the Kissimmee, Oklawaha and Anclote river basins (Figure 1).

Initial selection of streams was based on streams with discharge data records. Since few first order streams are gauged by the U.S. Geological Survey (USGS) or Water Management Districts, several streams were selected which had no available discharge data. The major factor in selecting study sites was the amount of disturbance in headwaters, midreach and lower reach areas. Areas with excessive cattle grazing or recent logging in the floodplain were not used.

MATERIALS AND METHODS

Watershed Parameters

Basin parameters of slope, stream length, percent hydric soils and watershed area were measured.

Watershed boundaries for the twelve streams in this study were delineated using USGS 7.5 min series topographic maps and drainage basin maps, both at a scale of 1:24,000. The area of each watershed was measured on these maps using a polar compensating planimeter.

Percent hydric soils, calculated from the ratio of area of hydric soils to the total watershed area, were measured from Soil Conservation Service (SCS) Soil Survey maps of various scale, most frequently with 1:20,000. Watershed boundaries were transferred to soil maps by a grid system. Wetland soils were defined using Hydric Soils of Florida as defined by the SCS (1986). Areas of hydric soils were measured using a Houston Instruments Complot Series 7000 digitizer.

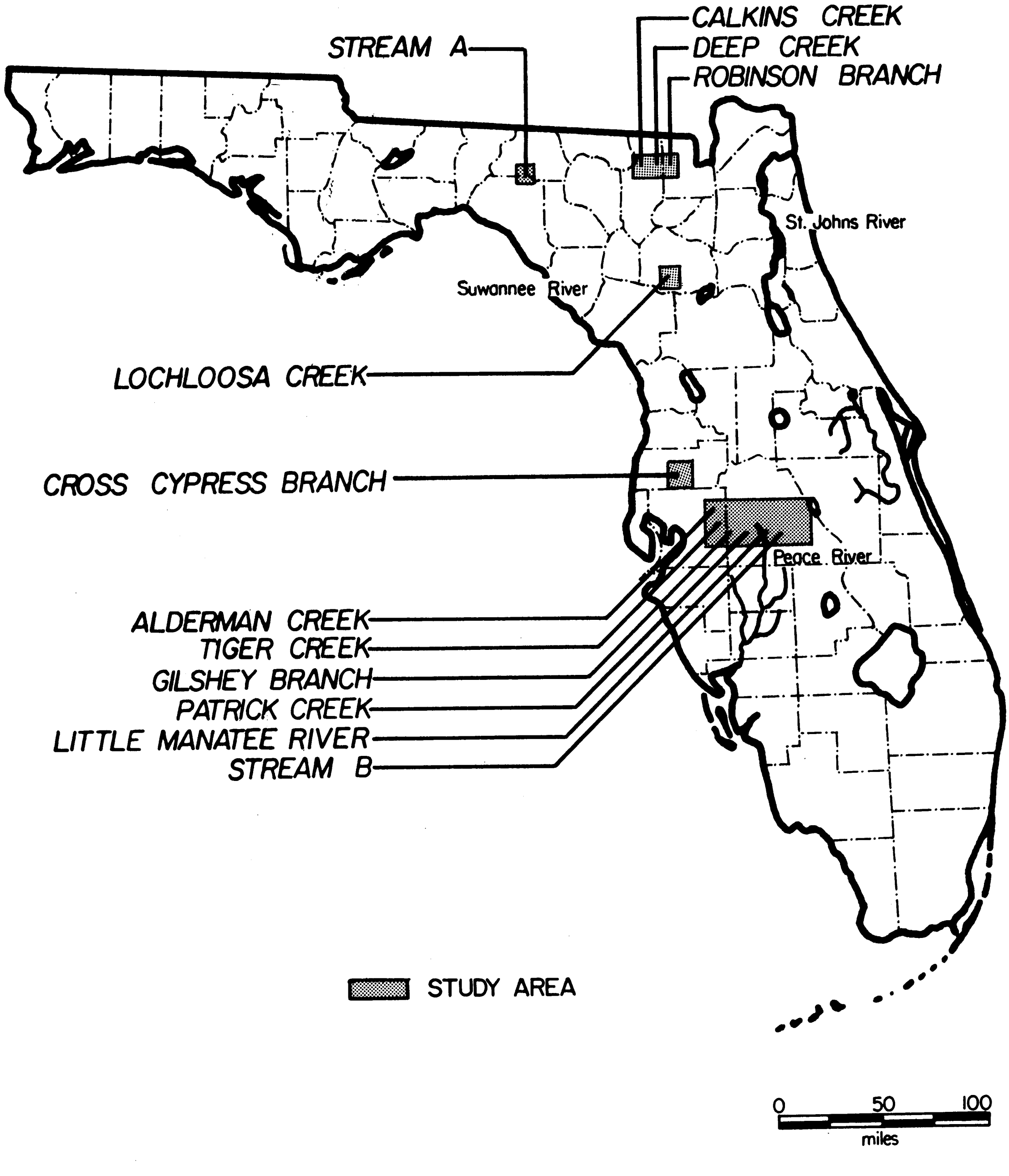

Figure 1. Map of Florida showing locations of study sites.

Stream lengths were measured using a chart wheel along lines delineating stream channels, measuring the length in inches, on USGS 7.5 min series topographic maps. The lengths were then converted to kilometers. Lengths at each crossing of contour lines were noted, and these were converted to slope, in meters per kilometers (m/km).

Stream Parameters

Three cross sectional quadrats were established on each of the twelve streams in areas situated at headwaters, midreach and lower reach locations. The lower reach site was generally located near the mouth of each study stream. The midreach quadrate location was chosen approximately halfway between the mouth and the headwaters. The headwater quadrate sites were located where no discernable channel was evident on USGS 7.5 min series topographic maps. Generally, the headwaters quadrate locations were characterized by sheet flow through broad, flat wetland areas. In all cases, USGS topographic maps were used to initially select quadrate locations, and aerial photography, at a scale of 1:24,000 was used to further identify floodplain areas and vegetative composition. Subsequent field verification established exact locations.

Vegetation sampling of canopy, subcanopy and herbaceous plants was taken from the upland on one side of the stream through the floodplain to the opposite upland. The sampling area consisted of linear quadrats 5 m wide and a minimum of 40 m in length divided into 10 m intervals.

Physical data of temporary ground water level, organic matter depth, and elevation readings were determined along the length of each transect. Elevations of 2- to 3- m intervals along quadrats were measured to the nearest centimeter using a level and a stadia rod. Organic matter depth was measured using a 2.54 cm soil corer at 5 meter intervals along the transect. Temporary surficial ground water wells were dug with a 8.25 cm soil auger. Wells were measured every 10 m along transects, more frequently when conditions warranted.

A program was written to interpolate between the line distance, relative elevation, organic matter depth, and water level for each individual tree for use in the cluster analysis. Cluster analysis was used to sort individual trees into groups with similar environmental conditions. Statistical Analysis System (SAS) programs for average linkage cluster analysis and principal component analysis were used. In the first cluster analysis, tree species and environmental variables specific to each individual tree were analyzed to determine vegetation "associations" to determine generalized vegetation "types" defined by similarity of importance values.

RESULTS

Results of watershed measurements are presented in Table 1. These physical parameters include total area of watershed, basin slope, stream length, upland to wetland ratio, and percent of basin which contained hydric soils. Watershed area varied from 832 hectares to 26830 hectares. Basin slope, defined as the change in elevation from basin summit to mouth divided by basin length, varied from .19 m/km to 2.1 m/km. Stream length in these first order streams varied from 22.9 km to 3.7 km. Percent of hydric soils measured in ten basins varied from 48% in Cross Cypress Branch to 12% in Lochloosa Creek.

Streams with broad floodplains and wider channels had larger watershed areas, flatter basin slopes, shorter stream length and lower percent of hydric soils (Table 1). For incised streams average values of watershed areas were smaller, basin slopes were steeper, stream channel length was longer and the hydric soil ratio was higher. This indicates more wetland soils, therefore more wetland storage per watershed area.

Table 1. Watershed characteristics of incised and flat streams.

Stream Type	Stream Name	Watershed Area (ha)	Basin Slope (m/km)	Stream Length (km)	Percent Hydric Soil
Incised					
	Deep Creek	26830	.30	22.9	48
	Robinson Creek	8653	.48	14.8	26
	Calkins Creek	5047	.44	16.9	--
	Alderman Creek	2253	.79	9.0	13
	Little Manatee River	2020	.94	7.9	24
	Gilshey Branch	832	1.75	5.2	17
	Stream B	455	2.10	3.7	
Mean		6584	.97	11.49	24
Flat					
	Stream A	23007	.19	10.00	--
	Lochloosa Creek	10376	1.25	14.20	12
	Tiger Creek	5902	.48	6.27	15
	Patrick Creek	3847	1.27	6.50	14
	Cross Cypress Branch	2187	.37	7.24	34
Mean		9063	.71	8.80	18.75

Slopes of the channels at the reach quadrate sites, for each stream reach are listed for comparison in Table 2, divided into stream types (incised and flat). The mean values for stream channel slopes show the group of streams with incised stream channels have continuously steeper slopes as the stream flows towards the mouth. The mean slope values for flat streams indicate that stream channels tend to have steeper headwater areas than the midreach and lower reach areas, although still flatter than incised stream headwater slopes. The midreach and lower reaches of flat streams have similar slopes. The mean slopes for incised streams indicate minimum slopes in headwaters areas increasing to maximum slopes in their lower reaches.

Table 2. Channel slope of streams by reach (m/km).

	Reach		
Stream Stream	Headwaters	Midreach	Lower
Incised			
Alderman Creek	.69	1.93	3.06
Calkins Creek	.17	.57	4.30
Deep Creek	.31	1.00	.81
Gilshey Branch	6.31	4.98	7.28
Little Manatee River	.72	2.50	1.48
Robinson Branch	1.39	.93	4.98
Stream B	2.20	4.36	2.47
Mean	1.68	2.32	3.48
Flat			
Lochloosa Creek	.80	.32	.93
Patrick Creek	.25	.37	.37
Tiger Creek	3.16	.46	.48
Stream A	.36	.64	.45
Cross Cypress Branch	1.18	.68	.22
Mean	1.14	.49	.48

Figure 2 indicates typical cross sectional profiles for headwaters, midreaches and lower reaches indicating varying surficial ground water levels, depth of organic matter, and ground elevations. Typical headwater reaches had flat relief, sheet flow of water, and deep organic deposits. Typical midreach areas had more defined channels within which water flow was confined under normal conditions. The organic depth at the midreach areas varied from occasional deep pockets to shallow surface depths. Lower reach areas had the most defined stream channels and least amount of organic deposits. Water

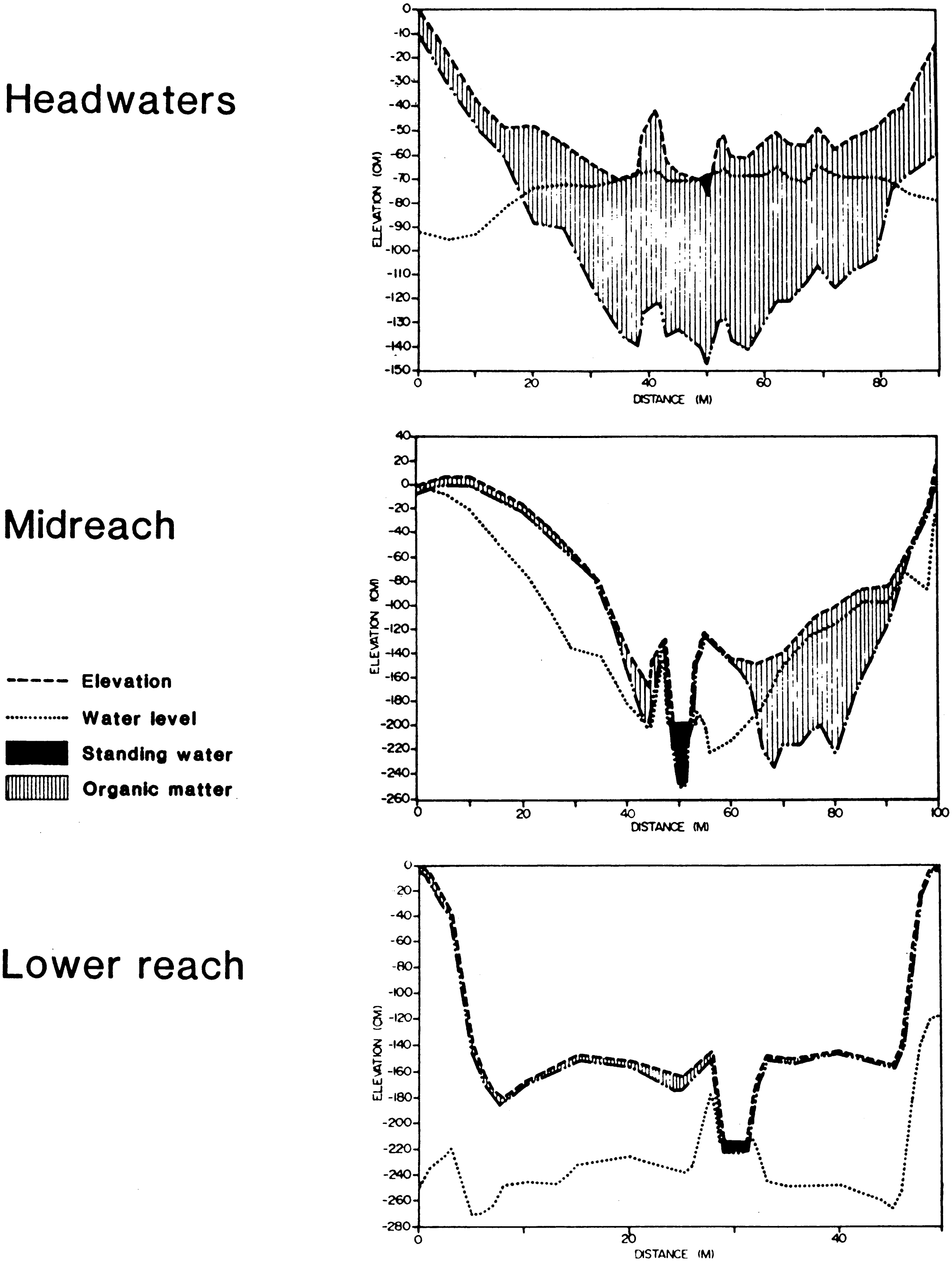

Figure 2. Typical cross-sectional profiles of headwaters, mid, and lower reaches showing variation of ground surface elevation and progressive deepening of stream channel way.

was confined to the stream channel under normal flow conditions.

Data from the statistical analysis on vegetation composition is presented in Table 3. Two cluster analyses were done to achieve the vegetation "types" indicated by tree species name abbreviations. In the first analysis, tree species data and environmental variable data from each quadrate were analyzed to determine vegetation "associations" based on environmental conditions. Secondly, data from the first analysis were clustered solely on the basis of importance values of the vegetation "associations" to determine vegetation types. The importance values were calculated as the average of relative dominance (total basal area of species divided by total basal area of all species) and relative density (number of individuals of the species divided by number of individuals in the plot). The twelve vegetation

Table 3. Composition of vegetation types indicated by importance values of major canopy tree species.

Vegetation Type	Species Abbreviation												
	AR	FC	GL	IC	LS	MV	NSB	PE	QL	QN	QV	TA	TD
MV	3				5	89	1		1		1		
PE					8		4	84		2			2
QN			1		3	3			1	72			
PE-NSB	1		1			1	30	35	1			27	1
TD	1			1			8		7				66
NSB	13	4	1	1	1	1	54	1	3	1		6	9
GL			71			17	3			6		4	
AR-MV	31		1	7	3	22	10	5	4	2	1		1
TA				10			1	1				87	
QL	5	3		4	2		3	4	57	3		2	
LS-QL -QN-PE	6	5		2	20	3	4	8	17	8	1	1	1
QV	6			11				23	4	52			

Key to abbreviations

AR *Acer rubrum*
CO *Cephalanthus occidentalis*
FC *Fraxinus caroliniana*
GL *Gordonia lasianthus*
IC *Ilex cassine*
LL *Lyonia lucida*
LS *Liquidambar styraciflua*
MC *Myrica cerifera*
MV *Magnolia virginiana*
NSB *Nyssa sylvatica* var. *biflora*
PE *Pinus elliottii*
QL *Quercus laurifolia*
QN *Q. nigra*
QV *Q. virginiana*
TA *Taxodium ascendens*
TD *T. distichum*

"types" were grouped from the cluster analysis of 123 vegetation "associations." Over all twelve streams, a total of twelve vegetation "types" were derived by the cluster analysis. The twelve vegetation type name abbreviations on the left column of the table represent types named by dominant canopy tree species. The composition of each vegetation type is shown in the columns to the right by the importance value for the major canopy tree species present.

Figure 3 is an example of a typical distribution of species across the floodplain of a midreach transect location. Note the changing distribution of species with change in organic matter depth, elevation, and water level. This particular example shows three communities along the gradient: one from 0 to 60 m; one from 60-200 m; and one from 200-240 m. The cluster analysis showed this system to have three "associations."

Table 4 summarizes characteristics of each vegetation type showing typical location on the quadrate from the edge to the interior of the floodplain (variable in location, edge of floodplain, or interior of floodplain), number of associations included in each stream type (incised and flat), and reach where associations were found (HW-headwaters, MR-midreach, and LR-lower reach).

Predominant species found in most vegetative quadrats are summarized by stream reach (Figure 4). The bar graph column represents a total of all individual canopy trees in all transects. This data was calculated separately from the cluster analysis to determine distribution of individual canopy tree species in headwaters, midreach, and lower reach sites. Species are distributed unevenly throughout the three stream reaches across both flatted and incised stream types. For example, some species, such as Gordonia lasianthus are more heavily represented in the headwater reach than in the other two reaches, while Taxodium distichum and Fraxinus Caroliniana did not occur in the headwater areas.

DISCUSSION

Reclamation and recreation of landscapes after drastic disturbance, such as phosphate mining, requires an understanding of the interplay between physical characteristics and ecological organization. Through studies of undisturbed landscapes a better understanding of their fundamental organization is possible and better reclamation may be realized. There is no doubt that many of the physical conditions and resulting ecological organization of the reclaimed landscape will be different from those which existed prior to mining. Yet, we believe that general principles for the design of reclaimed lands may be acquired through systematic study of natural landscape organization.

The most important influence on the organization of the Florida landscape is rainfall. With an average rainfall of over 125 centimeters per year, ground water levels, runoff, and evapotranspiration

Table 4. Summary of vegetation type characteristics.

Vegetation Type	Location on Quadrate *	Number of Associations by Stream Type **		Reach ***
		Incised	Flat	
MV	Variable	2	3	HW,MR,LR
PE	Edge	4	3	HW,MR,LR
QN	Edge	2	2	HW,MR,LR
PE-NSB	Variable	9	6	HW,MR,LR
TD	Interior	2	2	LR
NSB	Variable	4	14	HW,MR,LR
GL	Interior	2	3	HW
AR-MV	Variable	16	10	HW,MR,LR
TA	Interior	0	3	HW
QL	Edge	5	4	HW,MR,LR
LS-QL-QN-PE	Variable	14	7	HW,MR,LR
QV	Variable	2	0	HW,MR,LR

* Location of vegetation on the quadrate: variable in location, on edge of floodplain or in interior of floodplain.
** Frequency of occurrence of each vegetation type in incised and flat streams.
***Reaches where associations were found.
HW Headwaters
MR Midreach
LR Lower reach

are all high. The result is a landscape that is dominated by hydrologic processes. A prominent aspect of Florida's landscape is the abundance of small streams and drainage sloughs. Tighe (1988) has shown that, on average, every square mile of north and central Florida has a first order stream. As a consequence, the small stream should be an integral part of the reclamation of phosphate mined lands since the scale of mining is greater than the average drainage areas found by Tighe. Under most circumstances, however, streams are only created if they have been mined. It may be more appropriate to reconsider plans for the post mining landscape and include streams as an integral part of the landscape mosaic and design the morphology and vegetation to match the physical conditions created.

The long-term success of a reclamation project should be measured by how well the landscape functions, not by how well the design mimics the landscape that existed prior to mining. With general principles for landscape design, function may be restored, and with it ecological integrity. In this study, floodplain ecosystems throughout central and

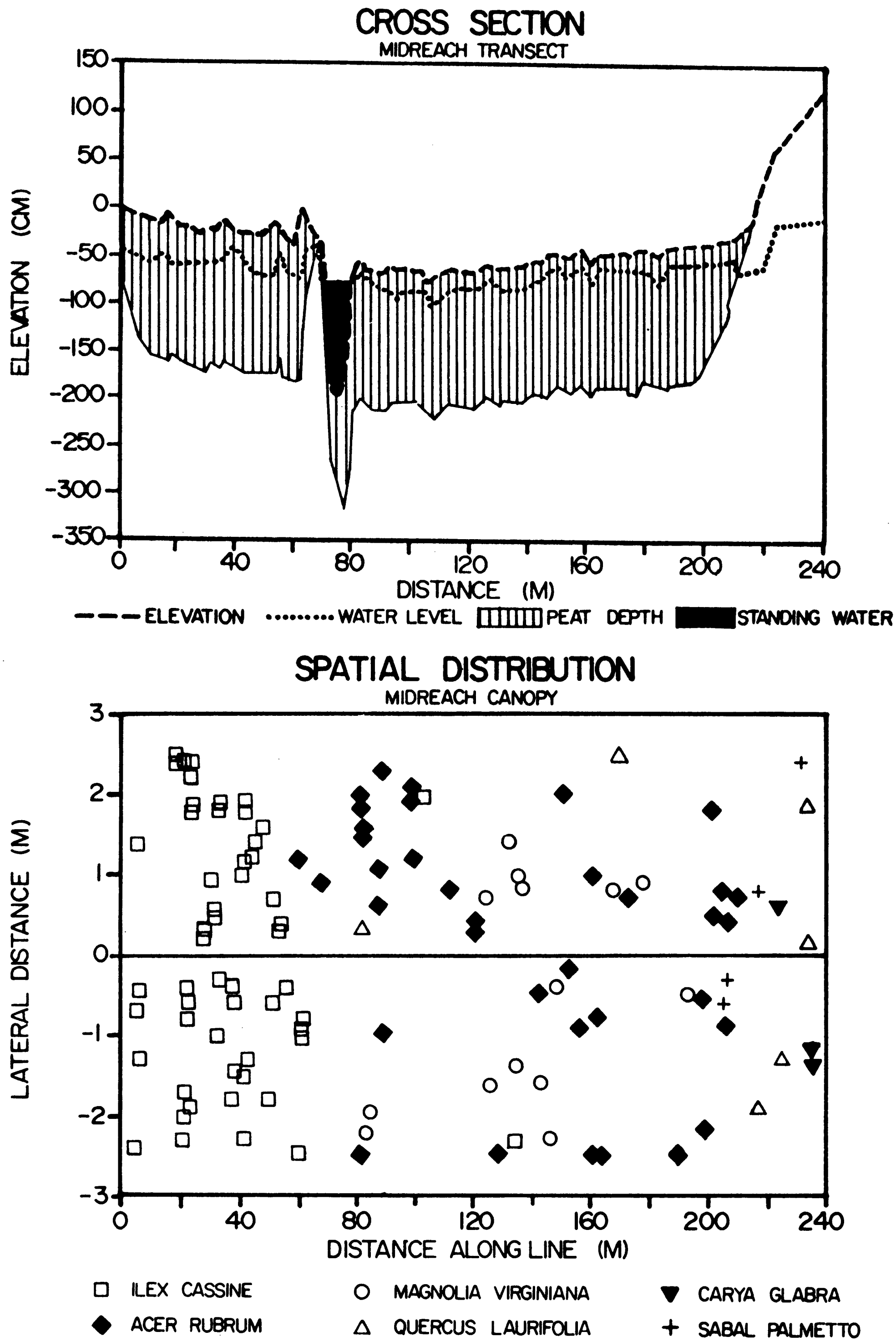

Figure 3. Cross-sectional profile of typical midreach transect showing water depth, organic matter depth, and standing water. Shown in the bottom graph is the spatial distribution of canopy species along the transect.

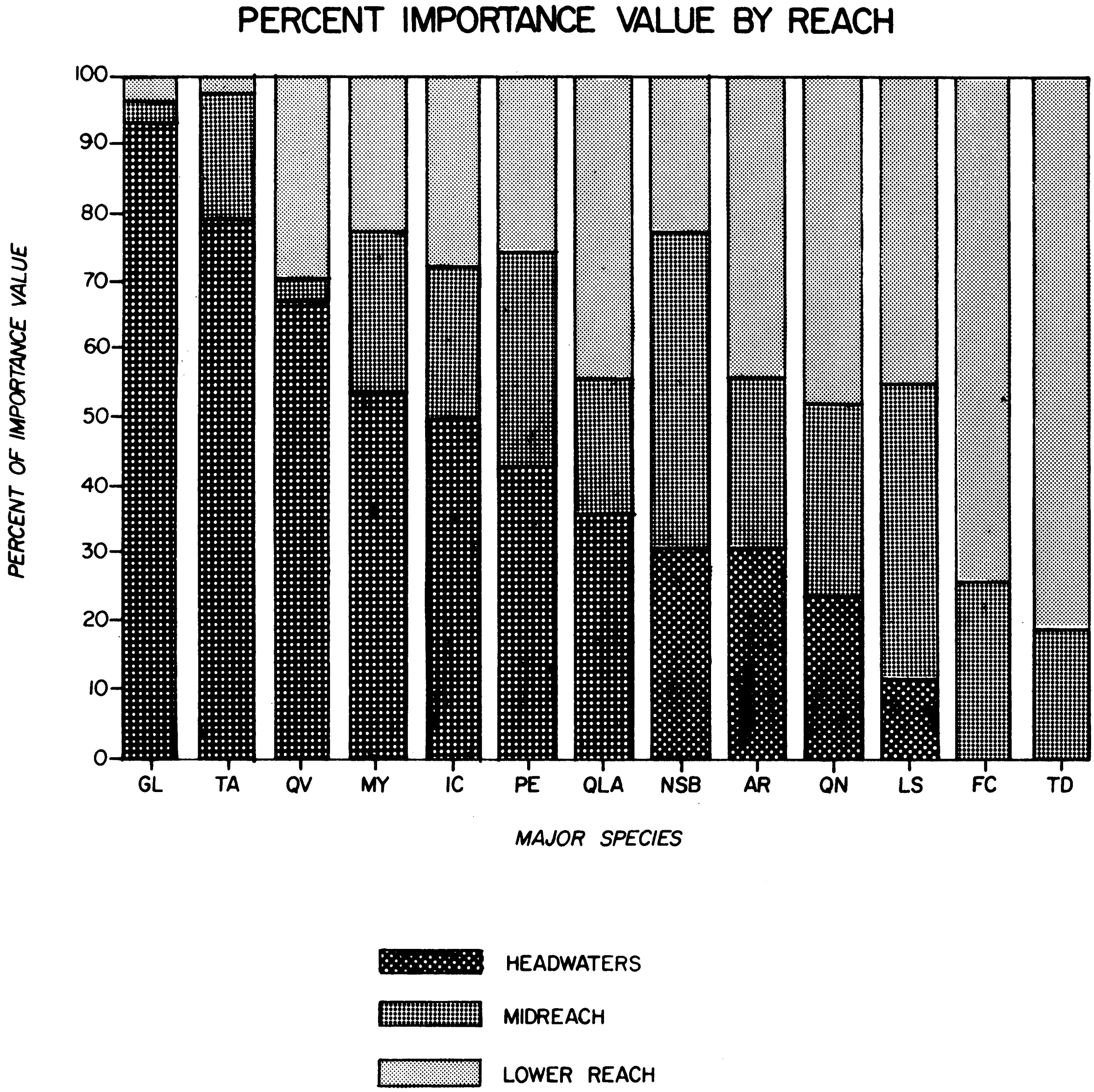

Figure 4. The distribution of major tree species between headwaters, mid and lower reaches. (See notes to Table 4 for list of major species.)

northern Florida were studied, and general stream morphology was related to vegetation as a means of predicting the community structure that is best suited for various physical conditions created as a result of reclamation. With these data, reclamation schemes may be designed that reflect the dominant role of rainfall and its controlling influence on the morphology and vegetation distribution of the landscape.

It is obvious to the untrained eye that streams are different along their lengths from headwaters to lower reaches, yet it is important to understand their differences, especially if the goal is to recreate a landscape following drastic disturbance. Cross section profiles, spatial distributions of vegetation along cross sections, and variation in vegetation types along the length of streams are important indices of landscape organization. These data, combined with data related to the ecological organization of watersheds, may help in making final design decisions concerning recreation of hydrologically and ecologically functional landscapes after mining.

Figure 5 indicates the physical characteristics and suggested planting scheme for a typical midreach stream section. Cross sections and plan views, like those in Figures 4 and 5, are being developed as a part of a larger effort to produce a phosphate reclamation manual that may help in providing guidelines for the recreation of diverse functional stream landscapes.

In order to provide information on important physical and biological parameters that may be helpful in designing and constructing reclamation projects, the variability in vegetation must be reduced to the most important characteristics and most typical situations that can be reproduced in recreated landscapes. This paper represents a consolidation of data from a much larger data base. It has been reduced in detail to present generalized principals. The following guidelines are given as a means of providing a better understanding of the organization of landscapes dominated by small streams and their associated ecological communities.

1. Floodplain ecosystem species composition varies with stream reach. The data from this study shows important differences in the location of vegetation types along the length of the stream channel from headwaters to mouth. For example, the vegetation type GL (*Gordonia lasianthus*) and TA (*Taxodium ascendens*) were found in the headwater areas only, while type TD (*Taxodium distichum*) was a vegetation type common in the lower reaches.

The type of stream cross section, incised or broad and flat, influences the distribution of vegetation types as well as tree species. The vegetation type LS-QL-QN-PE (*Liquidambar styraciflua-Quercus laurifolia-Quercus nigra-Pinus elliottii*) occurs more frequently, for example, in streams with incised narrow floodplains, while vegetation type NSB (*Nyssa sylvatica* var. *biflora*) occurs most frequently in streams with broad, flat floodplains.

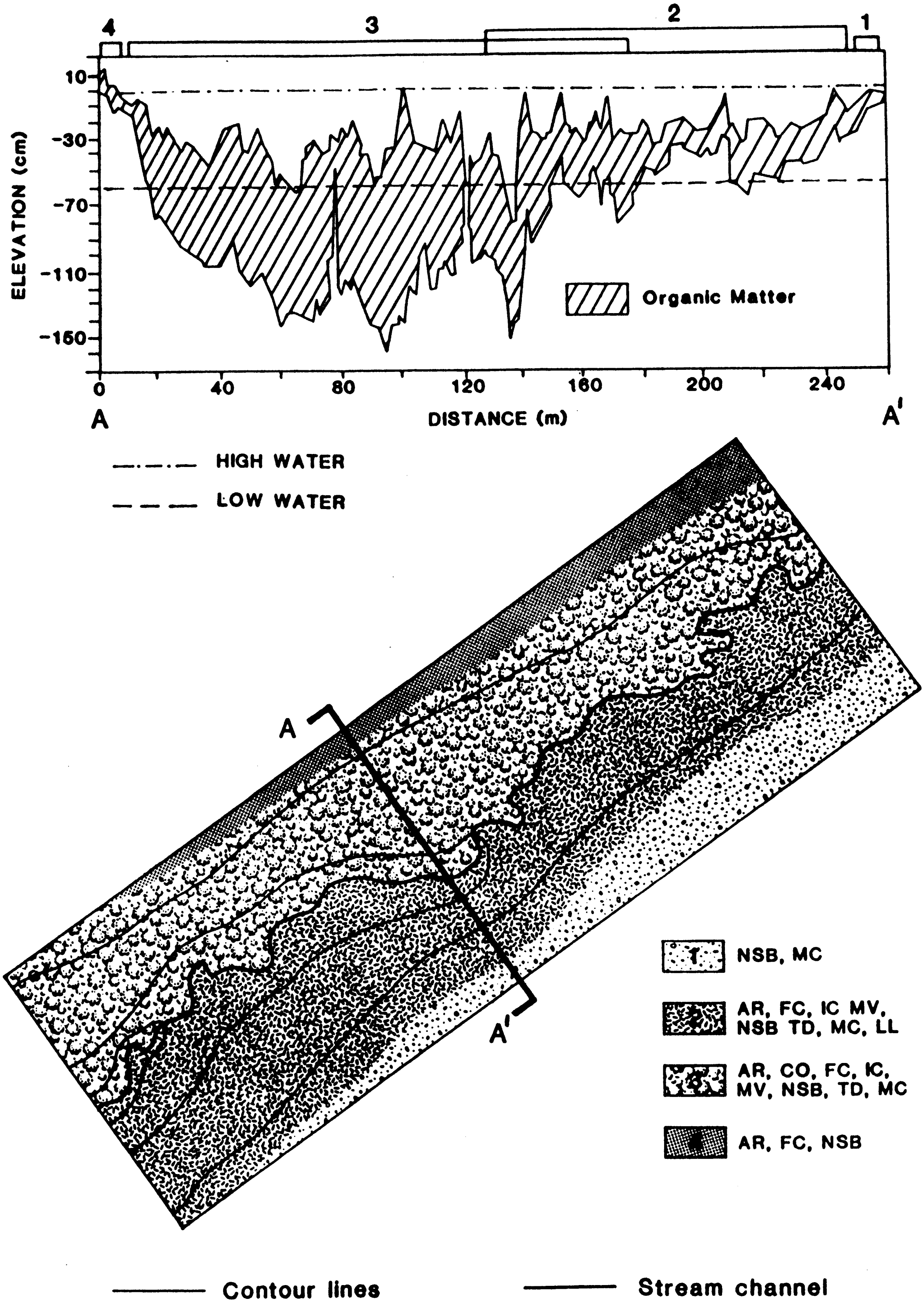

Figure 5. Cross-section and plan view for a typical midreach stream section showing variation in ground surface elevation, typical high and low water, depth of organic matter, and typical vegetation. (See notes to Table 4 for list of species.)

2. Species composition along the cross section gradient from upland to upland probably varies as a function of period and depth of inundation. Incised streams tend to be dominated by larger expanses of more transitional species and have much narrower bands of wet floodplains associated with the stream channel. Flat streams, characterized by broad floodplains tend to be dominated by more hydric species.

3. Physical characteristics of stream channelways vary along the length of streams from headwaters to mouth, probably as a function of discharge volume and channelway slope. Headwater channelways are broader and flatter, with braided channels; midreach channels are more defined; and lower reaches are most deeply incised. Headwater floodplains have higher accumulations of organic matter, probably a result of flatter channelway slopes and slower flow rates, acting more as still water wetlands than as floodplain wetlands. The least accumulation of organic matter is at lower reach locations where slopes are higher and water volumes and velocity are greatest.

4. Physical characteristics such as basin and channel slope, watershed area, and stream length determine the physical characteristics of the stream. Slope is the determining factor of whether a stream is incised or flat. Steeper sloping watersheds produce steeper stream channels which tend to have more incised stream types while wider floodplains occur within flatter watersheds having flatter channel slopes. Factors such as sinuosity and soil type also influence the type of stream.

5. Storage and retention within the watershed affect general stream characteristics. Increased wetland storage has been shown in other studies to minimize peak flows and maximize base flow. In this study, the area of wetland storage in the watershed varied with the stream type. Generally, incised stream types showed larger storage capacity in the watershed as measured by area of hydric soils.

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WETLANDS BY DESIGN: EXAMINATION OF TWO CASE STUDIES OF WETLANDS CREATION BY THE U.S. NAVY AT NAVAL SUBMARINE BASE, KINGS BAY, GEORGIA

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ABSTRACT

The U.S. Navy is developing the Naval Submarine Base (SUBASE) at Kings Bay, Georgia. The Environmental Impact Statement (EIS) prepared to satisfy the requirements of the National Environmental Policy Act (NEPA) for SUBASE requires creation of wetlands as mitigation for loss of comparable habitats resulting from construction. Borrow sites for fill material have been identified as particularly suitable for reclamation as wetlands at SUBASE. The Navy has developed a process which creates the potential for savings by incorporating wetlands design criteria and site configurations into borrow area construction contracts.

A problematic approach was used to design and build low maintenance functional wetland systems with maximum value as wildlife habitats. General case studies using the Navy's approach are examined for a tidal saltwater wetland and a freshwater wetland. These two case studies serve as examples of the planning and design process, and the types of information that scientists must provide engineers for successful creation of wetlands.

INTRODUCTION

National Environmental Policy Act Documentation

The Navy is nearing completion on the largest single peacetime construction effort in United States history. The Naval Submarine Base, Kings Bay (SUBASE), will be the East Coast homeport for the new OHIO class submarines. The first OHIO class "boat" to be stationed at SUBASE will be greeted by complete shore-based equipment maintenance facilities, personnel training facilities, and other support facilities.

SUBASE has been built in an area that was once largely covered by slash pine plantation, gum-cypress swamp, and saltmarsh. Several hundred acres of forest were cleared, and more than a million cubic yards of earth moved to construct the new base. The potential for great environmental impact was clearly present to planners during the initial stages of construction.

Before initial development, an Environmental Impact Statement (EIS) was prepared in accordance with the requirements of the National Environmental Policy Act (NEPA). The EIS spelled out the environmental effects of development at Kings Bay in great detail and demonstrated an important opportunity for development with environmental sensitivity (NAVFACENGCOM 1977, 1980).

From Concept to Implementation

The NEPA process is familiar to many. Its mandate is to include a consideration of a project's effects on man's environment and this mandate is now included as a routine procedure in Navy planning. Planners and environmental scientists review projects before construction to assess anticipated environmental effects. The results of their assessment also support the production of a concept of development. Physical, biological, historical, cultural, and other aspects of the environment are assessed and compared with alternative program plans to define a conceptual outline or boundary of a proposed action. Design and construction efforts normally do not proceed until plans and proposals for the project have sustained public review in the NEPA process. In addition, the irretrievable expenditure of resources to execute a federal project is an action precluded by NEPA until completion of the public review. The NEPA process works best when the project concept is defined with enough latitude to accommodate engineering design, but without so much latitude that an environmental assessment cannot be made. As program execution proceeds, facility design may be modified to meet specific engineering goals without going beyond environmental constraints. Extension of the design beyond these bounds occurs, however, on occasion, and should be preceded with additional NEPA considerations.

On occasion, concept definition during planning is considered to be the end of the matter. That is, during program execution, there exists a real temptation to think that the completion of NEPA requirements means the completion of environmental involvement. The exigencies of project design, procurement, and construction, each with deadlines and unexpected difficulties, strain program management.

Involving additional environmental considerations at this point seems duplicative to those without an awareness of the need or basis for such consideration.

Avoiding environmental considerations during project execution does not simplify program execution and may even cause more difficulty and added costs. We also remember, either through training or experience, that making a decision based on considerations of an EIS probably involved arguments of mitigating factors. Impact mitigation will lead one to conclude a net environmental effect which is less than that without mitigation. The validity of such arguments must be maintained throughout program execution, however, by referring back to environmental considerations made during the planning phase of development. When program managers and design engineers incorrectly view this "environmental presence" as duplicative, superfluous, or inconsequential and subsequently avoid addressing environmental constraints, arguments of impact mitigation and net effect are invalidated. The results of such an action could range from loss of aesthetic attributes of a project and loss of agency credibility to loss of cost efficiency, loss of efficiency in land use, and violation of federal legislation.

Because of the size of development at Kings Bay, environmental considerations were numerous and involved many aspects of salt and freshwater wetlands ecology. In the context of wetlands reclamation and creation, the Kings Bay EIS included non-specific, broad commitments accommodating wetland protection and enhancement activities. Specific commitments were also made to provide mitigative wetland habitat.

This paper examines the program for implementing environmental commitments and mitigation goals during construction at SUBASE in specific wetland creation commitments. Since the wetland installation techniques used at Kings Bay are available in the literature, they are not discussed in any great detail. Of primary importance is the approach to wetland mitigation program management within a larger development program. Lessons learned during the Kings Bay experience can serve as an example for other, similar projects.

STUDY SITE

SUBASE is located in Camden County, Georgia, approximately five miles north of St. Marys, Georgia. It comprises approximately 16,000 acres of upland and tidal wetlands on the western shore of Cumberland Sound, Georgia. Kings Bay itself is a small arm of Cumberland Sound, an estuarine sound separating Cumberland Island from the mainland (Figure 1). Ship traffic from the Atlantic Ocean to the base arrives via a channel at the St. Marys River Entrance northward into Cumberland Sound.

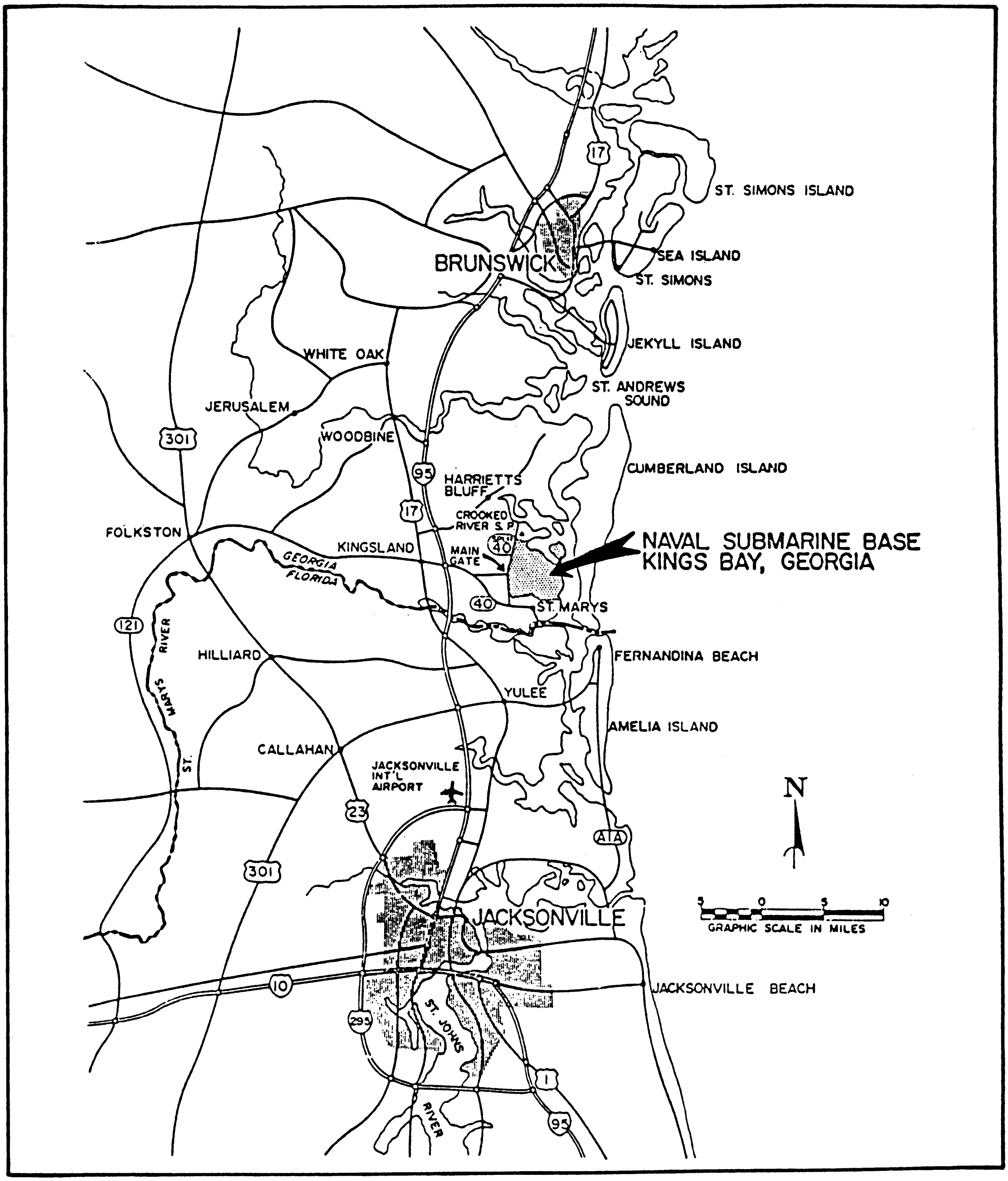

Figure 1. Location of Naval Submarine Base, Kings Bay, Georgia.

METHODS

Tidal Saltmarsh Creation

Even though the Kings Bay EIS included broad commitments for environmental protection and impact mitigation, all impacts could not have been envisioned during its preparation. Additional requirements for wetland creation and restoration occurred during project execution. These new requirements were integrated into the mitigation program as they occurred. In one case, saltmarsh creation as "in-kind" replacement was required through the regulatory involvement of the Army Corps of Engineers Clean Water Act Section 404 permit program.

Requirement. The EIS predicted only 12 acres of saltmarsh loss which would occur through waterfront and dredge disposal development and, considering the level of development, was evaluated as a relatively minor loss. Permits for waterfront construction and dredging in the state of Georgia acknowledged the 12 acre loss, but included no direct replacement requirement. The EIS made no specific commitment for wetland loss replacement either.

However, refinements in understanding of the Kings Bay estuary and its effects on ship traffic and of waterfront logistics led engineers to modify the waterfront designs. The conceptual boundary of the EIS discussed above was too "tight" in this case, requiring the Navy to address the environmental considerations associated with these design changes. The COE was requested to modify existing permits covering waterfront construction to accommodate the changes identified by the Navy. When issued, these permit modifications included conditional statements which required replacement of the anticipated loss through the creation of new saltmarsh from upland areas. Thus, a need for additional provision of newly-created saltmarsh in mitigation of saltmarsh loss was established.

Regulatory requirements caused implementation of saltmarsh creation along with construction rather than afterwards. In addition, the saltmarsh creation projects had to be responsive to any anticipated future losses in order to receive adequate funding during project execution. For maximum benefit, the regulatory requirement for saltmarsh creation was to be incorporated within the construction program. This task included not only planning and design of the mitigation marshes, but also coordinating the creation effort with contracts for facility construction.

Development. The first task to complete in providing mitigative saltmarsh was developing design criteria compatible with the Navy construction program. Since other federal and state regulatory agencies were interested in the progress of environmental protection at Kings Bay, the Navy hosted a wetland creation workshop in which these agencies actively participated in development of wetland mitigation criteria at SUBASE.

The goal of the workshop was to develop a sound scientific basis for upcoming wetland creation projects. Navy personnel, although trained and experienced in wetland creation techniques, were unfamiliar with regional saltmarsh characteristics. The workshop allowed Navy project managers to gather a substantial amount of firsthand information on a limited subject matter in a short amount of time. Federal agency participation included the COE, the National Marine Fisheries Service (NMFS), and U.S. Fish and Wildlife Service (FWS). The Georgia Department of Natural Resources (DNR) also participated, adding a knowledge of regional saltmarsh characteristics.

The workshop was also attended by knowledgeable and experienced consulting wetlands scientists. Representing the University of Florida, Dr. Ron Best added wide experience in herbaceous wetland species requirements. Mr. Roy R. Lewis, III, of Mangrove Systems, Inc. also added experience in development of wetlands in saline tidal environments.

The wetland creation workshop was a first major step in coordinated mitigation implementation. Recommendations developed during the two-day workshop were formalized in a report suitable for use as reference material for engineering use during construction project design (NAVFACENGCOM 1984). The report discussed proposed project location and general grading and planting criteria. In addition, a conceptual plan drawing was included to show how the mitigative wetland was envisioned by the scientist.

The intent was to provide workshop information to the design engineer for incorporation in construction contracts which would require earthen fill material. The requirement for fill could be met while contouring for excavation of mitigative wetlands. Cost efficiency alone justified this method of approach to wetland creation since the costs of excavation for the wetland are absorbed by costs of excavation for construction project fill.

The workshop report identified a location and site plan easily adaptable to phased construction and described the capability for expansion from approximately five acres to about twenty acres of regularly- and irregularly-flooded saltmarsh. The mitigation site is situated near a tributary to the North River on government-owned property at SUBASE (Figure 2).

After determining a suitable mitigation site, participants developed generalized grading and vegetative planting criteria adapted to the area designated for wetland creation. These criteria were refined using information available in the literature. The CRC Press review of wetland creation (Lewis 1982), for example, provides information and source data of the type used. The mitigation marsh concept for SUBASE would be primarily of Spartina alterniflora planted at 1.0 m centers in the lower, regularly-flooded marsh. At the higher irregularly-flooded marsh, vegetation would grade from brackish species such as Spartina bakeri to peripheral shrubs such as Ilex vomitoria (Table

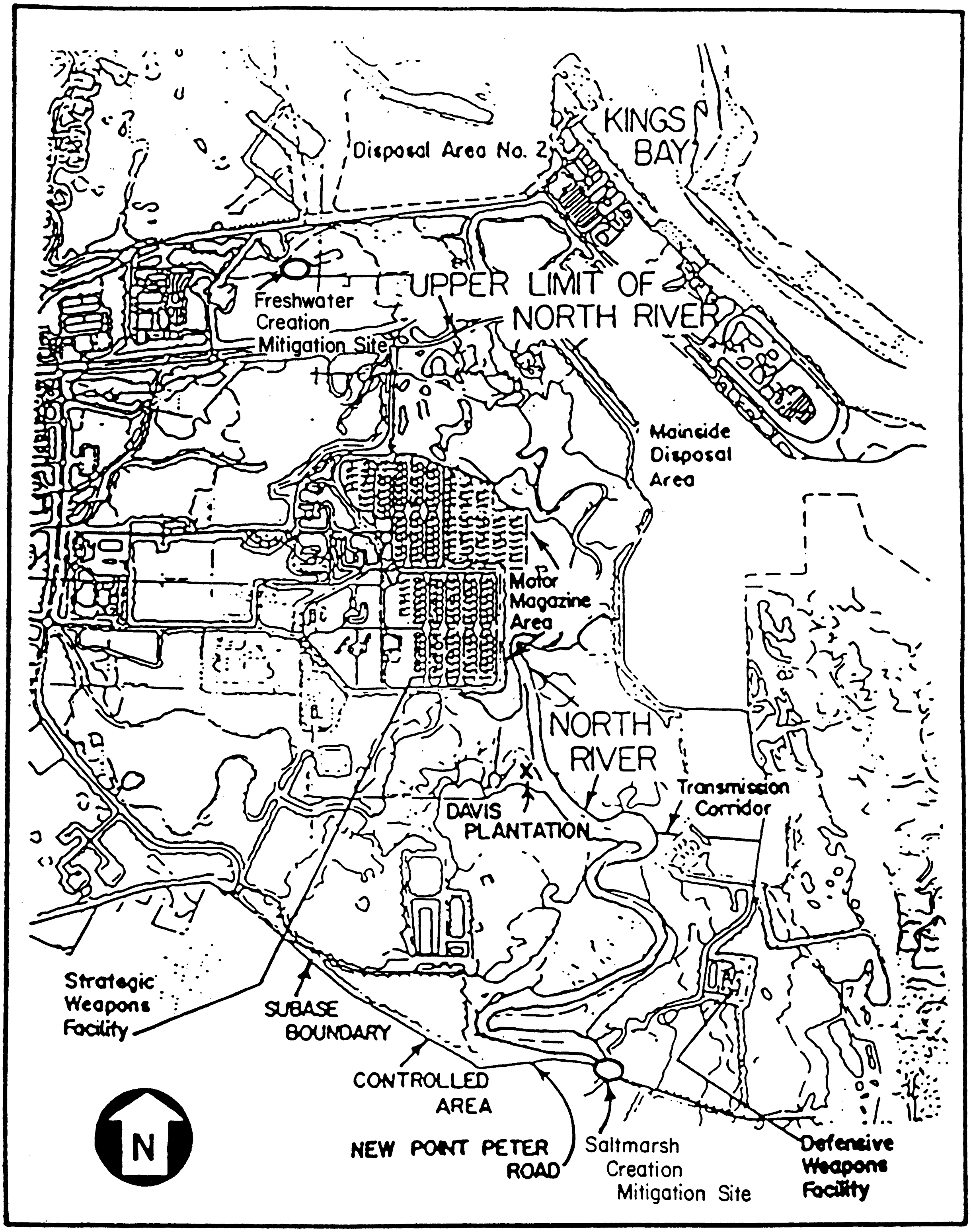

Figure 2. Saltwater mitigation wetland site location, Naval Submarine Base, Kings Bay, Georgia.

1). Fertilization of the planting was recommended for recently-excavated sandy soils.

Several mitigation projects were implemented using the information obtained during the workshop. Specifications were prepared and included in the construction contract. Several items were specified which are peculiar to saltmarsh planting that are not usually found in the standard landscaping specifications. These items involved earthwork, fertilization, transplant stock requirements, and spacing requirements. In these saltmarsh planting projects, specifications required the use of transplants from identified donor sites on base.

Table 1. Listing of representative species according to topographic planting criteria.

Elevation	Representative Species
MLW to +3.3 feet	Mud and sand flats, and *Ruppia maritima*
+3.3 to +6.6 feet	*Spartina alterniflora* *Spartina cynosuroides* *Scirpus robustus*
+6.6 to +8.8 feet	*Baccharis angustifolia* *Spartina bakeri* *Iva frutescens*
+8.8 to +10.0 feet	*Ilex vomitoria* *Myrica cerifera* *Persea borbonia*
+10.0 to +15.0 feet	*Paspalum vaginatum* (sod)

MLW = mean low water

Wherever possible, the specification had to conform with Navy policy and guidelines. Because Navy contracts specify product performance, specifications were prepared to avoid dictating construction methodology. The earthwork specification identified reference elevations where available and required positive drainage with no ponding in the graded slope. Where reference elevations were not available or variable, reference was made to horizontal controls at slope toe and crest. Transplants were specified to be of a young stature by requiring each stem of *Spartina alterniflora* to have no more than three fully emerged leaves. Other transplant requirements were specified according to the particular requirements of the species. For example, *Spartina bakeri* transplants were specified as having three or more stems per transplant in order to obtain adequate rhizome and root material.

Cost estimation was detailed for design engineers according to information available in the literature. Costs for saltmarsh creation/restoration were placed within the formal government cost estimate as part of project procurement (Table 2).

Table 2. Representative unit item costs used at Naval Submarine Base, Kings Bay.

Zones	Cost per hectare	Comment
Transition vegetation	\$1,000 to \$7,500	1,000 plants/hectare
Intertidal vegetation	\$7,500 to \$15,000	Handplanting, 1.0 m centers

Cost per hectare includes planting guarantee for replacement if failure occurs.

Freshwater Creation

Requirement. The EIS contained a commitment to create at least 100 acres of freshwater wetland and wildfowl habitat coincident with construction. Unlike the saltmarsh mitigation requirement, this requirement was generated during the planning process and included in the EIS in an evaluation of net environmental impact from development.

The workshop approach had provided general criteria for saltmarsh creation and restoration. In addition, adequate experience had been gained to implement small restoration projects without outside support. However, the information obtained thus far was either too general for a freshwater wetland, or pertained only to tidal saltmarshes.

Development. A consulting scientist with personnel experienced in freshwater wetland creation was employed by the Navy to develop design concepts and cost estimates for the freshwater wetland. The goal, as in the wetland creation workshop, was to obtain a sound scientific background to provide to engineers for their design effort. Also, the plan was to incorporate the wetland mitigation projects within the overall construction program by supplementing individual construction projects.

In addition to the basic earthwork and vegetative planting requirements, the consultant prepared detailed information regarding wetland configuration, multi-specific detailed planting requirements, complete project specifications, and detailed construction drawings.

Initial site selection proposals were made by Navy environmental scientists and planners with the concurrence of program sponsors. This selection process included a multidisciplinary effort to strike a balance between environmental requirements and construction program requirements.

As indicated in the EIS, freshwater wetlands were required to mitigate habitat losses and should, therefore, be located primarily according to habitat. Such a habitat would include, as a basic criterion, relatively undisturbed openwater with vegetated shallows away from developed or inhabited areas on base. In addition, substrate characteristics were evaluated before selection. Acid forming soils or soils with unusually low pH which could affect vegetative planting results were avoided. Area hydrology was also evaluated at the candidate site to avoid adverse site hydrologic or watershed conditions.

Construction requirements, on the other hand, indicated a need for borrow sources within short hauling distance from the construction site. Since more than one construction project would use a mitigation wetland site for borrow, the site should be centrally located and within a distance arc defined by the maximum acceptable estimated cost per cubic yard for fill including transportation (Figure 2).

Drawings were produced full-size in the standard Navy format and signed by a registered Professional Engineer. Drawings also included title and index pages, horizontal and vertical geometry, plan views and cross sections, and construction details of appurtenant structures. Several pages were devoted to planting plans, schedules, and details. The vegetative planting was structured so that it included a specified number of several "required" species and a specified number of "elected" species the contractor selected from a list placed on the plans. The required species represented a predetermined mix of emergent, floating, and submergent species for a particular area of the wetland identified on the plans. The mix was keyed to a planting schedule with the requisite number of each species.

When followed correctly, the plans would allow appropriate planting according to wetland zone without the responsibility for major decisions regarding plant site selection that could affect the outcome of the wetland. Thus, the contractor is not involved in determining ultimate wetland success and avoids additional liability. Also, a wider range of contractors are capable of wetland installation when the contract is so structured.

Specifications were developed to cover the additional work created by the wetland project. These were also provided in standard Navy format. Specifications for Environmental Protection, Earthwork, Erosion and Sediment Control, Landscaping, and Wetland Vegetative Planting were assembled into a package suitable for inclusion with other contracts. These specifications were reviewed by Navy engineers for consistency with recognized engineering practices and federal procurement regulations.

The formal structure of plans and specifications allowed cost estimation for programming and bidding purposes. Since wetland planning was soundly-based, cost estimation was simply a matter of reviewing plans and specifications for wetland installation requirements. Earthwork cost estimates were developed individually for each construction project assigned to a particular borrow/wetland site. Vegetative or landscaping cost estimates for programming purposes was based on the anticipated cost to install the required species plus anticipated cost to install representative elected species. Because plans and specifications for wetland creation were clear, the contractor could produce bid estimates through an evaluation of his selection of materials and methodology in accordance with contract requirements. Tables 3 and 4 present cost estimates used in planning for emergent macrophyte and tree seedling plantings.

Table 3. Cost estimates for planting emergent macrophytes at Naval Submarine Base, Kings Bay.

Macrophyte Density (#/acre)	Planting Interval[a]	Tublings Cost/plant, ($)	Tublings Cost/acre ($)	Planting Labor Cost/acre[b] ($)
200	15.0	0.25-0.50	50-100	11-22
400	10.4	0.25-0.50	100-200	22-44
800	7.3	0.25-0.50	200-400	44-88
1200	6.0	0.25-0.50	300-600	66-132
1600	5.2	0.25-0.50	400-800	88-176

[a]feet between plants
[b]assumes direct labor at $5.50 per hour

Table 4. Cost estimates for planting tree seedlings at Naval Submarine Base, Kings Bay.

Tree Planting (#/acre)	Planting Interval[a]	Tublings $/plant,	Tublings $/acre	Potted $/plant,	Potted $/acre
200	15.0	0.25-0.50	50-100	0.60-1.50	120-300
400	10.4	0.25-0.50	100-200	0.60-1.50	240-600
600	8.5	0.25-0.50	150-300	0.60-1.50	360-900
800	7.4	0.25-0.50	200-400	0.60-1.50	480-1200
1600	5.2	0.25-0.50	400-800	0.60-1.50	960-2400

[a]feet between plants

RESULTS AND DISCUSSION

Design engineers accepted the wetland planning concepts when well founded and situated in a familiar engineering format. Initial planning with the support of a biological basis insured a sound foundation. Acceptance by the design engineer was obtained when wetland planning adequately addressed overall program goals and plans. Within the Navy, this also includes addressing the military mission. Further acceptance was achieved when wetland planning considered the need to minimize costs of site development and earthwork. Finally, wetland planning also considered construction contractors' needs such as the logistics of access ways and haul routes, equipment maintenance and storage, and clarity in design.

Using the information developed in the Wetland Creation Workshop, the Navy has begun to develop a large-scale tidal saltmarsh project. The single plan drawing of the wetland was placed in two construction contracts for use during borrow operations. As of this writing, the large-scale wetland has been approximately half excavated with plans to continue excavation during construction of future facilities. Wetland planting plans and specifications have been developed and await inclusion in future construction contracts.

Knowledge gained during preparation and installation of salt water wetlands has supported the planning for fresh water mitigative wetlands. Complete plans and specifications for two large-scale freshwater wetlands have been completed. One of these is now under construction. The site, a former shallow borrow pit has been partially excavated to specified contours during road construction. Construction of waterfront facilities also includes this fresh water mitigative wetland as a borrow site. Plans and specifications for vegetative plantings at these fresh water wetlands in accordance with EIS commitments are complete and ready for inclusion in future contracts.

The cost of excavating and rough grading the mitigative wetlands has been borne by the cost requirement of earthen fill for the construction projects. Since earthwork for construction is a necessary requirement of that project, the cost of wetland mitigation is incrementally reduced. Based on cost estimates provided in the literature, a comparison of the management approach utilized at Kings Bay with the more common after-the-fact wetlands approach showed a near order-of-magnitude reduction in cost when our approach is employed in wetland creation.

Decisions made during the planning process were also evaluated during construction. Cursory observations of the relative success of vegetative plantings in silty soils without fertilization versus sandy soils with fertilization suggest those planning decisions to be accurate. In one case, *S. alterniflora* planted to enhance an excavation into sandy subsoils received approximately 200 lbs. N per acre (slow-release) placed in the planting hole. At another site, *S. alterniflora* planted in silty marsh soils without fertilization produced a plant cover that was not different in appearance from the sandy subsoil site. Both planting sites were successful.

Contract specifications have been refined to accommodate unforeseen difficulties experienced by contract administrators. Specifying standards to facilitate measurement of contractor compliance was one aspect addressed only after observing difficulty in determining whether the correct number of plants had been actually planted. Simply measuring several inter-plant distances was adequate to calculate an average distance. The average inter-plant distance combined with evidence of area coverage was sufficient for a determination of contract compliance.

Decisions regarding the location of transplant donor sites were based on the personal experience of Navy environmental scientists and confirmed during construction. Still other decisions regarding such diverse issues as haul routes and transplant storage were made during planning and later evaluated during construction.

CONCLUSIONS

Implementable and Coordinated

Within the larger development program, integration of a strong environmental presence during design and construction can result in cost-effective and environmentally-sensitive wetland mitigation.

As shown by experiences during the development at SUBASE, cost-effective and coordinated wetland mitigation projects can be successful when implementable. Implementability results from a balance of overall program or mission goals with engineering and environmental needs in a multidisciplinary effort. Integration of the environmental presence in program execution is further supported by effective coordination between scientists and engineers during design and construction. Successful implementation of mitigation projects in this manner requires a sound scientific basis with a sensitivity to the ecology of impact mitigation and biology of wetland plant species.

Mitigation projects must also be coordinated with development master planning. Concerted efforts must be made to include mitigation projects within the overall development scheme rather than as follow-on projects. Thus, wetland mitigation and construction efforts may be streamlined to take advantage of one another. Federal programs should also be coordinated with EIS constraints and commitments. A knowledge of the basis of these items and how to implement them is a necessity.

Individual mitigation projects must be set in a format suitable for acceptance by the engineering community. Drawings must include registered professional engineer signatures as well as incorporate acceptable styles. Drawings should also show adequate horizontal and vertical control and adequate detail for successful execution. Finally, in federal and other program which involve competitive procurement, design must be adequate to allow contractor bidding.

The broad management approach employed during construction at SUBASE discussed in this report constitutes a more cost-effective and comprehensive mitigation program than a more isolated individual mitigation project approach. The apparent success of this program, including the high level of acceptance by design engineers and project managers, suggests that use of the described management approach can lead to an increase in land use efficiency.

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DESIGN OF A SUBMERGED AQUATIC OFF-SHORE WETLAND FOR DISPOSAL OF TREATED WASTEWATER EFFLUENT IN CHESAPEAKE BAY

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ABSTRACT

A $56-million water reclamation scheme for the Mayo Peninsula in Maryland will use alternative and on-site wastewater treatment methods. Included are individual septic systems, cluster leaching fields, sand filters, freshwater emergent wetland, ultraviolet disinfection, peat wetland, and off-shore wetland. The man-made off-shore submerged aquatic vegetation wetlands will include <u>Potamogeton pectinatus</u>, <u>Potamogeton perfoliatus</u>, and <u>Ruppia maritima</u>.

Construction of the on-site and cluster systems began in August 1986. Construction on the communal facility, which includes the off-shore wetland, was initiated in August 1987.

INTRODUCTION

Last summer's construction start on an award-winning $56-million water reclamation plan for the Mayo Peninsula in Maryland ends over 20 years of public debate and resistance to previous wastewater plans. It also will correct serious wastewater and public health problems that have plagued the 20.7 sq. km (8-sq.-mile) peninsula, located south of Annapolis. Several previous attempts to develop a wastewater plan for the peninsula were rejected by county officials and area residents who have opposed growth-promoting, centralized wastewater schemes.

Since 1980, the county has been under state order to correct serious wastewater and public health problems on the peninsula, which juts into Chesapeake Bay (Figure 1). Wastewater service on the peninsula is presently provided by individual on-site wastewater disposal systems. Fifty-eight percent of the area's residents have reported septic system problems, poor surface drainage, flooding, and adverse well water quality. The principal manifestation of the high rate of septic system failure on the peninsula was nitrate and coliform bacteria contamination of groundwater which is the primary source of drinking water for residents of the peninsula. The county was looking for a cost-effective, decentralized wastewater management solution that would solve these problems and that would be accepted by area residents who had vocal growth management concerns.

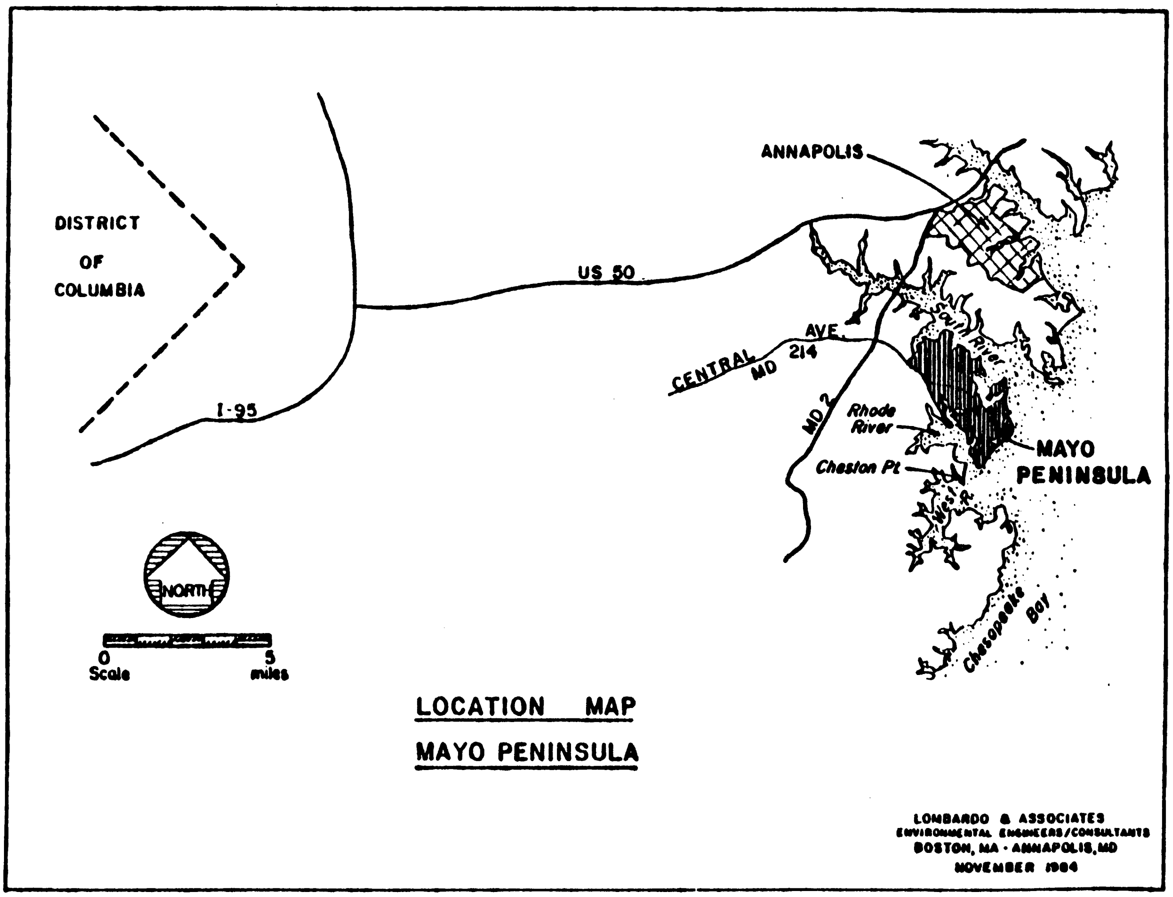

Figure 1. Location map--Mayo Peninsula

The current plan won the approval of area residents and various local, state, and federal environmental agencies because it was carefully tailored to local conditions. The plan does not subsidize growth or promote sewering undeveloped areas, but provides for orderly growth; it saves money as compared to previous plans; and it integrated on-site and innovative "natural" treatment processes to protect the health of residents and the fragile ecosystem of the peninsula.

The approved plan integrates three treatment approaches under a special Mayo Water Reclamation Subdistrict (MWRS) set up by the Anne Arundel County Department of Utilities (AACDU): on-site septic systems, cluster soil absorption systems, and a communal treatment system as shown in Figure 2.

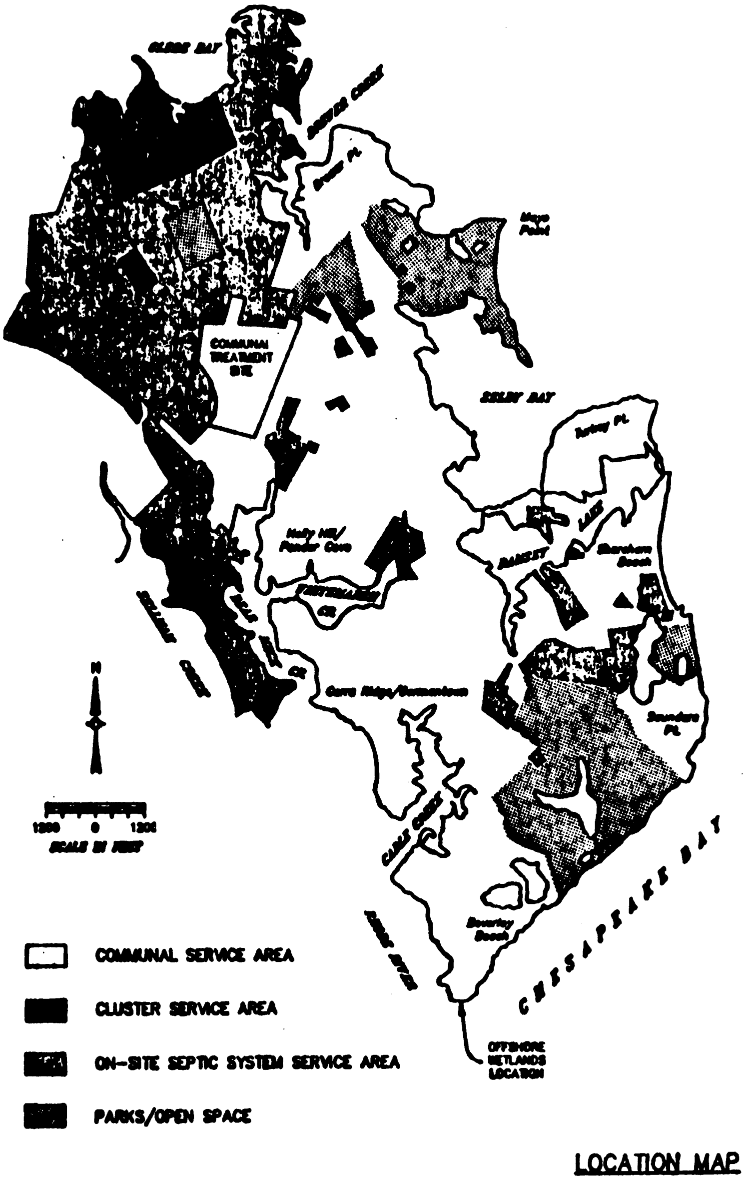

Figure 2. Mayo service area map.

THE TREATMENT SYSTEM

Individual Septic Systems

In areas of the peninsula where soil and groundwater conditions allow, existing problem on-site septic systems will be repaired and all on-site systems will be maintained by the MWRS (Figure 3). As designed, the MWRS will be responsible for over 100 individual septic systems. This is the first time in the U.S. that a major public utility will manage, finance, and operate individual septic systems as part of an overall wastewater management system.

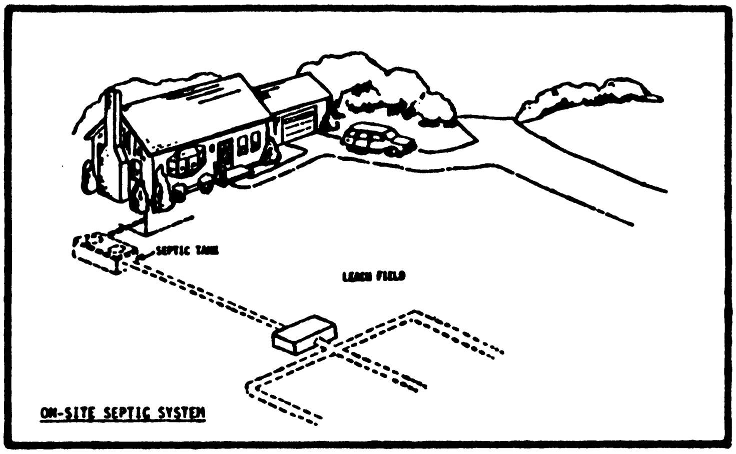

Figure 3. On-site systems.

Cluster Leaching Fields

In the northern part of the peninsula, a cluster leaching field will purify septic tank effluent from clusters of homes (Figure 4). The system will also be maintained by the county.

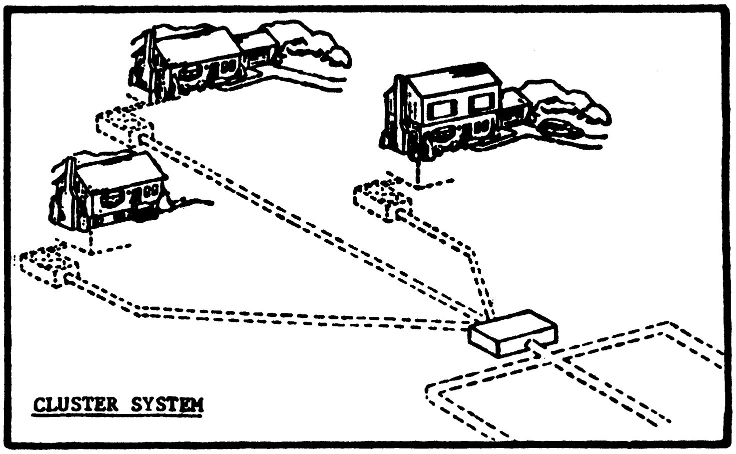

Figure 4. Cluster system

Communal System

The final component in this plan is a five-step, communal treatment system that will treat effluent from a septic tank/effluent collection system. The communal system has been designed to treat an existing wastewater flow of 1.58×10^8 liters/day (418,000 gpd) and a projected 20-year flow of 2.97×10^8 liters/day (768,000 gpd). The treatment system (Figure 5) consists of recirculating sand filters, man-made bulrush/cattail wetlands, ultraviolet disinfection, man-made peat wetlands, and, after a final, precautionary disinfection step, discharge into constructed off-shore submerged aquatic vegetation wetlands. Each component of the communal system is described below.

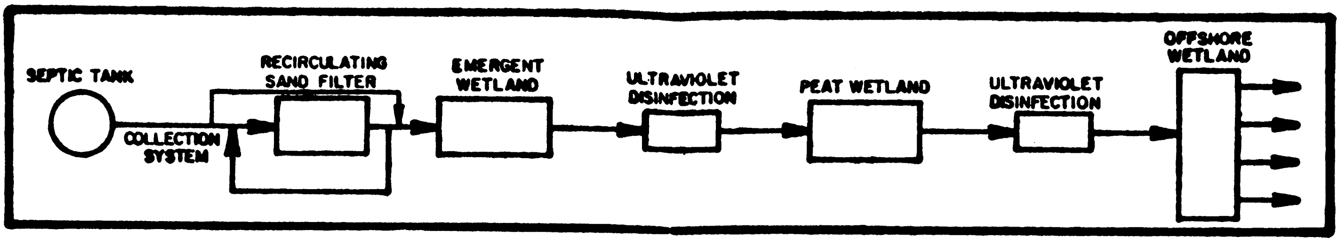

Figure 5. Communal treatment system process flow diagram.

SAND FILTERS

Although sand filters have been used for wastewater treatment for decades, they are currently undergoing a resurgence in popularity. The recirculating sand filter, developed in the 1970s, is an improved, more efficient sand filter type.

Sand filters (Figure 6) consist of beds of sand approximately two feet thick. Underneath the sand bed is a layer of gravel containing perforated collection pipes. In the treatment system, effluent will be evenly distributed over the bed of sand, which purifies the wastewater through a combination of physical and biological means. As the name implies, the recirculating sand filter recycles some of the treated effluent back to the recirculating tank where it is mixed with the untreated septic tank effluent. Approximately seven acres of sand filters will be needed to treat effluent from the peninsula's current residents and commercial establishments.

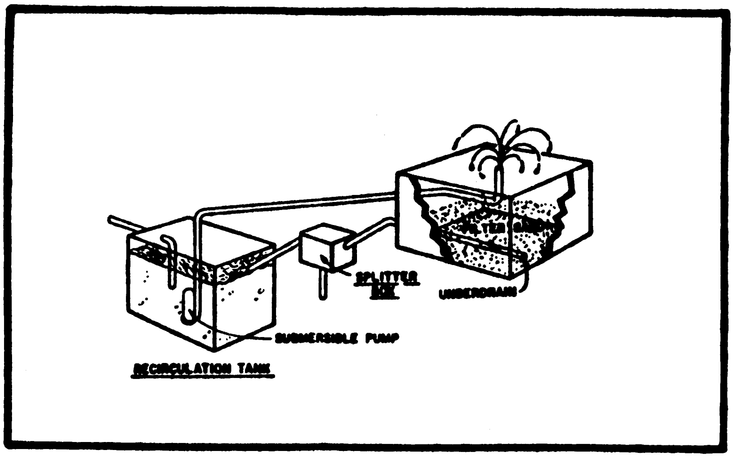

Figure 6. Recirculating sand filters.

FRESHWATER EMERGENT WETLAND

Filtered effluent will flow by gravity into a man-made 28,329 sq. meter (7.0-acre) freshwater emergent wetland (Figure 7). Wetland will consist of a constructed basin lined with 30 mil PVC liner to prevent interaction of effluent with groundwater and filled with stone as a planting base. The emergent wetland will be planted with *Typha latifolia* (cattail) and *Scirpus olneyi* (bulrush).

While this freshwater wetland will serve as a back-up to the sand filters for BOD_5 and suspended solids removal, its main function will be for nitrogen removal through denitrification. By mixing septic tank wastewater with sand filtered effluent, a carbon source will be provided to fuel denitrification within the emergent wetland.

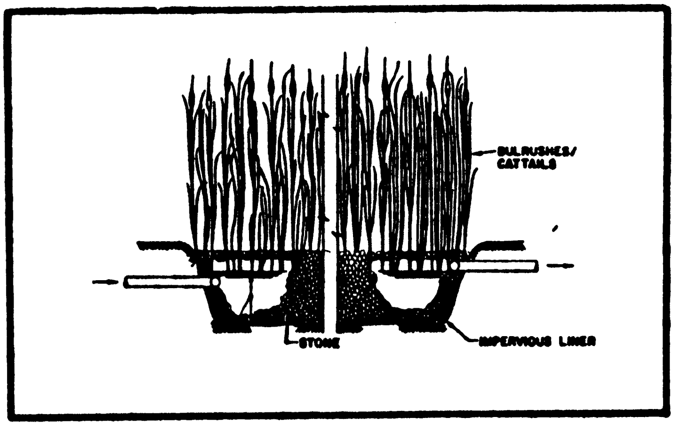

Figure 7. Emergent wetland.

ULTRAVIOLET DISINFECTION

From the freshwater wetland, the flow will pass through an ultraviolet disinfection chamber (Figure 8) to reduce pathogens.

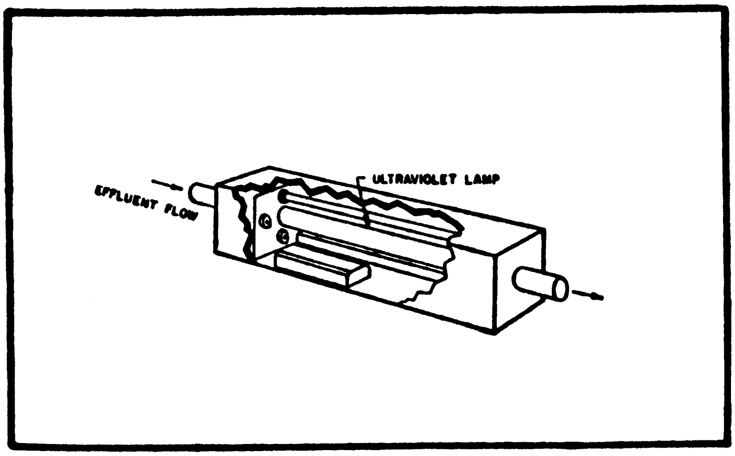

Figure 8. Ultraviolet disinfection.

PEAT WETLAND

For phosphorus removal, the effluent will be applied to a man-made peat wetland. The peat wetland (Figure 9) is designed primarily for phosphorus removal by physical/chemical absorption and microbial immobilization, with additional removal capability for nitrogen and other wastewater constituents. The wastewater will be applied to the surface of the peat by a sprinkler system. The wastewater will then percolate through the peat layer where the majority of the phosphorous removal will take place. The treated wastewater is then collected in an underdrain system.

The surface of the 33,945 sq. m (8.4-acre) peat bed will be seeded with grass. The uptake of nutrients by the vegetation aids in the overall performance of the peat wetland. The peat wetland was chosen over more conventional chemical addition/precipitation type phosphorous removal systems due to its high performance level and its low operation and maintenance requirements.

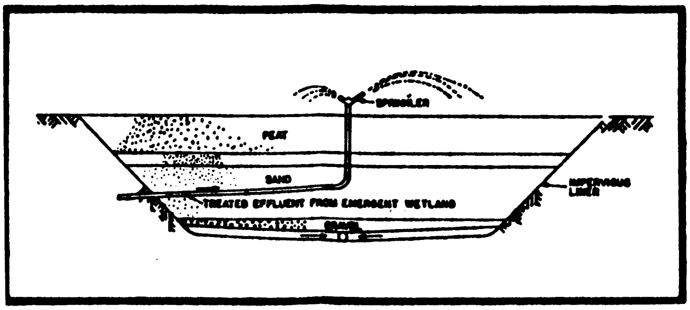

Figure 9. Peat wetland.

OFFSHORE WETLAND

After a final precautionary disinfection step, the treated and disinfected effluent will be transported to the southernmost tip of the Mayo Peninsula, off Dutchman's Point, where it will be discharged into constructed off-shore submerged aquatic vegetation wetlands, 42.67 m (140 feet) into the Rhode River (Figure 10). The main function of the off-shore wetland is to disperse the treatment system effluent. Additional nitrogen and phosphorus removal by the off-shore wetland is considered a bonus of this cost saving alternative to a Chesapeake Bay deep-water outfall. (The elimination of the outfall is estimated to save about $12 million in construction costs.)

Included as part of the off-shore wetland is a discharge pipe, diffuser pipes, submerged aquatic vegetation, a wooden trestle and catwalk, and various stone structures. Submerged aquatic vegetation to be planted in the off-shore wetland consists of *Potamogeton pectinatus* (sago pondweed), *Potamogeton perfoliatus* (redhead grass), and *Ruppia maritima* (widgeon grass) transplants. All of these submerged plants are presently, or were at one time, indigenous to the brackish waters of the Rhode River. The plants will be grown in a water depth of 0.6 to 0.9 m (2 to 3 ft.). The total off-shore planting area is 1728 sq. m (18,600 sq. ft.).

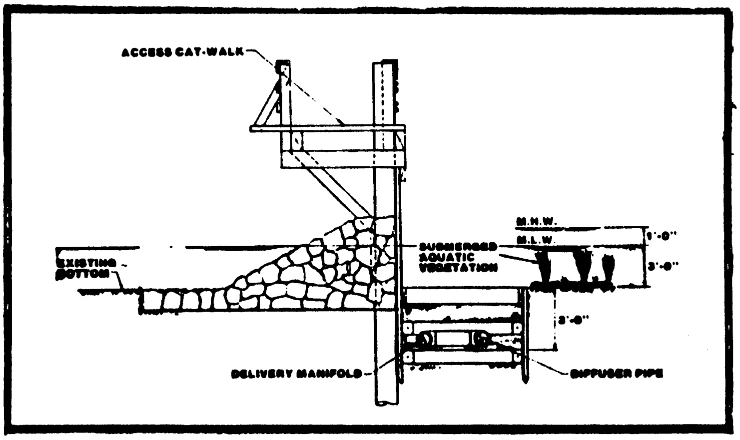

Figure 10. Off-shore wetland.

The treated effluent will enter the wetland through a perforated header located in a trench at the edge of the wetland. The velocity of the jets of effluent leaving the holes in the header create turbulent mixing with the surrounding water, thereby diluting the effluent.

The design of the off-shore wetland takes advantage of the density differential between the treated effluent and the saline receiving waters of the Rhode River for effective dilution. The treated effluent entering the wetland will be of a lower density than the receiving river water. As a result, a two-layer stratified flow will develop out of the wetland with the treated effluent near the surface and the receiving water on the bottom. The density differential also causes vertical mixing action, further diluting the wastewater. As a result of this mixing, the effluent can be diluted to approximately 30:1 at the point it exits the off-shore wetland. Additional dispersion will take place as the coastal wetland water mixes into the Rhode River/ Chesapeake Bay. The tidal currents in the river will disperse the treated effluent and result in a series of dilution contours shown in Figure 11.

A hydrodynamic study identified three potential sites for the off-shore wetland. The decision to designate Dutchman Point as the preferred site for the off-shore wetland was, in large part, based on a report by G. Han (1975). Han identified several features of the Rhode River which made it favorable for coastal wetland siting. The report notes the presence of enhanced vertical mixing in the river relative to the open bay. It also indicated that the average tidal current velocity of 5 cm/sec in the vicinity of the site combined with the tidal flushing flow of the river of approximately 1.89×10^9 liters/day

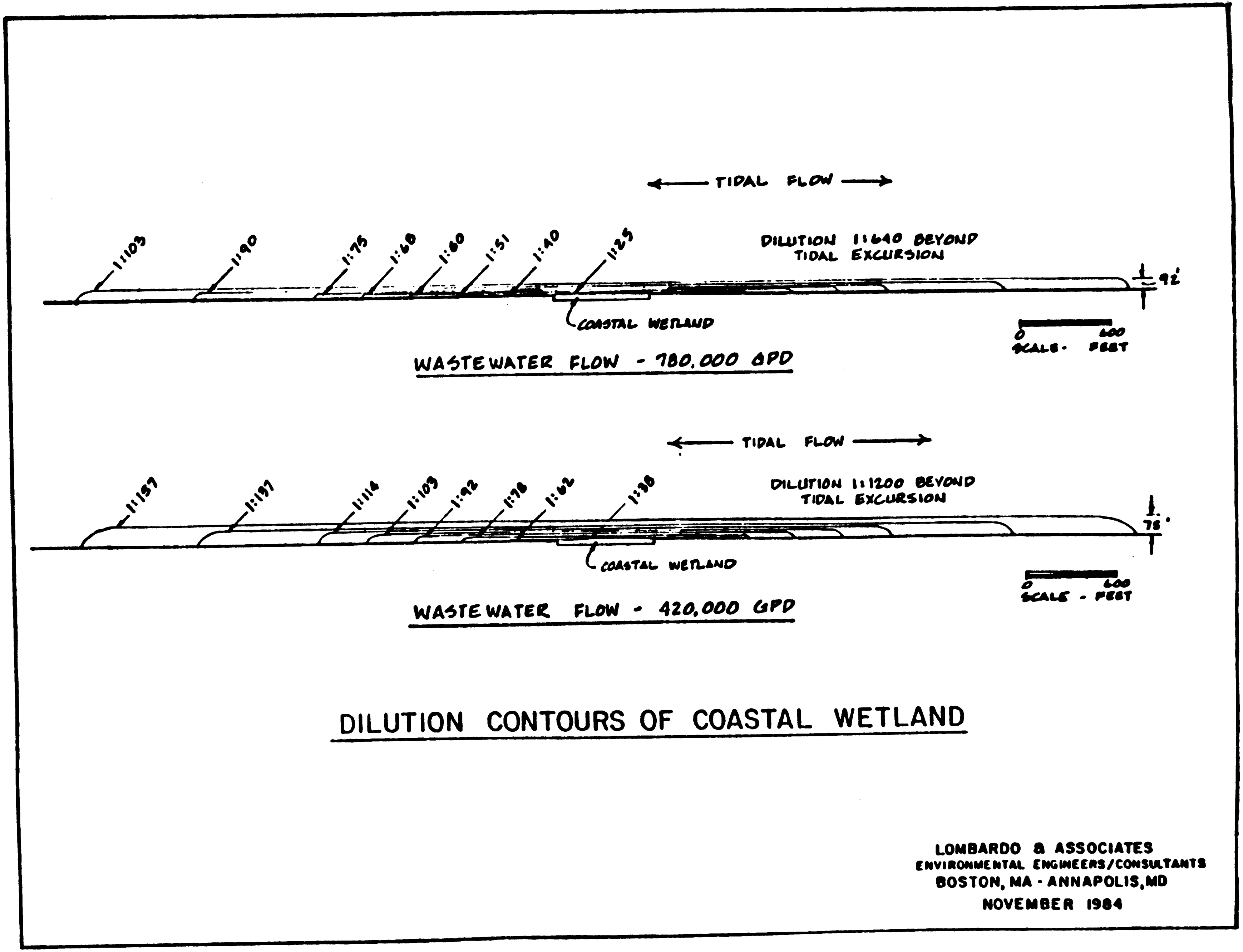

Figure 11. Dilution contours of coastal wetland.

(500 mgd) will result in a dilution level of approximately 1000:1, beyond the tidal excursion zone of influence in the wetland, for the present estimated flow of 1.58 x 10^8 liters/day (418,000 gpd) and approximately 600:1 for the projected 20-year flow of 2.97 x 10^8 liters/day (786,000 gpd).

The off-shore wetland structure has been designed to require minimal maintenance. Stone jetties and groins are provided to protect the off-shore wetland from damage due to wave action. Clean outs have been provided along the on-shore portion of the effluent discharge pipe to provide access for maintenance and flushing. Diffuser piping has been provided with valves to allow isolation of various sections of the diffuser. Underwater structures will be inspected periodically during and following storm events. This includes cleaning barnacles and other growth off of the timbers, repairing any damaged timbers or pipes and inspection of stone structures.

FINANCING THE SYSTEM

Eighty percent of the initial capital costs will be funded by federal and state grant programs. The remainder will be financed by the MWRS.

Because of the unique nature of the Mayo Peninsula wastewater management plan, a separate entity within the AACDU, the Water Reclamation Subdistrict, was set up to finance and manage the system. The revenues collected from customers in the MWRS will support the full cost of construction, rehabilitation, operation, management, and maintenance of wastewater facilities.

The user charge system is structured to recover about one-third of the local share of the initial capital costs at the start of service, and the balance spread over a 30-year period and recovered through quarterly payments by the customer base. User charges include normal annual operations and maintenance, plus future system replacement costs. Thereby, the financial integrity of the system will be maintained.

Future customers coming into the management area would pay the full cost of constructing the necessary wastewater facilities required to provide service. For the communal system, it is estimated that this would amount to about $15,000 per future residential user because of the lack of federal and state grant assistance beyond the initial construction stage of the project.

Customers on the communal systems (including cluster systems) would pay one uniform charge, and all customers with on-site service would pay another uniform, but lower, charge. Future on-site and communal customers, and owners of undeveloped land, will pay a lower annual charge that represents their proportionate share of current facilities that will serve their future needs.

As part of the annual user fee, all users will contribute to a capital fund which will be used to finance future repairs to on-site and communal systems.

The user fee for the communal systems is $413 per year. The on-site systems user charge is $229 per year. A one-time connection for the communal system is estimated at $1098, and $595 for on-site systems that are repaired. No connection charge is assessed to on-site users whose systems are salvaged. Currently, other residents of the county pay a connection charge of $3,625 and a yearly user fee of $500 if they are tied into existing facilities.

IMPACT

The Mayo Peninsula Wastewater Management Plan will correct the existing wastewater disposal and public health problems on the Mayo Peninsula utilizing three wastewater management techniques; on-site wastewater systems, cluster systems, and a communal treatment system. The communal treatment system will discharge, via the off-shore wetland, tertiary quality effluent to the Rhode River, an embayment of Chesapeake Bay. Effluent will be diluted up to thirty times at the point of exit from the off-shore wetland.

The plan has also provided the institutional framework designed to manage the on-site, cluster and communal treatment systems and taken into account critical growth management and environmental protection concerns that had blocked previous plans.

In recognition of its engineering excellence, the plan received a 1986 National Honor Award from the American Consulting Engineers Council.

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TREATMENT OF NATIONAL MARINE FISHERIES SERVICE RECOMMENDATIONS BY THE CORPS OF ENGINEERS IN THE SOUTHEAST REGION OF THE UNITED STATES FROM 1981 THROUGH 1985

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ABSTRACT

The seven coastal Corps of Engineers (COE) districts in the southeastern region of the United States manage a program which every year regulates development in thousands of acres of wetlands. The degree to which wetlands are conserved by this program depends largely on conservation recommendations of federal and state environmental agencies and the consideration given to these recommendations by the COE as part of their public interest determination. This determination dictates whether or not a permit will be granted.

The National Marine Fisheries Service (NMFS) has five years of data which track the degree to which the COE has included NMFS habitat conservation recommendations in their decision to grant permits. These data demonstrate considerable variation in the treatment of NMFS conservation recommendations among the seven COE districts surveyed. The effect in terms of acres of wetlands permitted by the COE over NMFS objections also is provided.

INTRODUCTION

In 1981, the National Marine Fisheries Service (NMFS), Southeast Region, initiated a computerized database which tracks actions related to permit applications submitted to the Corps of Engineers (COE) for permission to alter wetlands that come under the COE's regulatory authorities. This process is described in detail in Lindall and Thayer (1982) and Mager and Thayer (1986). The database contains information on the kind of project requested, its location, NMFS comments on permit requests, kind and extent of wetland alterations requested, and the area of impact the NMFS did not oppose. Mitigation acreage also is recorded. These data provide a measure of potential cumulative wetland losses and gains as well as wetlands potentially conserved.

The NMFS also surveys projects where permits approving construction in wetlands have been issued by the COE. We track those projects on which we have accurate information on the area and kind of wetlands requested for alteration and the area that the COE actually permits. We also determined whether the COE accepted, partially accepted, or rejected NMFS's recommendations. The term partially accepted was used when some, but not all, of the recommendations in our letters to the COE on individual projects were included in issued permits. Accepted was used when all of our recommendations were included and rejected was used when none of our recommendations were included in issued permits.

NMFS habitat conservation activities in the Southeast Region have been quantified by Lindall and Thayer (1982), Mager and Thayer (1986), and Mager and Hardy (in press). From 1981 through 1985, 184,187 acres of coastal wetlands were proposed for alteration by 5,385 permit applications submitted to the COE. The NMFS recommended the conservation of 135,687 acres and requested mitigation by restoration and generation of 110,406 acres. An analysis of 857 issued permits revealed that COE districts in the Southeast Region accepted NMFS's recommendations only about 50% of the time. Recommendations were partially accepted and rejected 24% and 26% of the time, respectively.

Explanations for not including NMFS's recommendations in issued permits by some COE districts usually centered around complaints that biological assessments and recommendations were too restrictive, the difficulty associated with resolving issues with applicants, the small amount of wetlands affected by their decisions, and inadequate staffing. The purpose of this report is to compare the seven COE districts with which the three NMFS area offices deal most frequently to determine if there are variations in NMFS area office recommendations that would account for differences in the way these recommendations are treated. The types of NMFS comments and recommendations related to whether or not permits should be issued, modification or relocation to minimize environmental impact, restoration, generation, or enhancement of wetlands as mitigation, etc. The results are provided below.

RESULTS

Treatment of NMFS's recommendations by COE district and the acreages of wetlands permitted for alteration and mitigation are given in Table 1. The percent of recommendations fully accepted was highest in the Savannah District (89%), followed by the Wilmington District (87%), the Charleston District (80%), the Galveston District (56%), the Mobile District (48%), the New Orleans District (39%), and the Jacksonville District (20%). Percent partial acceptance of NMFS's recommendations was highest by the New Orleans District (58%), followed by the Galveston District (31%), the Mobile District (29%), the Jacksonville District (21%), the Wilmington District (9%), the Charleston District (7%), and the Savannah District (0%). Percent of NMFS's recommendations totally rejected was highest in the Jacksonville District (59%), followed by the Mobile District (24%), the Galveston and Charleston Districts (13%), the Savannah District (12%), the Wilmington District (6%), and the New Orleans District (3%). The combination of rejected and partially accepted was responsible for permitting for alteration by the COE 2,556 acres of wetlands (857 projects) over NMFS's objections (Table 1, column 5 minus column 6).

Table 1. Treatment of NMFS recommendations on permit applications by the Corps of Engineers from 1981 through 1985, by district.

COE district	N	NMFS comments accepted (1)	NMFS comments partially accepted (2)	NMFS comments rejected (3)	Acreage proposed by applicant (4)	Acreage accepted by NMFS (5)	Acreage permitted by COE (6)	Acreage R/G recom. (7)	Acreage R/G permitted (8)
New Orleans	120	47 (39%)	70 (58%)	3 (3%)	9552	7540 (79%)	7914 (83%)	53688	53456 (99%)
Galveston	144	80 (56%)	45 (31%)	19 (13%)	7273	625 (9%)	1895 (26%)	3031	3054 (101%)
Mobile	59	28 (48%)	17 (29%)	14 (24%)	164	33 (20%)	68 (41%)	15	47 (313%)
Jacksonville	268	54 (20%)	56 (21%)	158 (59%)	2050	703 (34%)	1492 (73%)	334	104 (31%)
Savannah	26	23 (89%)	0 (0%)	3 (12%)	423	13 (3%)	58 (14%)	44	16 (36%)
Charleston	121	97 (80%)	8 (7%)	16 (13%)	2379	88 (4%)	111 (5%)	21	24 (114%)
Wilmington	119	103 (87%)	9 (8%)	7 (6%)	213	60 (28%)	79 (37%)	59	59 (100%)
Total	857	432 (50%)	205 (24%)	220 (26%)	22054	9061 (41%)	11617 (53%)	57192	56760 (99%)

N refers to the number of permits sampled
Numbers in () for columns 1-3 are % of N
Numbers in () for columns 5-6 are % of column 4
Numbers in () for column 8 are % of column 7
R/G refers to wetland restoration and generation acreage

A measure of the effect that the rejection of NMFS's recommendations had on the area of wetlands potentially conserved is indicated in Table 1 and Figure 1. The percent difference by COE district in the amount of wetland alterations the NMFS did not oppose and the alterations the COE districts actually permitted is given in Figure 2. The Jacksonville District permitted 39% (800 acres) more habitat alterations than the NMFS recommended. Jacksonville was followed by the Mobile District (21%, 34 acres), the Galveston District (17%, 1,236 acres), the Savannah District (11%, 47 acres), the Wilmington District (9%, 19 acres), the New Orleans District (4%, 382 acres), and the Charleston District (1%, 24 acres). However, all COE districts except Jacksonville and Savannah issued permits requiring more mitigation acreage than the NMFS recommended (Table 1, Column 8). This resulted mainly because more applicants are proposing mitigation on their own as a result of pre-application meetings or post-application negotiations with the NMFS and other conservation agencies, or requirements of other regulatory agencies.

Each NMFS field office covers several COE districts as follows: the Beaufort, North Carolina Area Office covers the Wilmington, Charleston, and Savannah districts; the Panama City, Florida Area Office covers the Jacksonville and Mobile Districts; and the Galveston, Texas Area Office covers the Galveston and New Orleans Districts. Tabulation does not include the infrequent review of permit applications issued by the Vicksburg or Fort Worth Districts. Since the success of having NMFS's recommendations incorporated into issued permits ranged widely from 89% in the Savannah District to a low of 20% in the Jacksonville District, an analysis was made of 21,840 permit applications to determine if NMFS's recommendations also varied in conservativeness by Area Office (Table 2). While the number of projects reviewed varied by COE district, NMFS treatment (e.g., the level or significance of advice provided to the COE) as percent of the total reviewed, was remarkably consistent between the Area Offices (Figure 3). Accordingly, variations in treatment of NMFS recommendations were a result of differing regulatory implementation by various COE districts rather than variations in the advice that NMFS Area Offices provide on permit applications.

Overall, 84% of the permit applications reviewed by NMFS during 1981-1985 resulted in no objection and insufficient manpower responses or were letters agreeing with another agency's position (Table 2). These require little or no effort (Minimal Concern in Figure 3) by the COE in resolving issues between the NMFS and applicants. Less than 15% were of moderate concern. These actions included responses where NMFS requested that applications be modified and permits conditioned or mitigation provided to reduce environmental impact. It was understood by the COE (under a previous agreement with NMFS) that these actions would not be elevated for review by higher authority in the COE if NMFS recommendations were not followed. For only about 1% of the projects the NMFS reviewed did we request outright denial of permits or major modification (Maximum Concern). These are the activities which the NMFS may ask to be reviewed above the COE District Engineer level if

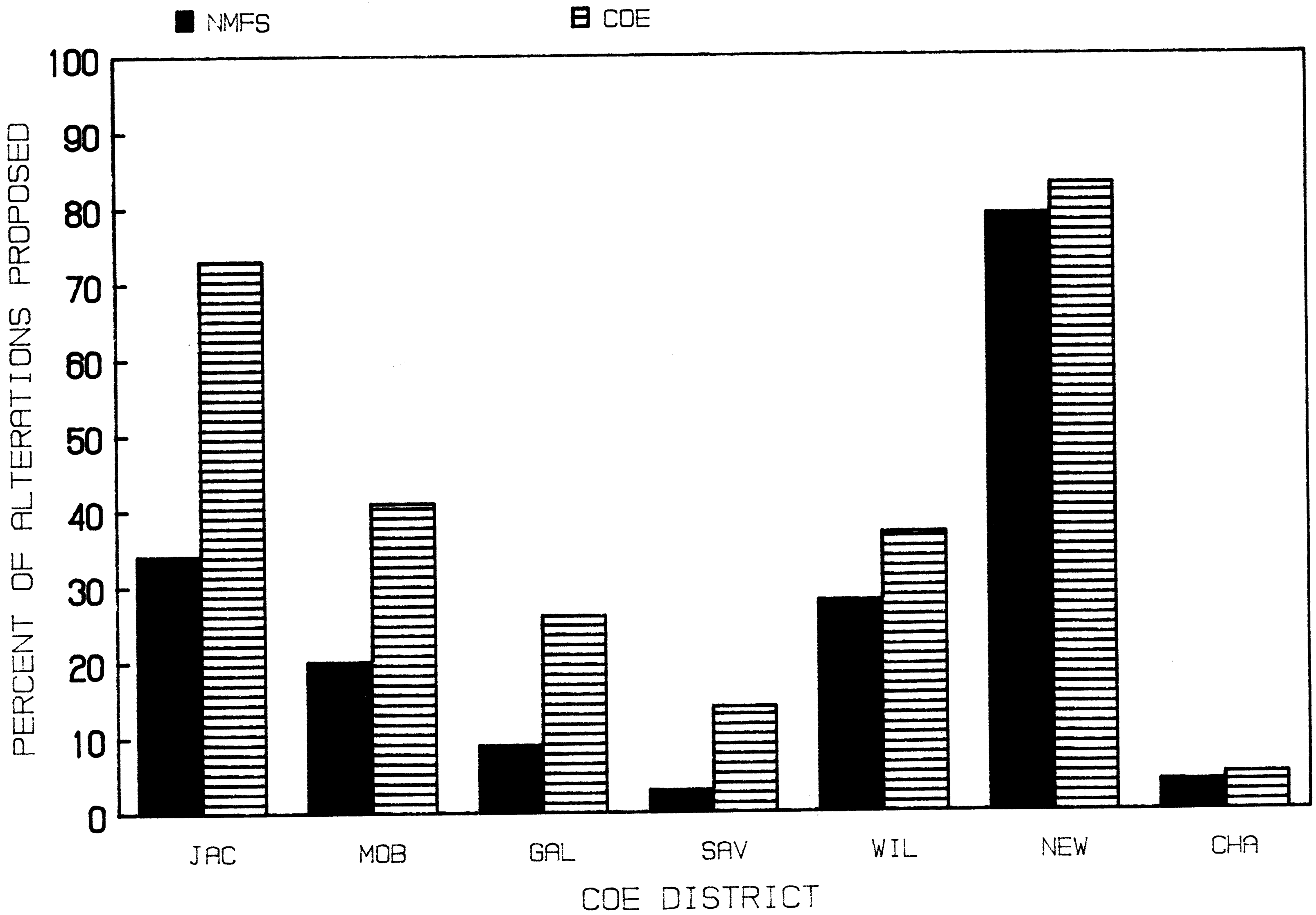

Figure 1. Wetland alterations accepted by NMFS and permitted by COE based on percent of acres proposed for alteration.

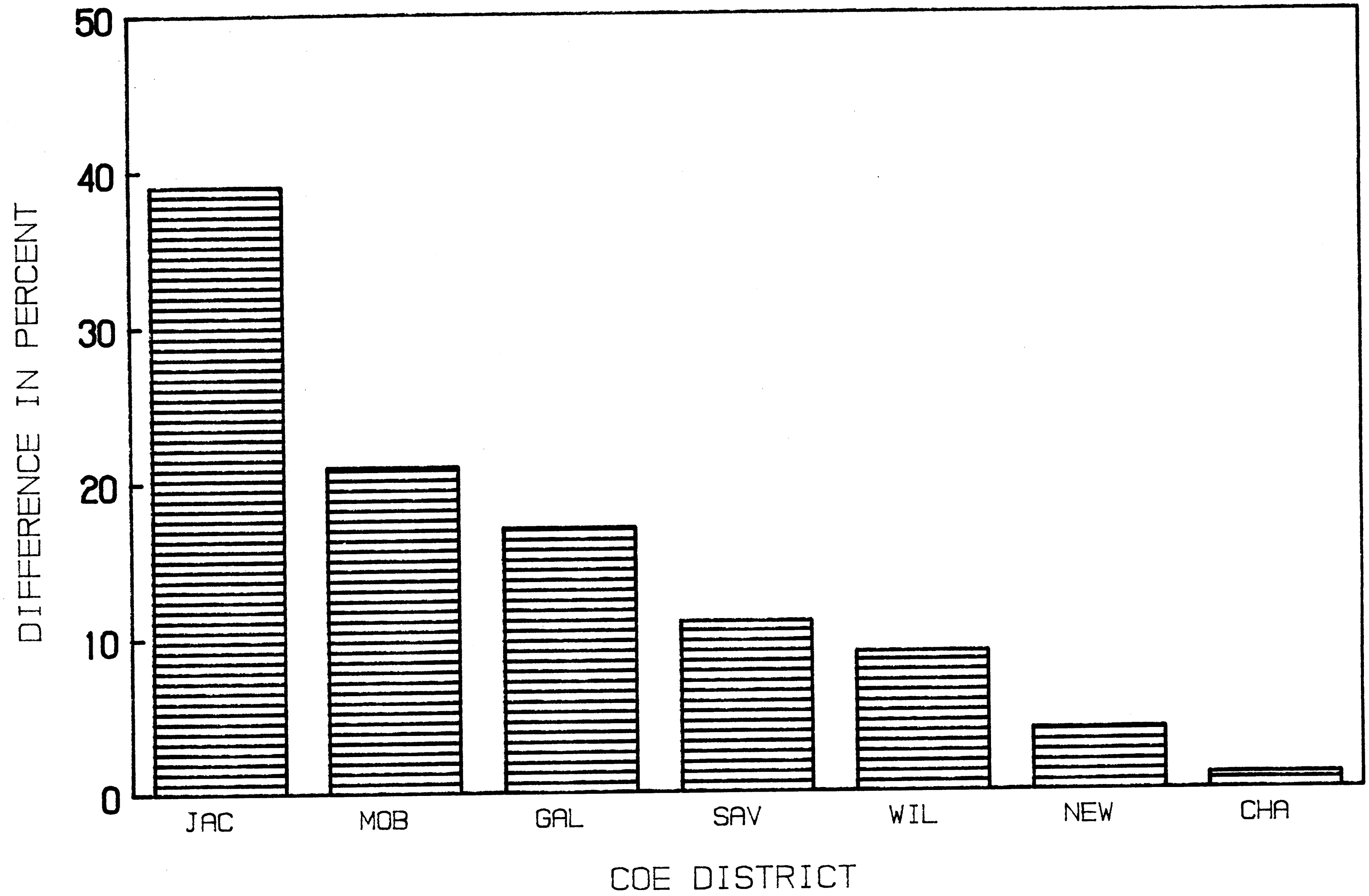

Figure 2. Difference between the percent of wetland alterations accepted by NMFS and permitted by the COE.

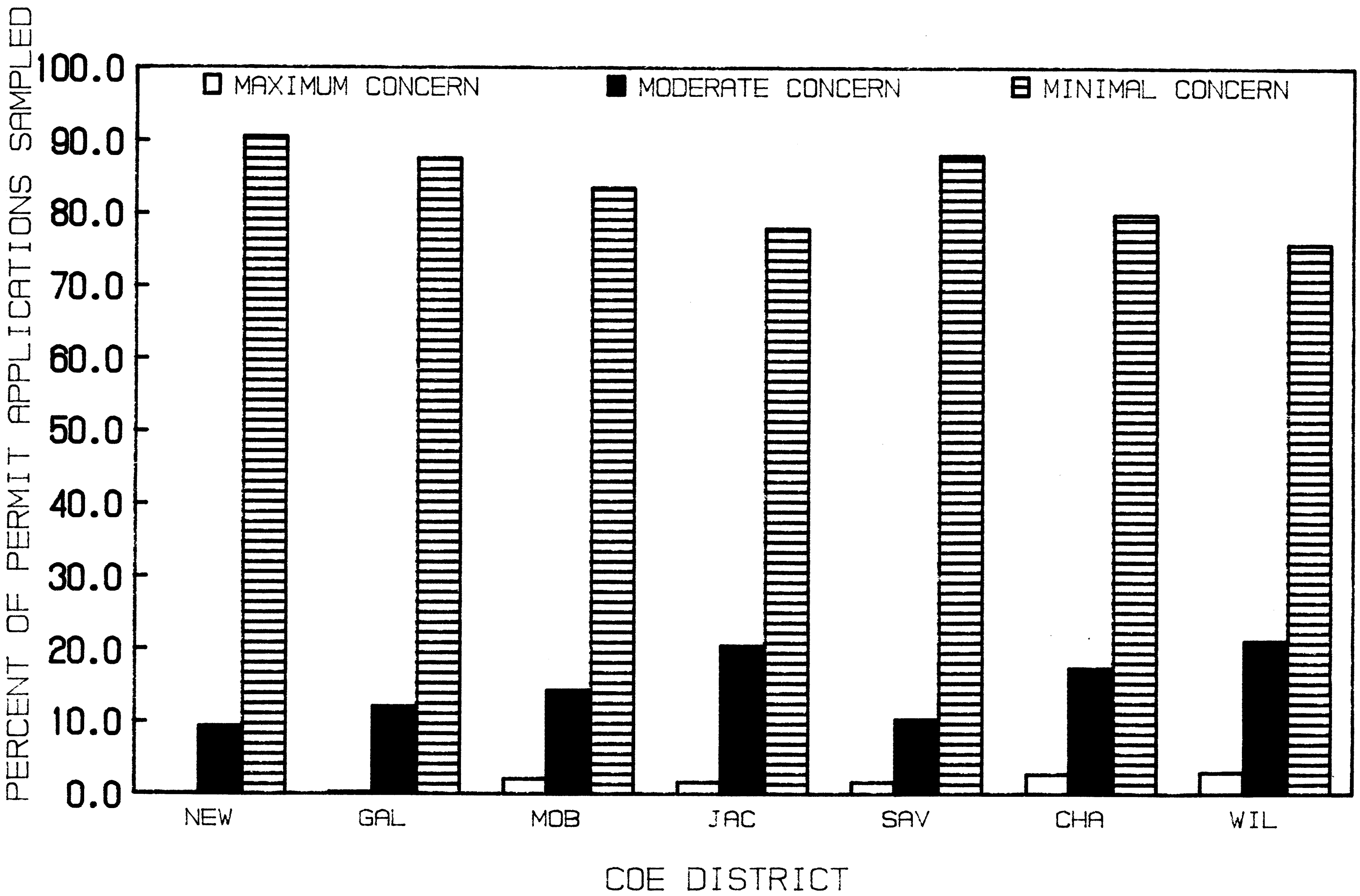

Figure 3. Analysis of NMFS responses to permit applications. NEW and GAL less than 0.5% maximum concern.

NMFS recommendations are overruled. However, since 1981, NMFS's Southeast Region has actually elevated fewer than five projects.

Table 2. Analysis of National Marine Fisheries Service level of concern on permit applications for 1981-1985.

COE District	N[1]	Minimal concern[2]	Moderate concern[3]	Maximum concern[4]
New Orleans	6,741	6,096 (90.5%)	635 (9.4%)	10 (0.1%)
Galveston	4,262	3,734 (87.6%)	515 (12.1%)	13 (0.3%)
Mobile	1,252	1,046 (83.5%)	179 (14.3%)	27 (2.2%)
Jacksonville	6,323	4,929 (77.9%)	1,289 (20.4%)	105 (1.7%)
Savannah	786	692 (88.0%)	81 (10.3%)	13 (1.7%)
Charleston	1,305	1,043 (79.9%)	225 (17.3%)	37 (2.8%)
Wilmington	1,171	888 (75.8%)	247 (21.1%)	36 (3.1%)
Total	21,840	18,428 (84.4%)	3,171 (14.5%)	241 (1.1%)

[1]refers to number of permit applications tracked
[2]refers to NMFS no objection and insufficient manpower responses and letters of coordination with other agencies
[3]refers to projects where NMFS recommended minor application modifications or conditioned permits
[4]refers to projects where NMFS recommended major modifications or permit denial

CONCLUSIONS

The three NMFS field offices have been relatively consistent in their recommendations on permit applications issued for review by the seven southeastern COE districts they interfaced with most often. However, treatment of NMFS recommendations by the COE varies considerably among those districts. Since NMFS recommendations were generally considered favorably by most of the COE districts, the lack of favorable consideration by the remaining districts appears to result from an inconsistent determination of the public interest rather than the way NMFS field offices comment on permit applications. A key factor may be that some COE districts do not give fish and wildlife production as well as other wetland functions adequate consideration in their public interest determinations.

It is evident that the potential cumulative loss of wetlands is large (184,187 acres for 1981-1985). This is considerable since less than 9.4 million acres of saltmarsh, freshmarsh, tidal flats, and swamps remain in the Southeast Region (Alexander et al. 1986). These remaining wetlands are vitally important to continued production of fish and other wetland-dependent wildlife as well as providing other public values. For example, Farber and Costanza (in press) determined that each acre of wetlands in Louisiana has a present value to society of roughly $2,500-$10,000/acre. Accordingly, the continued cumulative loss of these wetlands should be minimized by giving greater weight to wetlands values in the COE public interest reviews.

ACKNOWLEDGEMENTS

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WETLAND EVOLUTION IN MIDWESTERN RESERVOIRS

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ABSTRACT

Hundreds of reservoirs dot the midwestern landscape. Although most are less than fifty years old, many have extensive wetland systems. Builders, however, frequently plan to develop the shorelines of these impoundments for homesites and often include wetlands areas in their plans. When development activities affect wetlands, public interest might profit from carefully conceived plans which include mitigation and preservation.

To develop such plans, we must first understand what kinds of wetlands are best suited and, therefore, best to create for the reservoir systems with which we work. To that end, we have studied the natural establishment of wetlands in recently impounded reservoirs, their changes over time and the rate at which those changes have occurred. Observations suggest that wetlands establish quickly and spread continually in these systems.

Our research focuses on four central and southern Indiana reservoirs: Geist Reservoir, Morse Reservoir, Monroe Reservoir, and Patoka Reservoir, with varying ages: 44 years, 31 years, 22 years, and 9 years, respectively. The wetland vegetation at each location currently ranges between predominately herbaceous, both persistent and nonpersistent, deciduous shrub, and deciduous forest. Most of the information used to track the evolution of these wetlands came from historical records, principally aerial photographs, supplemented by site inspections to verify boundaries and check present conditions.

INTRODUCTION

Indiana has lost 80% of its historic wetlands, and most of those remaining are degraded (personal communication, Indiana Department of Natural Resources). The state's many reservoirs, however, provide areas where wetlands are expanding and serving as increasingly valuable fish and wildlife habitat. Yet, despite their apparent worth, reservoir wetlands face pressure from real estate developers who attempt to take advantage of high land values caused by the limited availability of lakeside property in Indiana.

By noting our observations on reservoir wetlands, we hope to assist regulators in understanding the potential value of these areas and help land developers in preparing mitigation proposals that involve creating wetlands which will mimic those that evolve naturally in reservoirs.

STUDY SITES

Figure 1. Locations of reservoirs included in this study.

The Indianapolis Water Company created both Geist Reservoir and Morse Reservoir to supply water for Indianapolis; Geist on Fall Creek in 1944, Morse on Cicero Creek in 1956. The two lakes are located in a region characterized by gently rolling or flat topography upon unconsolidated glacial deposits (Gray 1973). A recent state water resource survey rates the region's soils as having low to moderate erosive potential (Clark 1980). Streams within the watershed, however, carry considerable sediment. Agriculture, the dominant land use in both watersheds, may contribute to siltation of the streams. Geist Reservoir receives drainage from an area of 536 km^2. Morse has a watershed of 506 km^2.

Monroe Reservoir, the largest in Indiana, lies in a broad, flat valley among hills in the southern part of the state. The Corps of Engineers created it in 1965 for flood control. It also provides water for the City of Bloomington. Its upper basin is shallow and receives inputs from three major tributaries: the North, Middle, and South Forks of Salt Creek (Landers & Frey 1980). The reservoir's large watershed (1119 km^2) remains mostly untouched by glacial activity and is characterized by highly erodible soils on steep slopes, underlain by Mississippian siltstones. Agriculture in the watershed is largely restricted to bottomlands, less than 10% of the area. Forest covers over half the watershed (Landers & Frey 1980).

Patoka Reservoir also lies in a level valley among hills. The Corps of Engineers created it for flood control in 1978. Its upper basin holds only shallow water, except for the original Patoka River channel, and retains timber left standing after flooding. The reservoir drains a watershed of 383 km^2. The soils in the watershed are classified as highly erodible from their position on steep slopes and are underlain by siltstone and fine-grained sandstones (Gray 1973). Much of the watershed is forested (over 30%), although agriculture is the dominant land use (Clark 1980).

MATERIALS AND METHODS

We began our study with an assessment of aerial photos from the USDA Agricultural Stabilization and Conservation Service (ASCS) and the Indiana State Highway Department. Comparing the areal extent of wetlands, through diagrams and planimeter measurements, at different stages in the life of each reservoir allowed us to track the evolution and expansion of wetlands. Using aerial photos, however, presented some difficulties: compensating for differences in scale, finding photos taken during the growing season each year, and, since each photo represents only one point in time, determining what happened between photo dates. We, therefore, concentrated on identifying general trends in wetland evolution by combining information from aerial photos with data collected through ground surveys.

RESULTS AND DISCUSSION

Wetlands, like uplands, in the Midwest have a strong tendency to grow and evolve. We recognized two categories of wetland evolution in reservoirs; initial growth, and expansion. Initial growth occurs rapidly in newly flooded shallow zones early in the life of a reservoir. Expansion refers to the increase in wetland acreage that results from siltation.

Impoundment of streams can create large shallow areas. Our observations suggest shallow water zones are colonized quickly (within three years). Also, floodplain soils may hold the propagules of wetland plants deposited during past floods. Damming streams with broad, flat floodplains can, therefore, foster establishment of wide expanses of even-aged wetlands in only a few years. Rapid initial wetland growth in former floodplains occurred at both Monroe Reservoir

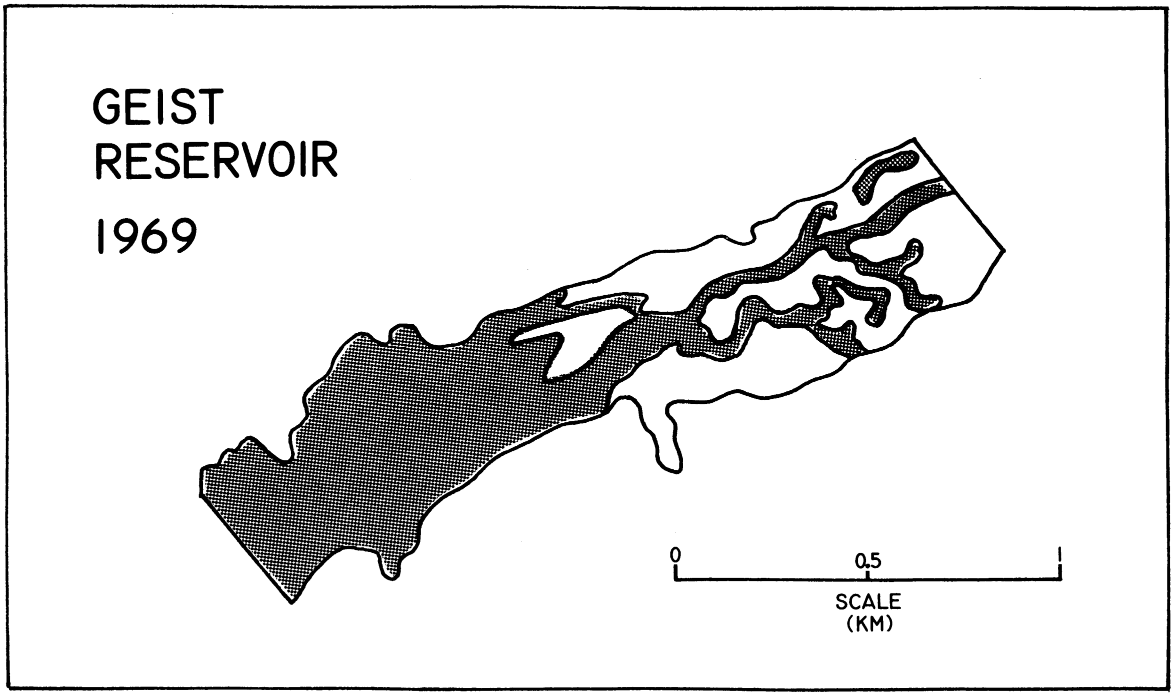
(59 ha in 3 years) and Patoka Reservoir (25 ha in 7 years).

Once existing shallow areas become fully vegetated, siltation along the wetland margins gradually reduces water depth and wetland expansion follows. As illustrated in the diagrams depicting Geist and Morse Reservoirs, and the more recent view of Monroe Reservoir, vegetation establishment proceeds more slowly in expansion than during initial growth. Wetlands at Geist expanded 5.5 ha in 17 years. Those at Morse expanded 5.7 ha in 25 years. After initial growth of 59 ha in three years, the wetlands at Monroe increased only 4.1 ha in the next 13 years.

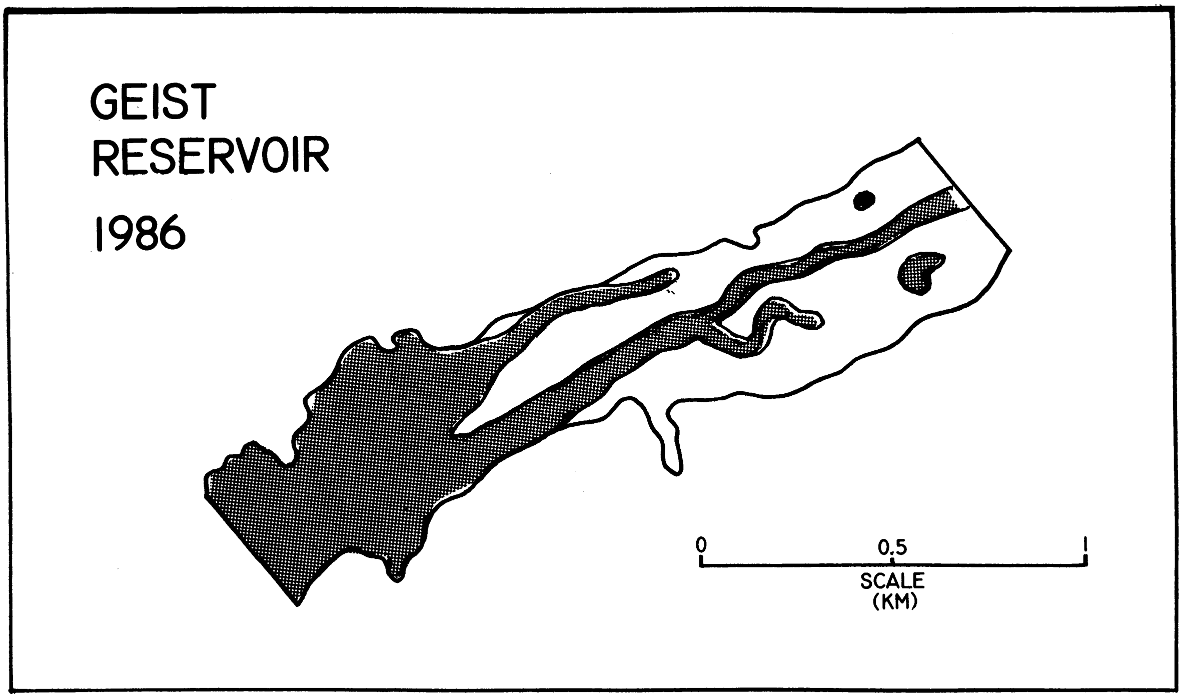
Wetlands established during initial growth show a recognizable pattern of change. They begin with mats of persistent herbaceous, emergent vegetation, usually reed canary grass (*Phalaris arundinacea*) mixed with cattail (*Typha latifolia*) and sedges (*Carex* sp.). The emergent mats that form are, during the growing season, fringed with bands of nonpersistent emergent plants--water smartweed (*Polygonum coccineum*) and water willow (*Justicia americana*)--and beds of submergent vegetation such as water milfoil (*Myriophyllum spicatum*) and coontail (*Ceratophylum demersum*). As silt and detritus accumulate among the plants and an increasingly stable substrate develops, water tolerant shrubs such as buttonbush (*Cephalanthus occidentalis*) and red osier dogwood (*Cornus stolonifera*), and trees, including black willow (*Salix nigra*) and cottonwood (*Populus deltoides*) begin to grow. Among the trees and shrubs, willow is favored by wetter areas (Teskey & Hinkley 1977) and usually dominates, although shrubs may eventually thrive as understory. Seedlings of silver maple (*Acer saccharinum*), Boxelder (*Acer negundo*), and sycamore (*Platanus occidentalis*) may establish at the same time as willow and cottonwood, but have slower growth rates. They can, however, continue to grow in the understory (Teskey & Hinkley 1977) and in some cases become the primary species. Silver maple dominates at Monroe Reservoir in areas that have supported wetlands for 20 years or longer.

Wetlands created by expansion show a similar sequence of succession, following siltation. Changes in water depth, however, determine the areal extent of plant growth, as well as the direction and rate of expansion. Substrates at depths of a meter or less (at normal pool level) usually support vegetation. Submergent vegetation (*Myriophyllum*, *Ceratophyllum*), the first plant type to establish during wetland expansion, grows in areas 70-100 cm deep. Nonpersistent emergents (*Polygonus*, *Justicia*) grow in slightly more shallow water, dominating in areas 30-70 cm deep. Persistent emergents (*Phalaris*, *Typha*, *Carex*) dominate in water less than 30 cm deep.

Prevailing conditions select the species most suited to colonize a shallow water zone (Van der Valk 1981). While it is relatively easy to predict where initial growth will occur, predicting the exact species that will develop, as well as the direction and rate of wetland expansion, may prove difficult. The pattern of water flow through the wetlands in these reservoirs and, hence, the pattern of siltation may



<u>Figure 2</u>. Section of Geist Reservoir in 1969 showing 39.4 ha of wetlands (unshaded areas).



<u>Figure 3</u>. Section of Geist Reservoir in 1986 showing 44.9 ha of wetlands.

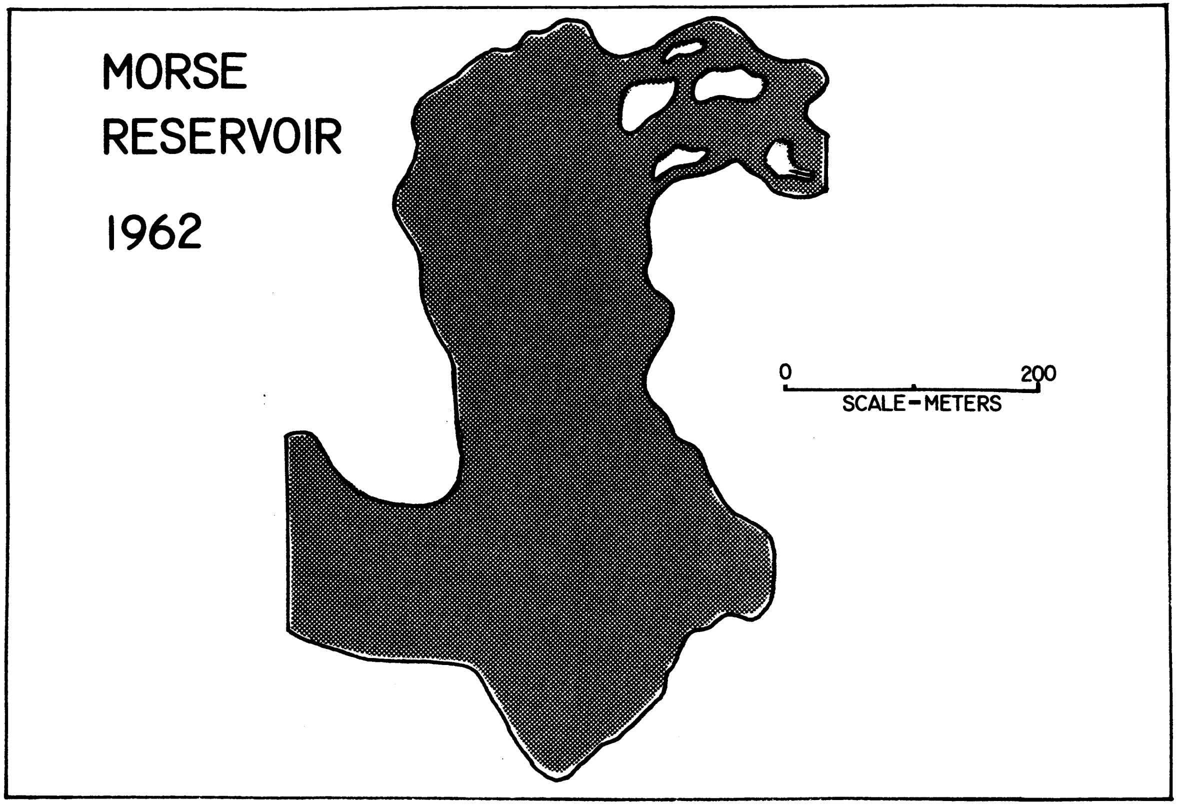


Figure 4. Section of Morse Reservoir in 1962 showing 0.4 ha of wetlands (unshaded areas).

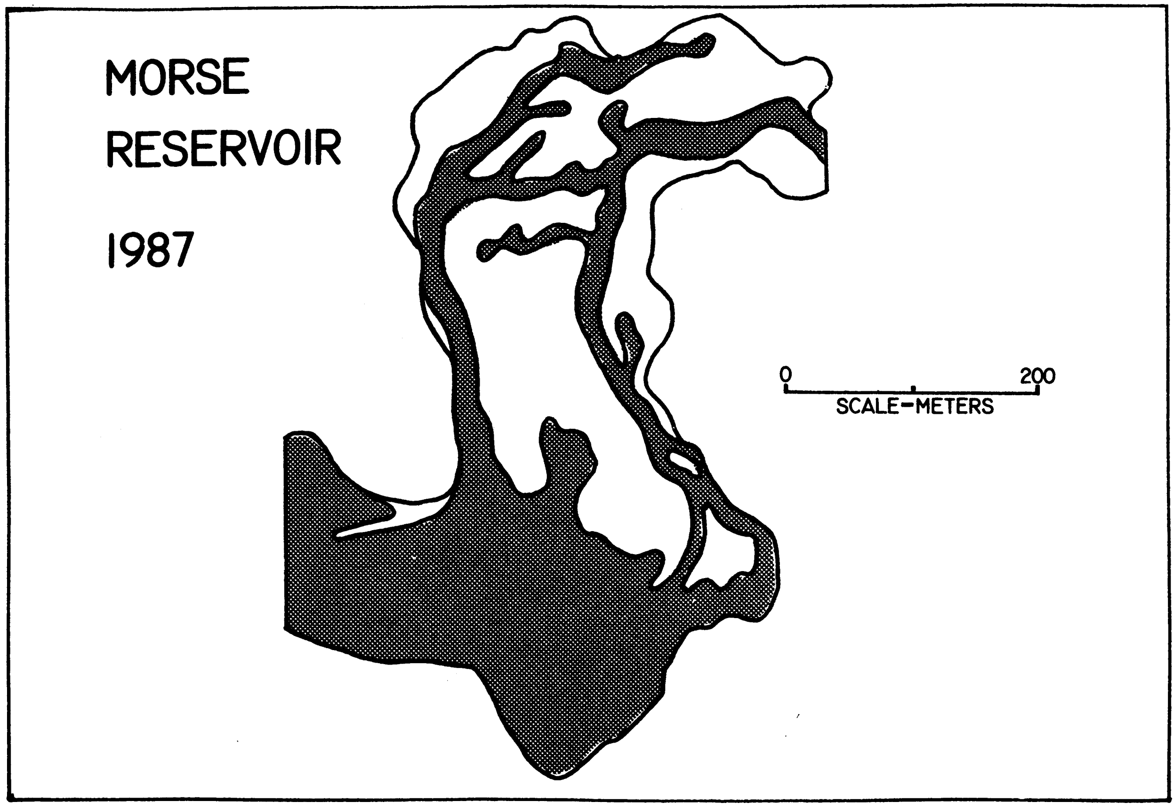


Figure 5. Section of Morse Reservoir in 1987 showing 6.2 ha of wetlands.

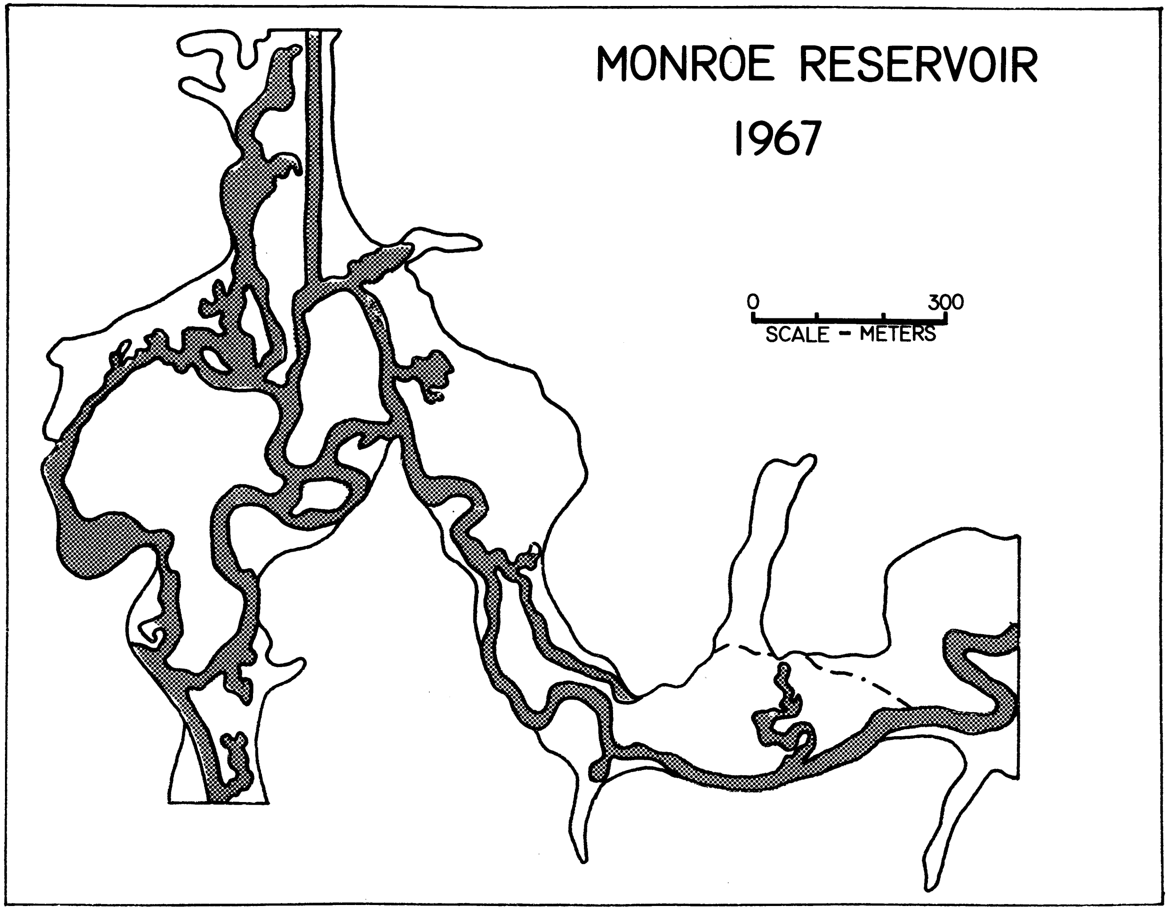


Figure 6. Section of Monroe Reservoir in 1967 showing 59.4 ha of wetlands (unshaded areas).

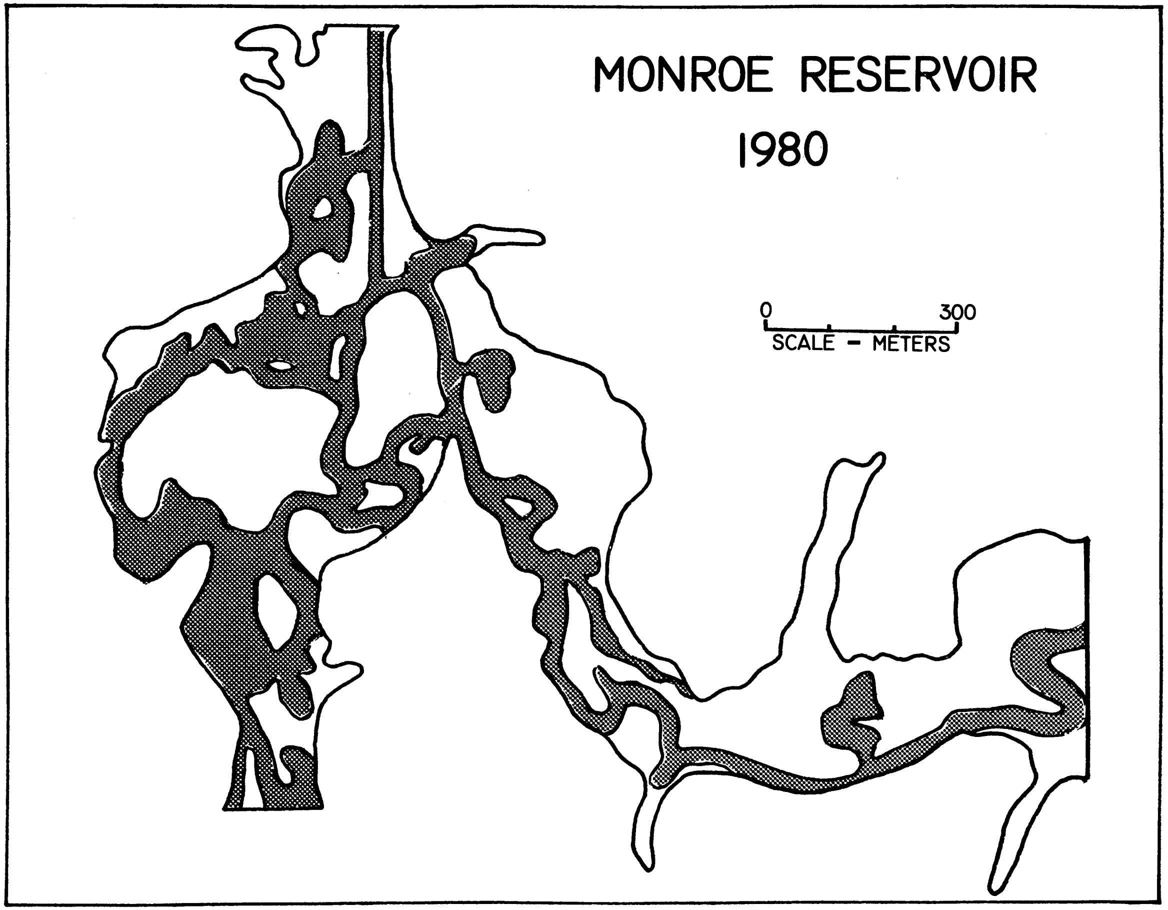


Figure 7. Section of Monroe Reservoir in 1980 showing 63.5 ha of wetlands.

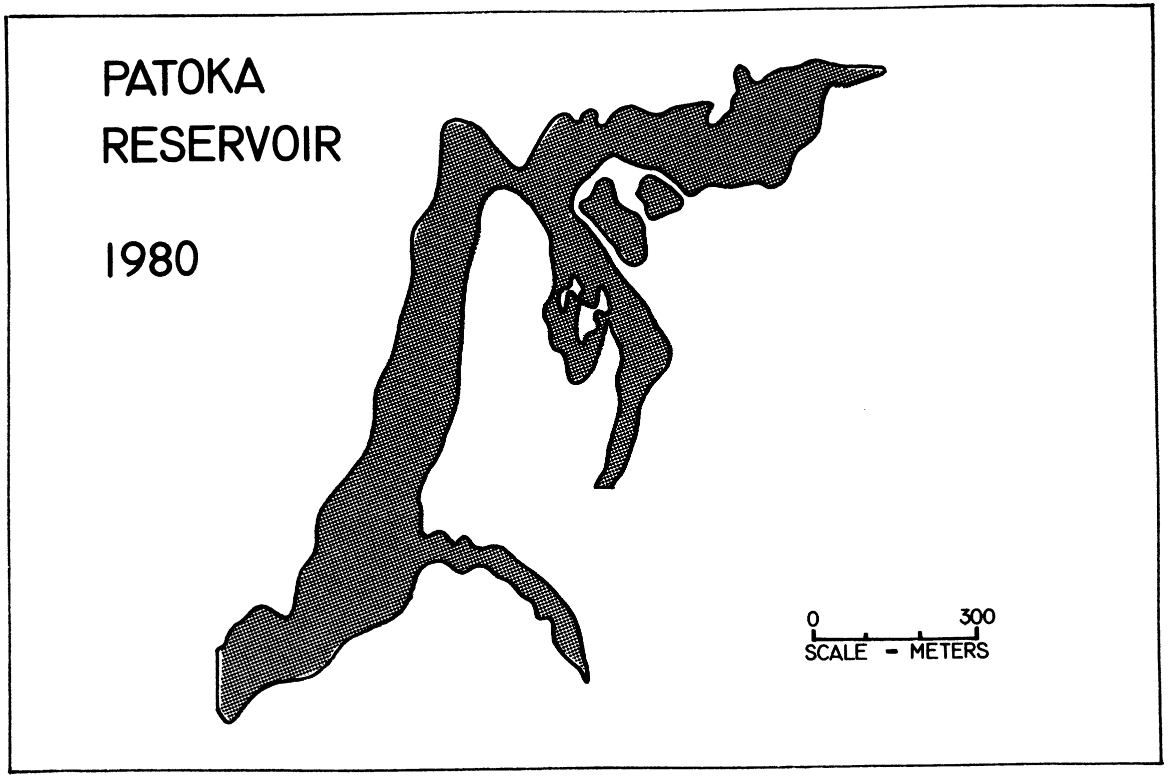


Figure 8. Section of Patoka Reservoir in 1980 showing 0.7 ha of wetlands (unshaded areas).

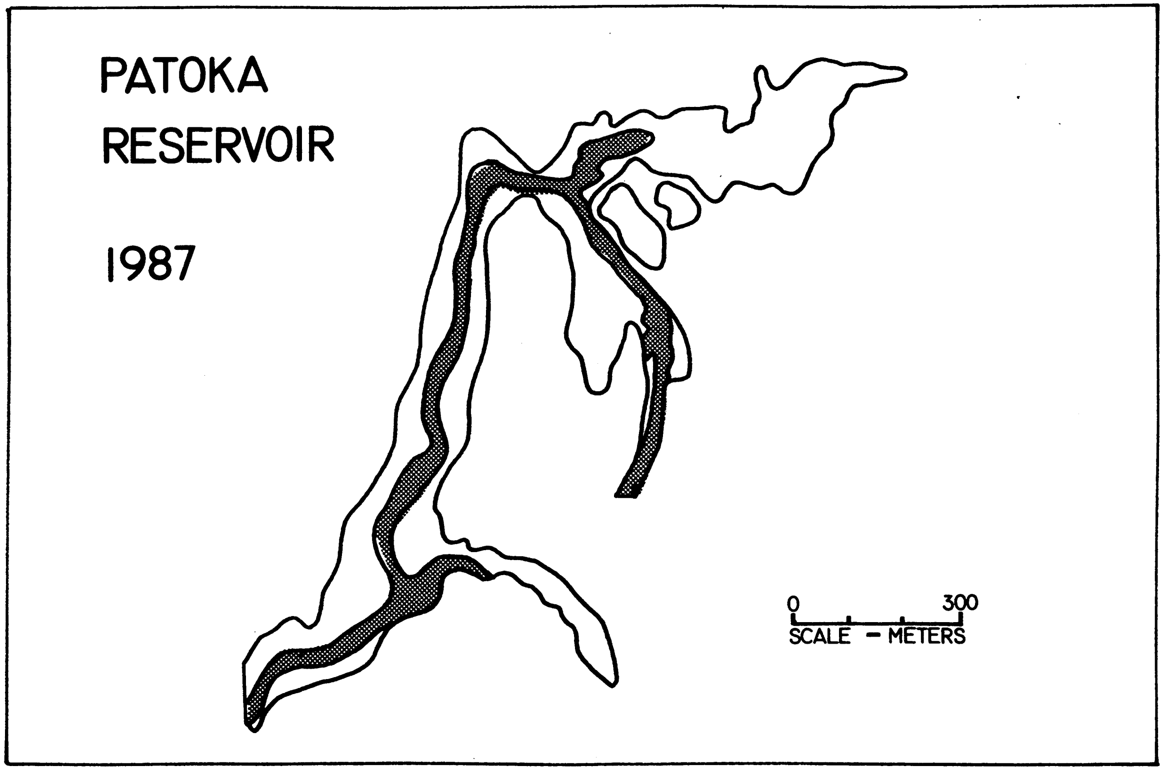


Figure 9. Section of Patoka Reservoir in 1987 showing 26.3 ha of wetlands.

change. Since plant community evolution is directly influenced by changes in water depth, caused by siltation, we cannot predict precisely what form a wetland will take as it expands.

Where wetland mitigation proposals involve raising the substrate to accelerate vegetation expansion, we suggest using fill material taken from wetlands destroyed by construction or dredging activities. Such substrata are likely to already contain the propagules of wetland species. We also encourage regulators to remain flexible in defining acceptable species compositions. Our observations indicate a diverse wetland will eventually evolve. The length of time required depends largely on the initial water depth and siltation rate.

Plotting wetland evolution in these reservoirs is further complicated by the influence of widely fluctuating water levels. Some species, such as American lotus (*Nelumbo lutea*), tolerate fluctuations better than others. In areas where lotuses have been introduced they grow densely at depths ranging from 10-100 cm, excluding all other species except duckweed (*Lemna* sp.). Wetlands that lack lotuses show a much greater species diversity, and, therefore, provide a more variable habitat for fish and wildlife. Where fluctuations are extreme, care should be taken to prevent the introduction of lotuses.

We also suggest caution in maintaining compatibility with other reservoir uses. Water supply companies may object to the loss of storage capacity that sedimentation and wetland generation may cause and may seek to remove wetlands from the reservoir basin. Outdoor recreation agencies may wish to control wetland growth to maintain access to reservoirs for boaters. Mitigation proposals should, therefore, contain agreements from the appropriate managing agencies that allow wetlands to develop.

Some characteristics of reservoir wetlands demand further attention. We know that wetlands expand in reservoirs--in the same way river deltas form--as inflowing streams deposit silt, but we have not attempted to correlate the rate of expansion with the flow rate or sediment load, both of which influence wetland evolution. Reliable estimates of the time required for functional wetlands to evolve would require such information.

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COMPARISON OF WETLAND HABITAT IN UNDISTURBED AND RECLAIMED PHOSPHATE SURFACE-MINED WETLANDS

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ABSTRACT

Macroinvertebrates inhabiting wetlands associated with two intermittent first order tributaries of the South Prong Alafia River, west-central Florida, were censused qualitatively to assess the phosphate industry's ability to reclaim wetland habitat in the wake of surface mining. The two stream channels eliminated by mining, Dogleg and Upper Hall Branches, were reclaimed with marshy reaches that retain water year round. Natural pools in Lower Hall Branch (undisturbed by mining) are permanently flooded but become isolated from one another during the dry winter months. Three reclaimed areas designated Dogleg Marsh (27 months old), Hall Branch Cypress Pond (15 months) and Hall Branch Marsh (1 month), and several natural pools in Lower Hall Branch, were sampled intensively with dip nets in 1986 prior to the onset of the summer rains. Czekanowski's Index indicated moderately high similarity (0.70) among samples collected within areas, but analyses between areas yielded moderately low to low similarity (0.17-0.39). The lack of similarity is largely attributable to the restricted distributions of the species of coleoptera, diptera, and heteroptera. Species richness was nearly twice as high in the older reclaimed areas than in the newly reclaimed area and the undisturbed channel. These results suggest that the richness of stream systems reclaimed with marshy areas will exceed that of undisturbed streams, but richness of reclaimed lotic sections will match that of similar undisturbed streams.

INTRODUCTION

The majority of freshwater wetland reclamation projects in Florida are being carried out by the state's phosphate mining companies. This large, concerted effort by a single group is due in part to the geomorphology of the land where the resource is located and partly to historical precedent.

Approximately 12% of the phosphate reserves in the Bone Valley district of Polk and eastern Hillsborough and Manatee counties in Florida are overlain by wetlands; the percentage decreases slightly on lands that will be mined in the future in the Southern Extension, an area encompassing Hardee and northern DeSoto counties (Simons et al. 1984). The native wetlands include herbaceous ecosystems (wet prairies and marshes) and tree- and shrub-dominated habitats (cypress domes, cypress swamps, mixed cypress-hardwood swamps, bay forests and swamp thickets). In order to responsibly extract the phosphate resource, demonstration of successful wetland reclamation is critically important.

In addition to the large wetland acreage in the mineralized district, the phosphate industry has been required by law to reclaim wetlands for over ten years. The rules of the Florida Department of Natural Resources dealing with phosphate mine reclamation state that "wetlands which are affected by mining operations shall be restored to at least premining surface areas." In other words, wetlands must be replaced acre-for-acre. This requirement encouraged the phosphate industry to begin reclamation research much earlier, and on a much larger scale, than any other group.

In December 1982 and May 1983, the Florida Department of Environmental Regulation (DER) issued permits to allow Brewster Phosphates to surface mine phosphate ore in the upper reaches of several first order tributaries of the South Prong Alafia River in southeastern Hillsborough County, Florida (Figure 1). One of the tributaries was unnamed, and was designated Dogleg Branch by Brewster. The second tributary was Hall Branch.

Mining plans for Dogleg Branch called for the stream channel and the attendant riverine forest to be completely eliminated by mining and reclaimed as upland pasture. To compensate for the loss of hardwood swamp, the DER required Brewster to restore forested wetlands on adjacent sites. Mining was completed in the area selected for the replacement stream channel for Dogleg Branch in 1982. Backfilling and grading were completed early in 1983, and revegetation began in mid-1983.

Mining was completed at Hall Branch in 1984, and reclamation at the same location began with backfilling and grading. This was completed later that same year. Revegetation began early in 1985.

By mid-1986, the wetland area had been in place for two years at Dogleg Branch and for one year at Hall Branch. We were interested in assessing the development of aquatic habitat in these systems, and determining if the restored wetland areas had come to resemble similar unmined wetlands. We decided to compare the aquatic macroinvertebrate communities inhabiting wetlands associated with Dogleg and Hall Branches as an indication of the development of these systems after reclamation. Both of the two stream channels had been reclaimed with marshy reaches that retain water year round. Natural pools in Lower Hall Branch (downstream and undisturbed by mining) are permanently flooded but become isolated from one another during the dry winter and spring months. Three reclaimed areas designated Dogleg Marsh (27 months old), Hall Branch Cypress Pone (15 months) and Hall Branch Marsh (1 month), and several natural pools in Lower Hall Branch were the sites of this investigation which commenced May 1986.

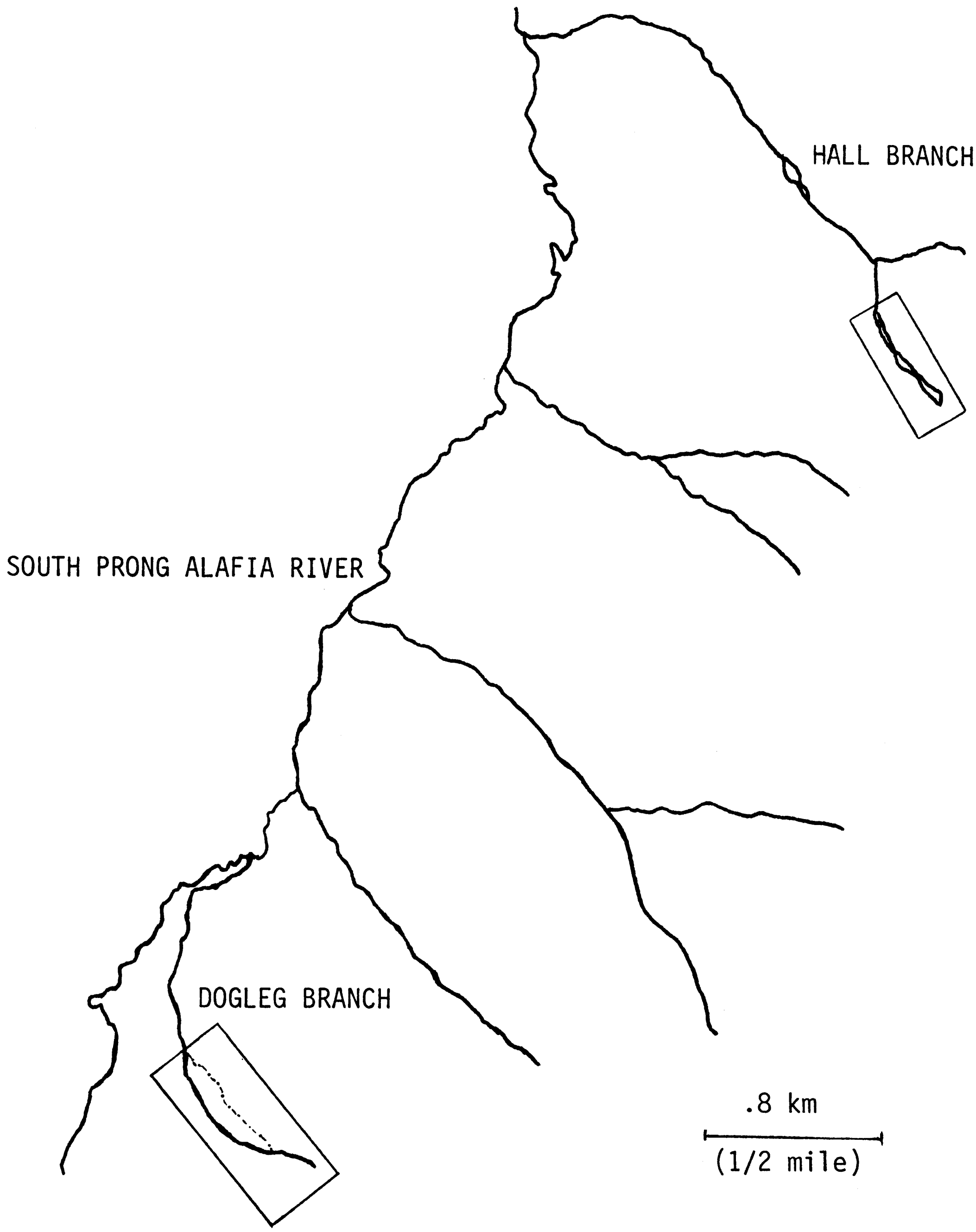

Figure 1. South Prong watershed. Rectangles mark the restoration sites.

SITE DESCRIPTION

Dogleg Branch

The Dogleg project (T31S, R22E, Sec. 20) includes a 19.75-acre restoration area and a six-acre tract of riverine forest at the head of the stream channel that was not mined and was subsequently incorporated into the project as the Dogleg Preserve (Figure 2). The reclaimed stream channel extends for 0.6 km. When grading on the site was completed, peat salvaged from the original channel was transferred to the reclaimed area and spread to a depth of about 30 cm. Some of the topsoil was stockpiled nearly three months before being spread, and the rest was spread the same day it was removed from the original forest. In order to detain runoff, encourage percolation, and inhibit erosion, several low berms were created on the slopes nearly parallel with the topographic gradient. Water collected in swales behind each berm and percolated into the groundwater. Annual grasses were sown on the area to retard runoff and erosion.

Over 400 cabbage palms were transplanted into the restoration area from the original Dogleg Branch during clearing operations. Stumps of 10 hardwood species felled during clearing were also moved to the restoration area. Promising initial survival rates for the transplants declined during subsequent years. Tree seedlings representing 21 species indigenous to the riverine forests of the Alafia River were planted in October 1983 and April 1984 as tubelings and saplings in gallon-sized containers. Additional plantings of gallon-sized trees were made in December 1984 and June 1985 in order to insure that more than 400 trees/acre survived. Red maples, sweetgum, water oak, laurel oak, sweetbay, and loblolly bay made up over two-thirds of the trees planted.

In the spring of 1984, several loads of topsoil from a marsh were deposited near the stream channel and spread as an inoculum for herbaceous plants. Maidencane was sprigged into the peat. A marsh was installed at the lower end of the reclaimed stream channel by similar inoculation with marsh soil. This marsh served as the location for our aquatic habitat assessment because it was the only permanent water body in the watershed.

The marsh encompasses approximately 1.5 acres. Pickerelweed is the most conspicuous species in areas that are continuously inundated. Cattails are also present among the pickerelweed. Maidencane grows profusely along the margins of the wetland where water depth is shallow and water coverage is intermittent. Vegetational coverage exceeds 80%. The only open water traces the stream channel through the marsh. Azolla and duckweed often cover the open water. Clewell (1986b) has documented the development of this site.

Hall Branch

The Hall Branch project (T31S, R22E, Sec. 10) includes five acres draining into the original Hall Branch stream system (Figure 3). After mining, the watershed was reclaimed with sand tailings capped with a layer of overburden. Two shallow depressions were excavated with a bulldozer to create marshes in the stream channel. The upper marsh (subsequently extensively planted with cypress and hereafter called Cypress Pond) comprises 1.85 acres and the Lower Marsh 0.4 acres. In March 1985, these two depressions were mulched with topsoil from a natural donor marsh. The topsoil was spread to a depth of 15 cm in the Lower Marsh and 7.5 cm in the Cypress Pond. A dense growth of marsh plants sprouted from the mulch and covered both sites in a few weeks. Overflow from the Lower Marsh cut a channel that connected directly with unmined Hall Branch.

In March 1985, cabbage palms were transplanted around the periphery of the project area. In June, several thousand gallon-sized tree seedlings were planted in the marshes and on the slopes immediately surrounding the marshes and stream. The species represented were bald cypress, pond cypress, popash, red maple, sweetbay, and American elm. Four-hundred nursery-grown maidencane plants were planted in the marshes at the same time.

The Hall Branch tributary was subject to erosion problems that required additional earthmoving and channel modifications below the Cypress Pond. Erosion has now been checked by a combination of dense growth of herbaceous aquatic vegetation (especially marsh pennywort, spikerush, and seaside paspalum), logjams to divert flow, and erosion control matting. However, prior to the establishment of these techniques, the Lower Marsh filled with eroded soil twice between September 1985 and March 1986, requiring additional excavation at the site. Trees planted on the shore of the marsh were temporarily moved during the earthmoving operations and additional herbaceous vegetation (softrush, giant bulrush and pickerelweed) was sprigged into the marsh basin following each excavation. Although erosion has not stopped completely, sand is being deposited in the marsh at a lower rate and marsh vegetation is colonizing the soil as rapidly as it is deposited.

At the same time the project was sampled, the initial cover of desirable marsh plants in the Cypress Pond had been colonized by cattails, primrose willow, and willow (Clewell 1986a). Cattails formed a dense stand in the deep water in the lower third of the marsh. Primrose willows were not dense, but their cover was appreciable because of their large size. Willows, on the other hand, were much more dense and covered nearly all of the area not dominated by cattails. Sampling was conducted among the cattails in the lower third of the pond. The area had not been significantly disturbed for 15 months prior to sampling.

The Lower Marsh supported little vegetation at the time of sampling. It had recently been excavated to remove deposits of eroded

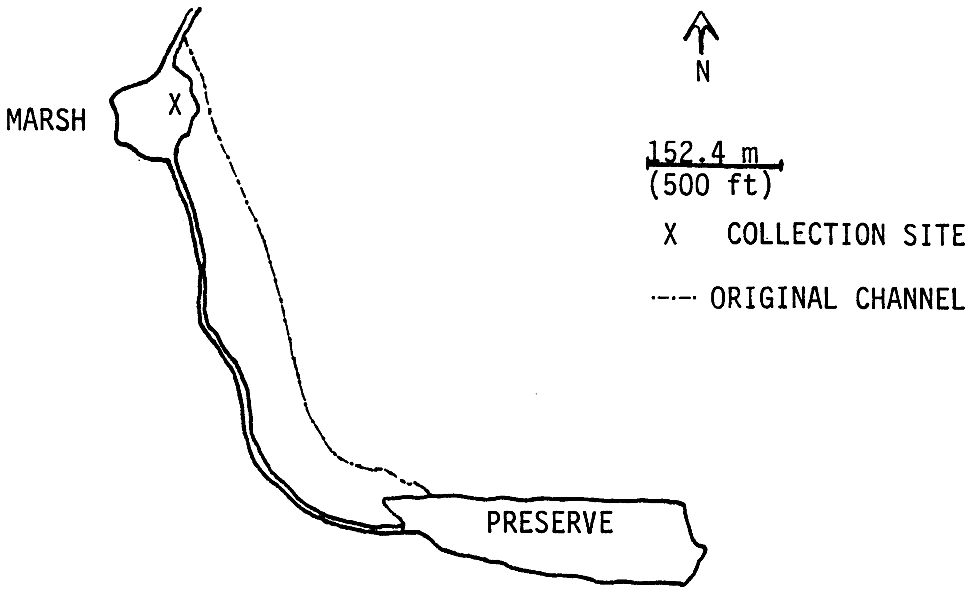

Figure 2. Dogleg Branch

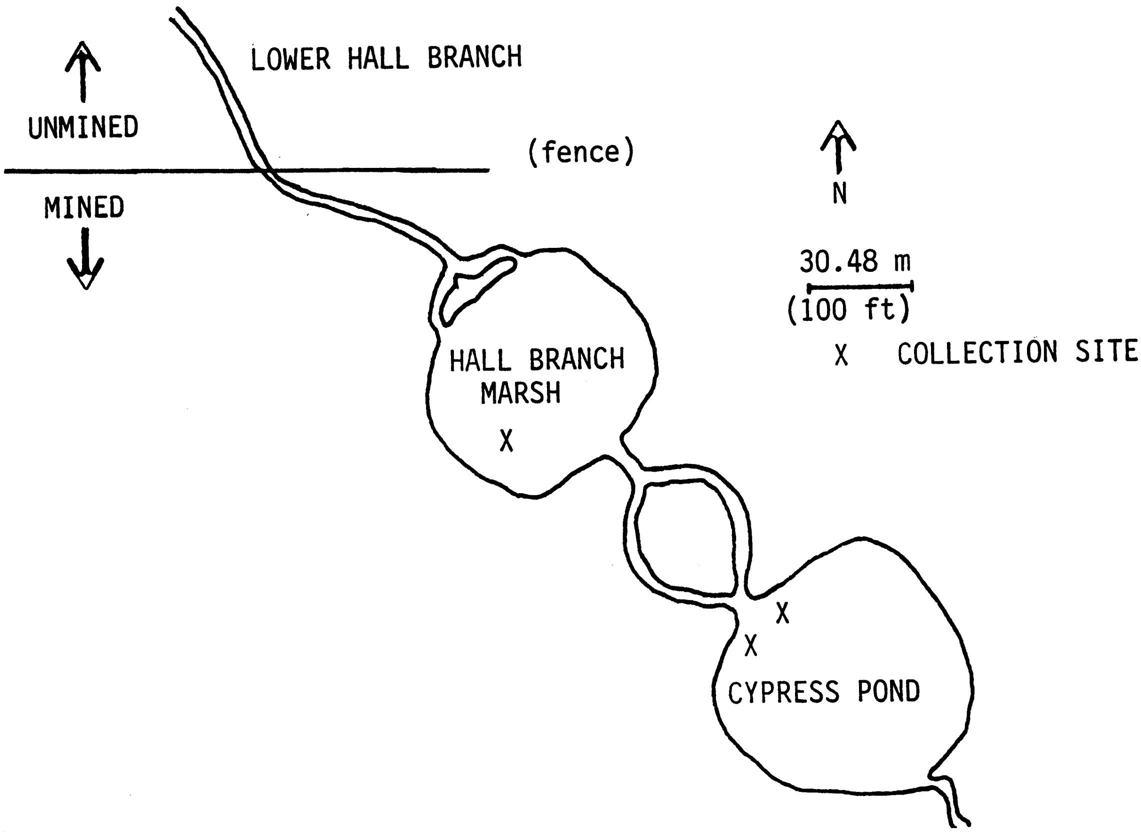

Figure 3. Hall Branch

sand. Clumps of bulrush and pickerelweed had been transplanted into the basin only a week earlier.

Lower Hall Branch

We made additional aquatic invertebrate collections immediately downstream of the restored Hall Branch project in an area that had not been mined. The bottomland area was clothed with masses of grape vines draped over wax myrtles and other shrubby plants. A continuous stream channel was lacking. There were a few pockets of inundated peat, and some ill-defined segments of channel were discernible. Intervening bottomland between stream segments was covered with deep leaf litter. We sampled the aquatic invertebrates inhabiting several of the discontinuous pools in the stream channel.

METHODS

Aquatic macroinvertebrate samples were collected on May 16, 1986 with long-handled nylon dip nets protected with muslin skirts designed to prevent the mesh from snagging in vegetation. By shaking and dragging the net in the vegetation, organisms were dislodged and collected in the net. In an effort to make the collections semi-quantitative, sampling was restricted to three minutes for each netting. Two such three-minute collections were made in each of the wetlands sampled and the collections composited into one sample. Hall Branch Cypress Pond was sampled at two locations. The collections at each location included two three-minute dip net samples. The material collected in the net was transferred to a plastic container, preserved with 70% ethanol, and returned to the laboratory for additional processing.

After washing twigs, leaves and pebbles with a water spray over a sieve (125 mesh) to remove lodged organisms, these large pieces of debris were discarded. The material retained by the sieve was composited with the remainder of the preserved sample. Small portions of the preserved sample were then decanted into white plastic trays and sorted under strong light. Large pieces of detritus were agitated in water in the pan, examined for clinging invertebrates, then discarded. Detritus and sand grains too small to remove from the tray by hand were transferred to a watch glass and examined at 12X magnification. All sorted invertebrates were placed in labeled vials, preserved in 70% ethanol and identified.

Richness (number of distinct taxa present) was determined for each macroinvertebrate sample. Habitat similarity was then estimated by comparing the aquatic macroinvertebrate samples using Czekanowski's Index,

$$\frac{2a}{2a + b + c}$$

where "a" is the number of taxa common to both collections, "b" is the number of taxa unique to sample 1, and "c" is the number of taxa unique to sample 2. Values for Czekanowski's Index can range between 0.0 (for samples with no taxa in common) to 1.0 (for samples with identical taxon lists).

Taxonomic diversity of the samples was calculated using the Shannon-Weaver Index,

$$H' = \sum_{i=1}^{s} P_1 \ \text{In}\ p_1$$

where "s" is the number of taxa and "p_i" is the proportion of the total number of individuals belonging to the i-th taxon.

RESULTS

We were able to identify a total of seventy distinct invertebrate taxa from the four wetlands on the two tributaries. However, lack of taxonomic expertise for some groups, immaturity of some specimens, and poor quality of other specimens required that some of the organisms be grouped in more generalized classifications. For example, our group, Hydracarina, undoubtedly contains numerous species that we were unable to distinguish. Likewise, three of the odonate nymphs could be identified only to the family level because of their poor condition. The taxa comprising the samples were distributed among three phyla, five classes, 16 orders, and at least 35 families and 63 genera. A complete list of the taxa collected in presented in Table 1. Among the reclaimed sites, the two Hall Branch Cypress Pond collections had the greatest richness (40 and 34 taxa), followed by Dogleg Marsh (31 taxa) and Hall Branch Marsh (18 taxa). The pools sampled in the intermittent channel of Lower Hall Branch contained 14 taxa (Table 2).

The results of the Czekanowski similarity comparisons are summarized in Table 3. Two samples were collected from the Hall Branch Cypress Pond, and Czekanowski's Index reveals 0.70 similarity between the collections, the highest value of all comparisons in our investigation. All other comparisons which were made <u>between</u> locations instead of within, revealed that the wetlands support invertebrate communities that are moderately low to low in similarity (0.17-0.39).

Shannon-Weaver diversity values were highest for the samples collected in the oldest reclaimed wetlands (Dogleg Marsh: 2.682; Hall Branch Cypress Pond I: 2.308; Hall Branch Cypress Pond II: 2.108). The Lower Hall Branch pool sample had slightly lower diversity, 2.068. The collection from the newly recontoured Hall Branch Marsh had the lowest diversity (0.832).

Table 1. Macroinvertebrate Collections, May 1986.

SAMPLE IDENTIFICATION[1]	HBCP	HBCP II	HBU	HBM	DL
ANNELIDS	1	-	3	-	-
MOLLUSKS					
Valvata sp.	-	1	-	-	-
Physa sp.	23	31	8	-	-
ARTHROPODS					
CRUSTACEANS					
Ostracods	48	251	-	5	1
Copepods	200	900	2	-	1
Isopods					
Hyallela azteca	103	103	10	1	3
Crayfish	-	-	2	-	-
Unknown	-	-	1	-	-
ARACHNIDS					
Hydracarina	-	1	-	-	1
INSECTS					
ODONOTA					
Anomalagrion sp.	93	166	11	1	-
Erythemis sp.	11	21	2	-	-
Aeshnidae	1	-	-	-	1
Libellulidae	-	-	-	-	1
Anax junius	-	2	-	-	-
EPHEMEROPTERA					
Baetis sp.	4	7	9	3	1
Caenis sp.	14	22	4	-	-
HETEROPTERA					
Merragata sp.	2	32	-	-	-
Mesovelia sp.	8	57	1	1	1
Notonecta sp.	1	-	-	-	-
Neoplea striola	9	19	-	-	-
Pelocoris carolinensis	1	20	-	-	9
Microvelia pulchella	-	2	-	-	-
Ranatra nigra	-	-	-	-	1
TRICHOPTERA					
Oecetis sp.	-	-	1	-	-
COLEOPTERA					
Dytiscidae	-	-	-	-	1

Table 1. Macroinvertebrate Collections, May 1986 (cont'd).

SAMPLE IDENTIFICATION[1]	HBCP	HBCP II	HBU	HBM	DL
Dytiscus sp.	2	25	-	-	-
Hydrovatus pustulatus	2	-	-	-	-
Laccophilus sp.	3	8	-	3	-
Laccophilus larvae	-	-	-	2	-
Laccornis sp.	11	48	-	-	-
Onychylis sp.	1	1	-	-	-
Rhinoncus longulus	-	-	-	-	9
Peltodytes sp.	9	-	-	-	2
Peltodytes larvae	-	24	-	-	-
Berosus sp.	1	1	-	3	1
Cymbiodyta sp.	1	1	-	1	-
Hydrobius tumidus	-	1	-	-	-
Tropisternus sp.	-	12	-	30	2
Tropisternus larvae	-	-	-	-	6
Hydrocanthus iricolor	1	-	-	-	30
DIPTERA					
Anopheles sp.	1	18	-	-	-
Culex sp.	-	8	32	1	-
Odontomyia sp.	1	2	-	-	8
Larsia sp.	27	88	-	-	4
Zavrelimyia sp.	1	2	-	-	-
Ablabesmyia sp.	17	24	-	-	-
Tanypus sp.	1	-	-	-	-
Polypedilum sp.	16	13	-	-	11
Limnochironomus sp.	-	1	4	-	3
Endochironomus sp.	-	-	-	-	2
Pseudochironomus sp.	-	1	-	-	1
Glyptotendipes sp.	-	-	1	-	-
Chironomus sp.	-	-	-	428	-
Paracladopelma	1	-	-	-	-
Parachironomus sp.	-	-	-	4	1
Tanytarsus sp.	-	-	-	36	1
Cladotanytarsus sp.	-	-	-	-	1
Eukiefferiella sp.	-	-	-	-	1
Cricotopus sp.	-	-	-	4	9
Orthocladinae	-	-	-	-	1
Tipulidae larvae	-	1	-	-	-
Atherix lantha	-	1	-	-	-
Psychoda sp.	-	-	-	1	-
Telmatoscopus albipunc	-	1	-	-	-
Atrichopogon sp.	-	-	-	2	-
Alluaudomyia sp.	-	-	-	1	-
Palpomyia sp.	3	15	-	-	1 2
Sphaeromias sp.	-	-	-	-	3

Table 1. Macroinvertebrate Collections, May 1986 (cont'd).

SAMPLE IDENTIFICATION[1]	HBCP	HBCP II	HBU	HBM	DL
LEPIDOPTERA					
Eumorpha sp.	1	4	-	-	-
Lithacodia carneola	-	2	-	-	-

[1]HBCP = Hall Branch Cypress Pond I
HBCPII = Hall Branch Cypress Pone II
HBU = Lower Hall Branch
HBM = Hall Branch Marsh
DL = Dogleg Marsh

Table 2. Species richness.

Hall Branch Cypress Pond Collection 1	40
Hall Branch Cypress Pond Collection 2	34
Dogleg Marsh	31
Hall Branch Lower Marsh	18
Lower Hall Branch	14

Table 3. Czekanowski's similarity.

HBCP I Hall Branch Cypress Pond Collection 1
HBCP II Hall Branch Cypress Pond Collection 2
LHB Lower Hall Branch
HBM Hall Branch Marsh
DM Dogleg Marsh

HBCP I	vs.	HBCP II	0.70
	vs.	DM	0.40
	vs.	LHB	0.37
	vs.	HBM	0.31
HBCP II	vs.	DM	0.38
	vs.	LHB	0.35
	vs.	HBM	0.34
LHB	vs.	HBM	0.31
	vs.	DM	0.17
HBM	vs.	DM	0.33

DISCUSSION

Taxonomic richness assessed with a dip net is not a quantitative measure of the species present in an aquatic habitat because the coarse mesh size of the net allows small and immature macroinvertebrates to escape. In addition, it is impossible to quantify the volume of habitat that has been sampled. Nonetheless, we attempted to make our collections comparable by sampling for three minutes and by collecting in a consistent manner in each wetland. As such, we feel that while the dip net collections may be biased toward larger invertebrates, the samples from all the sites are similarly biased and are therefore comparable. It is also worth noting that previous work conducted by Robertson and Piwowar (1985) in similar reclaimed streams in central Florida demonstrated that dip nets yielded the highest taxonomic richness of four aquatic sampling techniques employed.

In general, richness was nearly twice as great in the older reclaimed areas (Hall Branch Cypress Pond and Dogleg Marsh) than in the newly reclaimed area (Hall Branch Marsh) and in the undisturbed intermittent channel (Lower Hall Branch). In fact, many of the organisms collected had very restricted distributions. Of all 70 taxa collected, over half (37 taxa) were present in only one out of four wetlands. The majority of these 37 were restricted to the reclaimed wetlands, with 26 found only in one of the two older wetlands and seven found only in the newest wetland, Hall Branch Marsh. Only four taxa were found exclusively in undisturbed Lower Hall Branch.

In the case of taxa found in only two out of four wetlands, nearly all were restricted to reclaimed areas. The oldest reclaimed areas shared eight taxa, the new and old reclaimed wetlands together shared six taxa, and the reclaimed and undisturbed wetlands had three taxa in common.

Eight taxa were present in three out of four areas. Three were restricted to reclaimed sites and five were present in both the reclaimed wetlands and the undisturbed stream channel. Only three taxa were ubiquitous, the isopod, *Hyalella azteca*, the mayfly nymph, *Baetis* sp., and the water treader, *Mesovelia* sp.

The restricted distribution of such a large proportion of taxa (57% were found in only one out of four wetland areas and another 26% were restricted to only two areas) led to generally low similarity between samples (0.17-0.38). The lack of similarity is largely attributable to the restricted distributions of the three largest orders of aquatic insects in the wetlands: true bugs, beetles and flies. A very large majority of the bugs (71%) and flies (64%), and over half (55%) of the beetles were restricted to only one out of four wetlands. In addition, most of these taxa with restricted distributions were found only in the older Hall Branch Cypress Pond and Dogleg Marsh.

Most of the results obtained during the course of this investigation would have been intuitive to an aquatic ecologist touring the wetland areas. The undisturbed Lower Hall Branch pools had the lowest richness of all habitats sampled. These pools, while containing water when they were sampled, probably dry out completely in some years with very low spring rainfall. In addition, the small size of the pools may produce stressed conditions such as anoxia even when open water is present.

The low richness of Hall Branch Marsh is a result of repeated disturbances and lack of suitable habitat for aquatic organisms. The marsh basin had filled with sediment on two occasions during the 18 months prior to the time that samples were collected for this study. Each time the marsh filled with sand, Brewster Phosphates reclamation personnel were forced to excavate a new basin with heavy equipment, completely eliminating any vegetation that had colonized the area. After each recontouring, new herbaceous and arboreal vegetation was introduced onto the site. When we made our collections, we concentrated our sampling efforts in the water around the few clumps of newly planted bulrush and pickerelweed.

The older wetlands in Hall Branch Cypress Pond and Dogleg Marsh, in contrast, had never been resculptured after the initial basin excavation. As a result, they supported dense emergent, submerged and floating vegetation, all of which provided diverse macroinvertebrate habitat. In addition, water levels have remained fairly constant and predictable. These wetlands have never fully dried or even shrunken to the extent that they have become anoxic.

While our findings were predictable, the implications of our results are not as straightforward. The Florida DER dredge and fill permits include success criteria for restored wetlands. Although heavily emphasizing the rehabilitation of wetland vegetation, the Department often includes an assessment of aquatic macroinvertebrate colonization as one of its measures of success. In order to meet the criteria imposed by the DER, macroinvertebrate diversity must be similar to that in a reference wetland, either the identical wetland monitored before mining occurred, or in a nearby wetland with similar characteristics.

Our results suggest that the richness of watersheds reclaimed with marshy areas will exceed that of the small first order headwater drainages that they replaced. The richness of the older Hall Branch Cypress Pond and Dogleg Marsh was far greater than the richness of the intermittent pools sampled in undisturbed Lower Hall Branch, the type of habitat that was present prior to mining in the upper reach of Hall Branch and Dogleg Marsh.

Differences in taxonomic diversity between the samples collected from the older reclaimed wetlands and the undisturbed Lower Hall Branch pools are not as striking as the differences in richness. Diversity values for Dogleg Marsh and the Hall Branch Cypress Pond collections were slightly greater than the value for the reference wetland. The recently recontoured Hall Branch Marsh supported low numbers of few organisms leading to low diversity for the samples collected there. While diversity of the older reclaimed wetlands was similar to the diversity of Lower Hall Branch, the two types of habitat are significantly different and would be expected to support different species assemblages, as the richness data demonstrate. Nonetheless, if macroinvertebrate richness and diversity are among the criteria that must be met, marshy reaches included in headwater streams will almost certainly be at least if not more rich and diverse than the intermittent stream channels that formerly occupied the sites.

Additionally, if watersheds are reclaimed as a series of marshy basins connected by reaches of lotic habitat, the richness and diversity of the reclaimed lotic sections will more than likely closely match those of similar small headwater streams. Although we didn't census macroinvertebrate populations during the course of this investigation in the small reclaimed channels between the marshes because of lack of water, the conditions in these reclaimed channels are very similar to those in Lower Hall Branch (e.g., interrupted flow, small pools formed by logjams). As such, the richness and diversity of these areas would probably be very similar to those in the Lower Hall Branch pools.

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THE UTILITY OF BREDER TRAPS FOR SAMPLING MANGROVE AND HIGH MARSH FISH ASSEMBLAGES

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ABSTRACT

Rectangular plastic fish traps, developed by Breder (1960), were tested in mangrove and salt marsh habitats by comparing them to 1.0 m_2 throw nets in high marsh environments and pull nets (modified seines) in tidal creeks. While subject to behavioral bias, the traps are non-destructive and rapidly sampled. Breder traps can supply excellent spatial and temporal resolution of fish movement at reasonable cost in marsh studies where 1) relative densities or catch per unit effort data will suffice, and 2) the principal marsh resident fish species can be used as indicator species. Our study suggests that Breder traps may be useful for the functional assessment of restored and newly created marshes as well.

INTRODUCTION

The functional assessment of restored and natural wetland habitat is one of the most important tasks facing estuarine scientists and managers today. Knowledge of the carrying capacity of estuarine habitats and the habitat requirements of all life stages of economically important fish species is critical in making decisions about habitat preservation. The success of restored and created wetlands must be judged by functional attributes rather than plant survival. As fish comprise an important biological flux mechanism coupling wetlands to estuarine food webs, techniques are needed to measure the absolute or relative densities of wetland fish species.

Many techniques have been used to sample fish in mangrove and salt marsh habitats (Table 1) They can be grouped into two general categories, active gear and passive gear, based on their general principles of operation. Active gear, such as seines and throw nets, catch fish by physically surrounding them through active participation of the operators. Active gear types are potentially quantitative on an areal basis. Although some gear avoidance by fast and wary species may occur, active gear types have less overall bias than passive gear types. Most active gear types are restricted to areas of open water or limited succulent vegetation. Those that can be used in thick undergrowth are extremely destructive to habitat. Intensive labor is often required to sample with active gear types.

Table 1. List of fish sampling techniques used in mangrove and salt marshes, comparing major advantages and disadvantages.

TECHNIQUE	ADVANTAGES	DISADVANTAGES
ACTIVE GEARS SPECIES,	LESS BIAS, MORE QUANTITATIVE	LABOR INTENSIVE, AVOIDANCE BY SOME SPECIES, NOT PRACTICAL IN DENSE VEGETATION
SEINES	collects large numbers of individuals and species	feasible only in open water with flat bottoms
THROW TRAPS	works in herbaceous vegetation	very destructive, labor intensive
PULL NETS	quantitative, low avoidance bias, low labor, only moderate habitat disturbance after set up	suitable only in creeks and ditches, initial set up can be expensive and laborious
DROP NETS	quantitative	destructive, selective capture due to avoidance, expensive, complicated to build and set up
BLOCK NET & ROTENONE	low bias, quantitative	destructive, suitable only where poisons can be used, cannot be frequently repeated
CAST NET	low labor, high replication, inexpensive & simple	selective capture due to avoidance, not quantitative
PASSIVE GEARS	NOT AS DESTRUCTIVE, SUITABLE FOR USE IN DENSE VEGETATION	BEHAVIORAL BIAS, NOT QUANTITATIVE
FYKE NETS		limited to suitable locations
HEART TRAPS	suitable for use in moderately dense vegetation and with low water levels	behavioral bias, cumbersome to set and retrieve, typically low yields

Table 1. List of fish sampling techniques used in mangrove and salt marshes, comparing major advantages and disadvantages (cont'd).

TECHNIQUE	ADVANTAGES	DISADVANTAGES
MINNOW TRAPS	low labor, high replication, inexpensive & simple	behavioral bias, low collecting rate, most require a minimum water depth
FLUMES	low behavioral bias, quantitative	limited to suitable locations, potentially destructive, labor intensive, recovery efficiency variable
BREDER TRAPS	suitable for use in all types of vegetation and water levels, replication with low labor, inexpensive & simple	behavioral bias
GILL NETS	ease of sampling, excellent for predators	not suited to dense vegetation, size selective, not suitable for collecting small fish

Passive gear (traps) are stationary devices into which fish swim of their own accord. Only gear set-up and retrieval of fish require human attention. Passive gear types are not as destructive to habitat, and many are suitable for use in dense vegetation. Unfortunately, they cannot be used to estimate absolute densities. Because they are stationary, passive gears collect active foraging species while under-representing other species, especially predators.

Although all the aforementioned gear types work well under the circumstances for which they are designed, each has limitations. Most are destructive to habitat (which is extremely undesirable in restored or created wetlands), and none of them can be used with the same efficiency in all the types (and varying densities) of vegetation encountered in mangrove marshes. This hinders comparisons between marshes as well as nearly eliminating any possibility of spatial resolution of fish abundance among vegetation zones of the same marsh.

A single method which can be used to compare densities of fish among different marshes, or different zones of the same marsh, is needed. This method must perform efficiently in even the densest vegetation while causing only minimal habitat disturbance. Statistical validity through replication with reasonable cost and effort is also desirable. The plastic fish trap designed by Breder (1960) appears to meet these requirements.

We did not intend to compare fish sampling gear as part of our stable isotope food web study, but we tested Breder traps after initial sampling with throw nets destroyed an unacceptable amount of vegetation on the high marsh and proved impossible in the forested zones of the site. Our preliminary comparison of Breder traps with other sampling techniques suggests that Breder traps may be a useful alternative to other sampling techniques. This study was funded by the Department of Environmental Regulation Office of Coastal Zone Management and the NOAA Office of Ocean and Coastal Resource Management.

STUDY SITE

This study site is a low-lying mangrove island located in the Indian River Lagoon 3 km north of Fort Pierce, Florida. A tidal creek which runs along the west edge of the island broadens at its northern end into a small lagoon. Vegetation on the island is arranged in concentric zones. The outermost (and lowest elevation) zone is a monotypic fringe of *Rhizophora mangle* which grades into a mixed zone of *Avicennia germinans* and *Rhizophora mangle* forest. The third zone is a transitional community with short (2-3 meter) *Avicennia germinans* and *Laguncularia racemosa* with an understory of *Batis maritima*. A succulent high marsh community, composed of *Batis maritima*, *Salicornia* spp., and *Avicennia* seedlings, occupies the centermost and highest portion of the island.

Runoff of water from flooding tides and rain is impeded by a slight berm near the outer edge of the *Rhizophora* fringe. This can cause 5 to 10 cm of standing water to remain occasionally on the marsh surface in between flooding tides. Large numbers of fish, mostly *Gambusia affinis* and *Cyprinodon variegatus*, are commonly observed foraging in standing water on the transitional and mixed zones of the marsh.

METHODS

The marsh was sampled monthly from March 1986 to April 1987. Sampling trips were coordinated to coincide with new or full moon spring tides which we hoped would flood the entire marsh.

Breder Traps

A Breder (1960) trap (Figure 1) consists of two parts, a rectangular funnel which directs fish into the trap and a box (30 cm x 15 cm x 15 cm) where they are held until collection. Pre-cut Plexiglass pieces (1/8 inch thick) joined by Plexite® cement were used to construct traps at a unit cost of $9.00. Two rectangles, approximately 30 cm long and 15 cm high, form the sides of a rectangular funnel which guides fish into the box. The width of the slit at the narrow end of the funnel depends on the size of fish the trap is to catch; we used an opening of 12 mm. The vertical slit collects fish over its entire 15 cm height. Therefore, they can collect fish in any water deep enough for fish to swim. Traps can be set directly on the marsh surface, or attached to a float to keep it at the water's surface or at any intermediate depth. Traps are sampled by removing the wings and pouring the contents into a small dip net or a holding container.

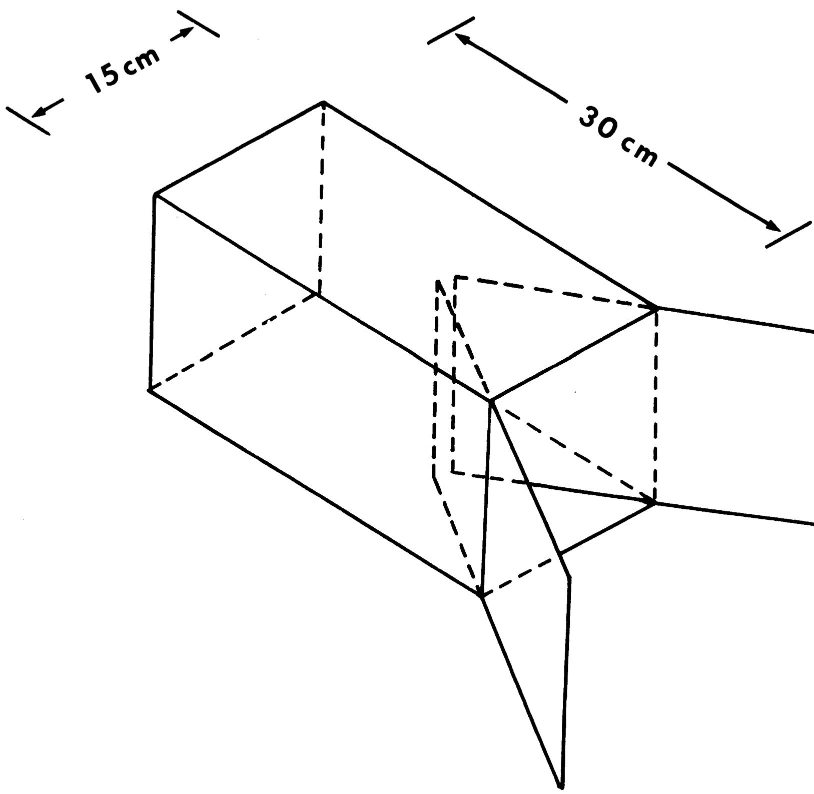


Figure 1. Perspective diagram of Breder (1960) trap.

The Breder traps were set on the marsh in a stratified design based on major vegetation zones. Four traps, separated by at least 5m where possible, were set in each of the following vegetation strata: 1) inner lagoon (IL) at mouth of tidal creek, 2) tidal creek (TC), 3) red mangrove fringe (FR), 4) mixed black and red mangroves (BR), 5) transitional zone (TR), and 6) high marsh (HM). Traps were set in place on a mid-flood tide before any water covered the marsh and sampled on the following ebb tide, when water had receded off the marsh. Sampling was conducted again over the next three or four consecutive high tides. On occasions when the marsh did not flood, sampling could be conducted only at the lagoon and tide creek.

A soak time experiment was conducted in April 1987 to determine the fishing characteristics of the traps. Twelve traps were placed in the tide creek for three hours between high tide and mid-ebb tide. Every 15 minutes, four randomly chosen traps were emptied and replaced. Total catch was regressed against time using SAS (SAS Institute 1985).

Pull Nets

The pull nets used to sample the tide creek are modified seines developed by Gilmore (1987). Two permanent docks were built across the creek 18 meters apart. A barrier net, consisting of a fine-mesh seine, was dropped from one dock and the pull net, a bag seine with a many-ended foot rope, was dropped from the other dock entrapping fish between the nets. Ropes were used to pull the bag seine down the creek to the barrier net dock. The lead line was carefully brought to the surface while keeping it against the barrier net to prevent fish from escaping. Once the pull net was lifted up onto the barrier net dock, fish were removed. Pull net sampling was conducted on both high and low tides over the course of at least one full tidal cycle. This provided four to six collections each month. No pull net samples were taken on those occasions when there was no water in the tide creek at low tide.

Throw Nets

Throw nets, consisting of a square aluminum box without a top or bottom, were used to sample the high marsh. A single person quietly moved into position and threw a 0.5 m^2 or 1.0 m^2 net over a selected target area. The box was then pushed down and sealed into the sediment. All vegetation within the box was clipped to ground level, checked for fish, and discarded. Fish were then collected from the box with a large square framed dip net until ten successive sweeps with the dip net caught no additional fish.

Comparison of Techniques

To estimate gear bias, the percent occurrence, total abundance, and percent dominance of several representative fish species were compared among gear types. We define percent occurrence as the number of samples for a given gear type which contained at least one individual of a particular species divided by the total number of samples for that gear type. Percent dominance, calculated from total catch data (Table 3), is defined as the total catch of a particular species divided by the total catch of all species for a given gear type. Only data from high marsh Breder traps were compared with throw net data, and only data from tide creek and inner lagoon Breder traps were compared with pull nets.

RESULTS AND DISCUSSION

The fish assemblage of the study site included both marsh resident species, which carry out their entire life cycle in the marsh or adjacent waters, and transient species which spend only a portion of their life cycle in the marsh. A total of 35 fish and invertebrate species were collected in various gear types throughout the study area, but the 20 species listed in Table 2 comprised 97% of total catch on the marsh surface and 81% of total catch in the tidal creek. Four marsh resident fish species, Cyprinodon variegatus, Fundulus confluentus, Gambusia affinis, and Poecilia latipinna, represented 91% of total catch for Breder traps on the marsh surface and 73% of the total catch of throw nets on the high marsh (Table 3). These four species are active foragers, which are quite common and abundant in coastal marshes.

Anchoa mitchilli was the most abundant transient species collected in the tidal creek, even though it occurred in only 23% of the pull net samples. While the total catch of other transients was not large, Centropomus, Diapterus, Eucinostomus, Leiostomus, and Mugil species all occurred in at least 10% of pull net samples.

Few transient species were collected in high marsh samples of any gear type. Centropomus, Eucinostomus, and Mugil were collected in Breder traps, while one Mugil was collected in a throw net sample. These results reflect both the low densities of most transient species on the high marsh and the excellent gear avoidance behavior of transients, most of which are predators with excellent sight. The Centropomus captured in Breder traps may have been "baited" by Poecilia already inside. Callinectes sapidus and Penaeus duorarum were also collected in Breder traps on the marsh surface.

The percent occurrence values of the common marsh resident fish species in Breder traps and other gear types were comparable in both high marsh and tidal creek collections (Table 2). However, the total catch and percent dominance values (here defined as the total catch of a particular species divided by the total catch of all species) of the marsh resident fish species indicate that Breder traps have a strong positive bias for species which actively forage on the marsh surface. While the percent occurrence of Poecilia latipinna on the high marsh was greater for throw nets than Breder traps, Poecilia comprised 64% of the total Breder trap catch, and only 22% of the total throw net catch.

Fundulus confluentus accounted for 12% and 16%, respectively, of the Breder trap catch in the high marsh and tidal creek, but only 1% and 2%, respectively, of the catch in other gear types in the same environments. In the tidal creek, the percent dominance of *Gambusia affinis* was greater in Breder traps than in pull nets, while the percent dominance of *Gambusia* in high marsh Breder traps was lower than in throw nets.

Table 2. Percent occurrence of several species of fish and invertebrates in several gear types tested on a mangrove island in the Indian River (Florida) lagoon.

HABITAT	HIGH MARSH		TIDE CREEK	
SPECIES	BREDER TRAPS n=141	THROW NETS n=14	BREDER TRAPS n=224	PULL NETS m=48
RESIDENT SPECIES				
Cyprinodon variegatus	28	21	10	31
Floridichtys carpio	7	0	0	0
Fundulus confluentus	29	14	39	27
Fundulus grandis	7	0	2	12
Gambusia affinis	15	36	40	42
Gobiosoma sp.	1	0	12	15
Lucania parva	4	0	5	6
Poecilia latipinna	37	64	55	42
Rivulus marmoratus	1	0	0	4
TRANSIENT SPECIES				
Anchoa mitchelli	0	0	0	23
Calinectes sapidus	4	0	6	10
Centropomus sp.	4	0	5	21
Diapterus sp.	0	0	1	31
Eucinostomus sp.	2	0	8	39
Lagodon rhomboides	0	0	0	6
Leiostomus xanthurus	0	0	1	17
Mugil sp.	1	7	1	46
Penaeus sp.	2	0	1	15
Pogonias cromis	0	0	0	8
Sciaenops ocellatus	0	0	1	6

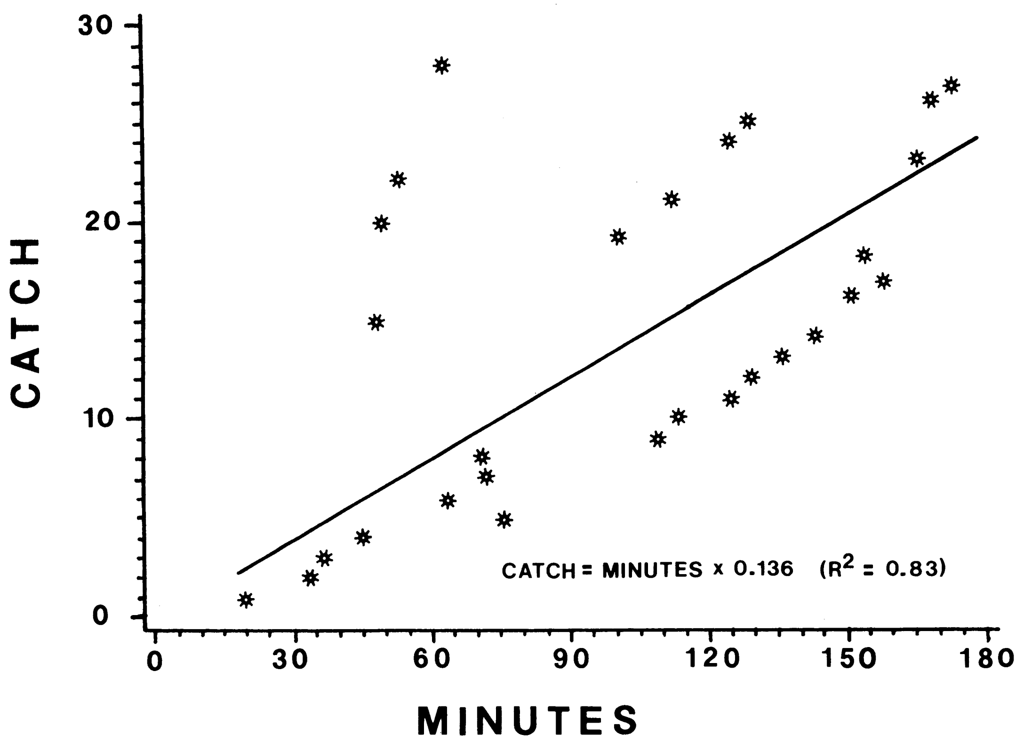
Table 3. Total catch of several fish and invertebrate species in several gear types compared on a mangrove island in the Indian River (Florida) lagoon.

HABITAT	HIGH MARSH		TIDE CREEK	
SPECIES	BREDER TRAPS n=141	THROW NETS n=14	BREDER TRAPS n=224	PULL NETS m=48
RESIDENT SPECIES				
Cyprinodon variegatus	143	50	46	324
Floridichtys carpio	25	0	0	0
Fundulus confluentus	144	3	350	299
Fundulus grandis	16	0	15	11
Gambusia affinis	50	84	490	2231
Gobiosoma sp.	1	0	37	66
Lucania parva	8	0	40	108
Poecilia latipinna	801	49	977	958
Rivulus marmoratus	1	0	0	2
TRANSIENT SPECIES				
Anchoa mitchelli	0	0	0	6342
Calinectes sapidus	10	0	18	17
Centropomus sp.	2	0	10	58
Diapterus sp.	0	0	1	268
Eucinostomus sp.	3	0	23	126
Lagodon rhomboides	0	0	0	4
Leiostomus xanthurus	0	0	3	77
Mugil sp.	17	1	1	249
Penaeus sp.	3	0	2	46
Pogonias cromis	0	0	0	12
Sciaenops ocellatus	0	0	1	118
ALL SPECIES	1252	223	2212	13665

Total catches of transient species in the tide creek were much greater in pull nets than in Breder traps, and active sampling gear such as pull nets, seines, and fyke nets are probably the best techniques for open water habitats. Pull nets, in particular, may be the only way to sample ditches with deep bottom mud deposits, where seines cannot be used or would cause an unacceptable amount of disturbance in the form of turbidity and hydrogen sulfide. However, pull nets have variable efficiency depending on the tide stage. Minimal sediment resuspension attracts forage species into the pull net transect, so repetitive samples are not possible.

Breder traps were most effective in vegetated marsh habitats. Throw net sampling was very destructive to high marsh vegetation, and square areas on the marsh surface denuded by throw net sampling remained barren for over a year. Furthermore, throw nets could not be used in very dense or woody vegetation.

The total number of fish collected in Breder traps was linearly correlated with soak time for up to three hours (Figure 2). Variation around the regression line suggests that 2-3 hour soak times might be the best choice. The four most common marsh species, <u>Cyprinodon variegatus</u>, <u>Fundulus confluentus</u>, <u>Gambusia affinis</u>, and <u>Poecilia latipinna</u> dominated the catch in most traps after only 45 minutes. <u>Eucinostomus</u> sp., <u>Gobiosoma</u> sp., and <u>Lucania parva</u> were the only other species collected during this experiment. Spatial resolution of the abundance of five marsh resident fish species was obtained with Breder traps (Figure 3). <u>Poecilia latipinna</u>, <u>Cyprinodon variegatus</u>, <u>Lucania parva</u>, <u>Gambusia affinis</u>, and <u>Fundulus confluentus</u> were most abundant in the inner lagoon (IL), tide creek (TC), transition (TR), and high marsh (HM). Relatively few fish were collected from the fringe (FR) or mixed (BR) zones.



<u>Figure 2</u>. Total catch (all species) vs. time in Breder trap "soak time" experiment in tidal creek, April 27, 1987.

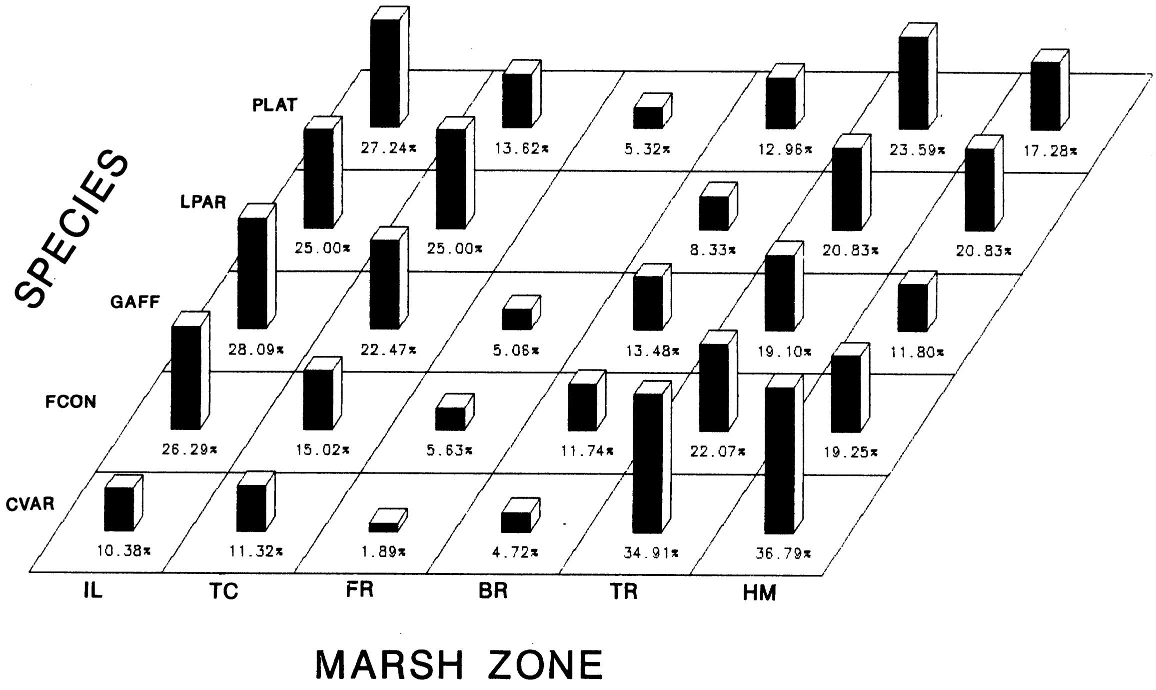


Figure 3. Spatial distribution of five resident fish species among zones of mangrove island, Indian River (Florida) lagoon. (See text for abbreviations of fish species and zones.)

The effort needed to sample the entire Breder trap array was less than that needed to conduct a pull net haul, seine haul, or throw net series. The major factor governing sampling time for an array of Breder traps was distance between traps and roughness of terrain. One person could sample the array of 24 traps in about 45 minutes. Two people needed 20 to 30 minutes to complete one pull net or seine haul. A series of throw trap samples could take one person an hour to finish.

We suggest modifying the original trap design to reduce breakage and simplify handling. Fixed funnel "wings" of Breder traps make the traps cumbersome and are easily broken. While broken traps can be easily mended with Plexite® cement in the field, we use removable wings on our traps. Triangular pieces of Plexiglass are glued to the inside top and bottom of the trap body, creating tracks for each wing of the funnel to slide in and out of the box. The traps can then be stored and transported with wings stored safely inside the box.

The gear bias of Breder traps is an important factor which must be taken into account any time they are used. Transient species, which are under-represented by Breder traps, are an important link between wetlands and estuaries. Breder traps would, therefore, be of limited value in estimating the densities of these economically important species.

CONCLUSIONS

Breder traps are a single method which can be used to compare relative densities of certain resident fish species in mangrove and salt marshes. They perform efficiently in even the densest vegetation while causing only minimal habitat disturbance. Statistical validity through replication is provided at reasonable cost and effort. Breder traps underestimate transient and some resident species which inhabit marshes, however, and a complete sampling scheme for resident and transient marsh fish species requires use of several gear types. If the Cyprinodontids and Poeciliids can be used as indicators of fish utilization of marshes and, thus, habitat function, Breder traps are an excellent method of sampling fish in these habitats. Further research should be done to test the effectiveness of Breder traps in *Spartina alterniflora* and *Juncus roemerianus* marshes. We conclude that Breder traps, within the limitations stated above, are useful tools in the functional assessment of natural and created wetlands communities.

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LACUSTRINE VEGETATION ESTABLISHMENT WITHIN A COOLING RESERVOIR

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ABSTRACT

The history of a large scale mitigation project of a cooling reservoir (L-Lake) for a reactor on the Savannah River Plant, South Carolina, is presented. The National Pollution Discharge Elimination System permit for thermal effluents discharged into the reservoir requires establishment of a balanced biological community (BBC). As a good faith effort toward establishment of a BBC, wetland/littoral vegetation was planted along 427 m (14,000 ft.) of shoreline in 1987. Approximately 100,000 plants were transplanted. Species planted were representative of submergent, floating-leaved, emergent and woody zones found in regional South Carolina lakes and reservoirs. The transplants have been growing well and reproducing, but project success will not be determined until spring and summer, 1988.

INTRODUCTION

The Savannah River Plant (SRP) is a nuclear production facility, operated by the Department of Energy (DOE), that produced weapons grade plutonium and tritium for defense purposes. This 780 km^2 (300 mi^2) facility is located in South Carolina along the Savannah River (Figure 1) and has five production reactors constructed in the mid 1950s. These reactors used the Savannah River as a source of secondary cooling water, and discharged thermal effluents into several back water tributaries of the Savannah River. Among the significant environmental effects of these thermal discharges are the lethal effects of hot (50-90 C) cooling water; increased water flow rates, water depth, erosion, and sedimentation (Sharitz et al. 1974; Gibbons & Sharitz 1981).

One of the reactors, L-Reactor, discharged its thermal effluents directly into Steel Creek between 1953 and 1968 (Figure 1). Thermal discharges increased stage and discharge from relatively natural levels of 0.3 m^{3}s^{-1} (10.5 cfs) to flows as high as 11 m^{3}s^{-1} (338 cfs). P-Reactor also discharged hot water directly into Steel Creek between 1958 and 1963. Flows as high as 24 m^{3}s^{-1} (847 cfs) were recorded when

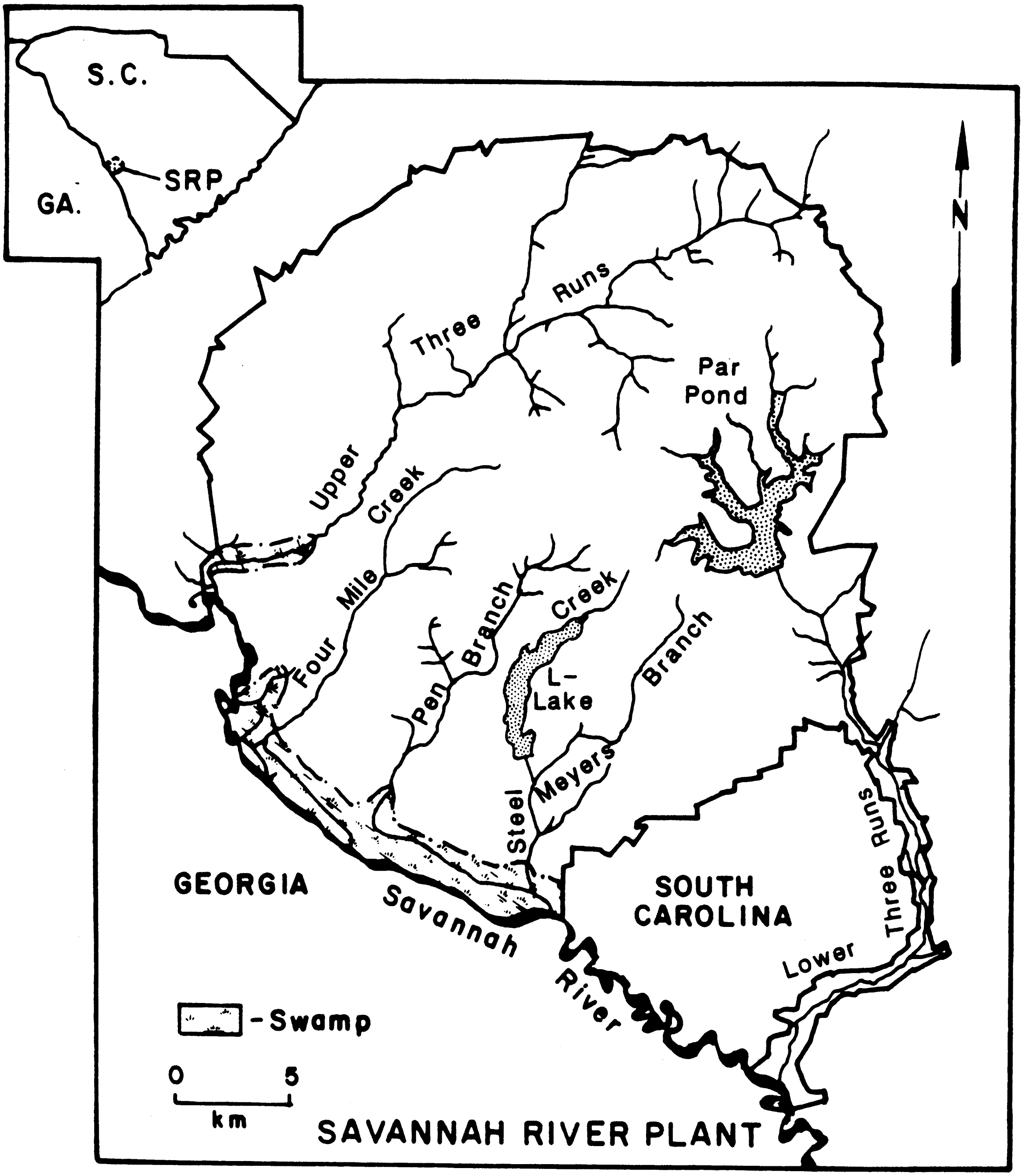

Figure 1. Location of the Savannah River Plant, South Carolina.

both reactors were operating simultaneously. Combined flows resulted in scouring of the stream channel and subsequent deposition of sediments in a delta where the creek enters the Savannah River. Sediments deposited within this delta are more than 1.0 m (3.3 ft.) thick (Smith et al. 1981).

In 1968, L-Reactor was placed on stand-by status due to a decrease in plutonium demand. Thermal discharges into Steel Creek stopped and flows returned to relatively natural levels. Partial recovery of the Steel Creek wetlands began to occur as scrub-shrub, persistent and nonpersistent emergent plant communities became established (Sharitz et al. 1974; Martin et al. 1978; Smith et al. 1981).

A decision to restart L-Reactor was made in 1980 due to an increased demand for plutonium. Under provisions stipulated in the National Environmental Policy Act, DOE was required to prepare an Environmental Impact Statement (EIS) for the project. In the EIS, alternatives to direct discharge of thermal effluents into Steel Creek were evaluated. In addition, a National Pollution Discharge Elimination System (NPDES) permit was required by the Clean Water Act.

A number of events, complicating the regulatory and mitigation options for the L-Lake project, occurred during preparation of the L-Reactor EIS. Biologists noted that the American alligator (Alligator mississippiensis) and the woodstork (Mycteria americana), both federally listed endangered species, used portions of the Steel Creek ecosystem as part of their home ranges. The Endangered Species Act called for scrutiny of potential losses of endangered species habitat. It was determined that if the L-Reactor project was to proceed, then in-kind mitigation of the loss of wetland habitat used by the storks would be required.

A total of 33 alternative cooling water systems were evaluated by DOE in the L-Reactor EIS (U.S. Department of Energy 1984). These cooling water systems included cooling lakes, cooling towers, and spray cooling systems. The alternative systems were grouped into five major categories: 1) direct discharge, 2) once-through cooling lake, 3) recirculating cooling lake, 4) once-through cooling tower, and 5) recirculating cooling tower. Criteria used to determine the preferred alternative included engineering feasibility, South Carolina water quality standards, production considerations, construction schedule, environmental effects, and cost.

The preferred alternative from the draft EIS was to return to the previous operating mode, with discharge of cooling water from L-Reactor directly into Steel Creek. However, the direct discharge alternative was changed in the final EIS because of public comments and the determination of the State of South Carolina that the NPDES permit regulations did not allow direct discharge of thermal waters.

In the final EIS, a once-through cooling lake (L-Lake) was identified as the preferred alternative. DOE acknowledged that environmental effects would have been less with some other alternatives. However, the need to start L-Reactor as soon as possible and the longer construction times of the other alternatives made them less desirable. Federal and State regulatory agencies approved the once-through cooling lake alternative and construction began in autumn of 1984. L-Lake was filled in October 1985 and L-Reactor was restarted.

Construction of L-Lake changed the structure and function of approximately one-third of the upper and middle portions of Steel Creek from a riverine to a lacustrine ecosystem. L-Lake covers 405 ha (1000 acres), is 7,000 m (23,000 ft.) long and 1,200 m (3900 ft.) wide at its widest point (average 600 m wide), and has a capacity of 31 million cubic meters (41 million cubic yards). A 1,200 m (3900 ft.) dam is located at the lake's southern end.

During construction of the lake and dam, approximately 418 ha (1,034 acres) were clear-cut, including 144 ha (356 acres) of bottomland hardwood and shrub wetlands, 145 ha (360 acres) of upland hardwood and pine forests, and 50 ha (125 acres) of other areas within the lake basin. Outside of the lake basin an additional 78 ha (193 acres) were clear-cut for powerline rights-of-way and other construction related sites. Downstream from the dam raised water levels, asynchronous flows, mild thermal loading, and sedimentation associated with dam construction and the overall increase in flows (back to 1 m^3s^{-1}) perturbed the post-thermal recovery of approximately 314 ha (775 acres) of wetlands (P. H. Brownell, unpublished data).

Regulatory agencies required three different mitigation components to obtain permits. The first required construction of woodstork foraging habitat at Kathwood Lakes near the SRP. This consisted of the construction of a 14 ha (35 acres) fish pond where the water levels are raised and lowered to concentrate fish in order to facilitate the tactile foraging of woodstorks. This project has been very successful; up to 97 woodstorks use the foraging area daily. The second component required setting aside habitat in proportion to habitat lost during the construction of L-Lake. The value of habitat lost was determined by using the U.S. Fish and Wildlife Service Habitat Evaluation Procedures (HEP). This continues and will result in the preservation of upland and wetland habitats on the SRP. The third component, and the one on which this paper will focus, is the establishment of a Balanced Biological Community within L-Lake.

BALANCED BIOLOGICAL COMMUNITY

Section 316a of the Clean Water Act requires that effluent limitations be imposed which "assure the protection and propagation of a balanced, indigenous population of shellfish, fish, and wildlife in and on that body of water" when thermal discharges could cause a significant environmental impact. The lake was designed so that water temperature at the dam outfall would comply with the State's requirement of less than 32.2° C, while the upper half of L-Lake could reach much higher temperatures (45-60° C) during reactor operation. Hence, only the lower 33 and 50 percent of the lake could support flora and fauna native to the region. According to the NPDES permit, L-Lake must support a "Balanced Biological Community" (BBC) with the following characteristics:

a. the lake must not be dominated by thermally tolerant species;

b. the lake must support biotic diversity and productivity that are similar to other lakes in the region;

c. the lake biota must include representatives of all trophic levels that are typical of lakes in the region;

d. the biotic communities of the lake must be self-maintaining and not require restocking.

Efforts have been made by DOE to enhance physical structures that could provide suitable habitat for fish and wildlife. These include construction of artificial reefs, coves and small bays, forested stands, and the establishment of wetland vegetation. All of this will aid in establishment of a BBC. The University of Georgia's Savannah River Ecology Laboratory (SREL) was asked by DOE to manage the establishment of wetland/littoral vegetation along the shoreline of L-Lake and initiate research in some aspects of wetland creation.

Wetland vegetation, characteristic of regional lakes near the SRP, can develop naturally. For example, a well developed herbaceous wet-land fringe developed naturally around Par Pond (Figure 2), a

Figure 2. Herbaceous wetland fringe along shoreline of Par Pond, a recirculating cooling reservoir located on the Savannah River Plant.

recirculating cooling reservoir on the SRP, since the pond's construction in 1958. Par Pond recently has been designated a BBC by the South Carolina Department of Health and Environmental Control (SCDHEC). Why, then, establish a wetland/littoral zone vegetation in L-Lake when it will develop naturally? The NPDES permit will be evaluated by SCDHEC in 1988 and does not provide sufficient time for establishment of wetland vegetation through processes of natural succession. As a good faith effort toward establishment of a BBC, DOE decided to accelerate natural succession by planting wetland vegetation along the shoreline within the cooler southern end of L-Lake. The establishment of wetland/littoral vegetation should provide habitat for fish, wildlife and insects, organic matter for soil development and decomposers, substrate for epiphytes, and a primary producer trophic level. Wetland and vegetation can play important roles in nutrient cycling (Godshalk & Wetzel 1978), sediment retention (Mitsch & Gosselink 1986), and shoreline stabilization (Allen 1978) and is a major component of a BBC.

WETLAND ESTABLISHMENT

A panel of experts in wetlands ecology and establishment was asked to visit the L-Lake site and Par Pond and make recommendations for development and management of the wetland habitats of L-Lake. The panel proposed that five zones of vegetation be established which are representative of species zonation patterns found in Par Pond and natural lakes (Figure 3). It was determined that Par Pond could serve as the primary source of plant material because its vegetation has been exposed to elevated thermal conditions and it is close to L-Lake (Whigham et al. 1985). Species recommended for establishment in L-Lake are those that:

1. can become established quickly and spread vegetatively;
2. have value for fish and wildlife;
3. are available locally or can be obtained easily;
4. are components of wetland/littoral vegetation in the area;
5. are native (not introduced) species;
6. have the potential to tolerate elevated temperatures;
7. have a high potential for shoreline stabilization; and
8. will trap sediments and organic matter.

Major limitations to successful vegetation establishment were identified (Whigham et al 1985). These included steep slopes, fluctuating water levels, and low nutrient substrates. The graded slope of the shoreline along portions of L-Lake is too steep and may not be conducive to the development of desirable widths of wetland/littoral

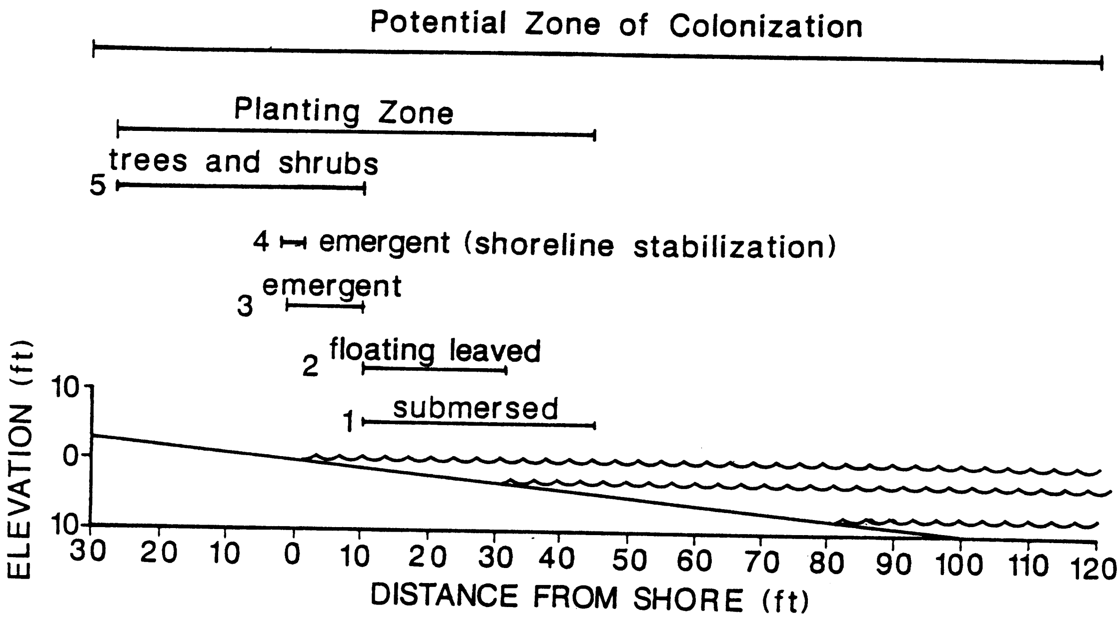

Figure 3. L-Lake planting zones (zones 1-5) and vegetation types. Horizontal distance varied with shoreline topography.

zones. The panel suggested additional grading of slopes but this turned out to be unfeasible after the lake had been filled. Large fluctuations in water level would probably result in the death of transplants, especially during the growing season. High water will also increase shoreline erosion; therefore DOE was requested to keep water levels from fluctuating more than 0.15 m (0.5 ft). Finally, the sandy substrate contains low levels of nitrogen and phosphorus which are nutrients critical to plant growth. It was suggested that all transplants be treated with a slow release fertilizer. In addition to identifying limitations that may hinder plant establishment, the panel developed three planting options:

1) no action: let the wetland become established naturally;

2) block planting: plant in selected areas of the shoreline; and

3) continuous planting: plant along all available areas of shoreline.

The third option was chosen because it would result in rapid stabilization of shoreline and development of habitat.

The University of Georgia's Savannah River Ecology Lab (SREL) manages the wetland establishment project for the Department of Energy. SREL hired a private contractor (Southern Tier Consulting) to implement the project. Transplants, shrub wands and commercial plants were planted in L-Lake from January to April 1987. During the planting period more than 4,270 linear meters (14,000 ft.) of shoreline were planted. Approximately 100,000 plants were installed with 15% in zone

1, 15% in zone 2, 50% in zone 3, and 20% in zone 5 (Figures 4 & 5). The shoreline stabilization zone (zone 4) will be planted during summer 1987. Details on species planted for each of the five zones, source of plant material and planting density are provided in Table 1.

Figure 4. *Polygonum* sp. planted within zone 3 along shoreline of L-Lake.

Figure 5. *Eleocharis quadrangulata*, *Juncus effusus*, and *Typha latifolia* planted in zone 3 with *Nymphaea odorata* planted in zone 2 along the shoreline of L-Lake.

Table 1. Description of plant material to be planted at L-Lake.

Zone	Plant Species	Density[1]	Type of plant material to be transplanted
1	-*Vallisneria americana*	4	Weighted cheese-cloth packets of bare root plants
2	-*Brasenia schreberi*	4	6-10" rhizome with shoot
	-*Nelumbo lutea*	4	Terminal meristem
	-*Nymphaea odorata*	4	6" rhizome with meristem
	-*Potamogeton diversifolia*	4	2" rhizome and shoot
3	-*Eleocharis quadrangulata*	2	Tuber or node
	-*Panicum hemitomom*	2	8" rhizome with shoot
	-*Pontederia cordata*	2	Bare root plant
	-*Sagittaria latifolia*	2	Corm[2]
	-*Scirpus cyperinus*	2	Rhizome section with shoot
	-*Typha latifolia*	2	Bare root rhizome
	-*Bacopa caroliniana*	2	10" rooted stem pieces
	-*Eleocharis equisetoides*	2	8" section of rhizome
	-*Juncus effusus*	2	Bare root clump
	-*Juncus acuminatus*	2	Bare root clump
	-*Carex comosa*	2	Bare root clump with shoot
	-*Lycopus rubellus*	2	Bare root rhizome
	-*Polygonum* spp.	2	10" stem section with roots
4	-*Leersia oryzoides*	1.5	Bare root stems
	-*Panicum hemitomon*	-	3 ft. rhizome sections
	-*Panicum* sp.	-	Seed[2]
	-*Glyceria* sp.	-	Seed[2]
5	-*Cephalanthus occidentalis*	10	14-16" wands
	-*Salix Nigra*	10	14-16" wands
	-*Alnus serrulata*	cluster	14-16" wands
	-*Nyssa sylvatica*	10	2-3' bare root plants[2]
	-*Taxodium distichum*	10	2-3' bare root plants[2]

[1]density number is distance between individual plants and represents plants placed on foot centers
[2]commercial source for plant material

Planting success will be determined in July and August 1987. Random areas of planted shoreline will be selected, and belt transects, perpendicular to the shore, will be used to measure plant survival in each zone. The minimum acceptable transplant survival percentages for contract compliance are: zone 1, 10%; zone 2, 30%; zone 3, 50%; zone 4, 50%; and zone 5, 30%. A few random areas selected in 1987 will be used as permanent plots to monitor plant colonization.

PROBLEMS

Problems encountered since the initiation of the L-Lake project can be grouped into three categories: 1)hydrological, 2) biological, and 3) logistical. Hydrological problems include water level fluctuations, wave energy, and thermal effluents. Water levels have fluctuated a maximum of 0.55 m 1.8 ft.) since March 1987, which during periods of low water has exposed large areas of shoreline. High water and winds have caused soil erosion and uprooted plants. Water level fluctuations have made it difficult to determine the boundaries of each zone, and periods of high water have slowed the planting process. The period of lowest water occurred just after the contractor finished planting in the spring and may have benefited the emergent species in zone 2 by allowing oxygen to reach the roots. There have been no problems with thermal effluents because L-Reactor has not operated continuously since the beginning of the planting.

Biological problems have included determining the proper time to transplant, herbivory by coots and beaver, and the presence of a pathogenic amoeba, Naegleria fowleri. Cattails planted early in the spring had higher mortality than individuals planted in April, suggesting that a later planting time should have been used. In addition, some nonpersistent plants (e.g., Potamogeton spp. and Brasenia schreberi) that were not available during the spring will be planted this summer. Almost all the Sagittaria latifolia corms were eaten by coots, and some of the Salix branches have been eaten by beaver. A pathogenic amoeba, N. fowleri, has been isolated from the waters of L-Lake and can cause Primary Amoebic Meningoencephalitis, a rare but fatal human disease. To prevent entry into the nose, surgical masks and gloves are worn by all individuals working in boats or along the shore of L-Lake.

Logistical problems have resulted primarily from the magnitude of the project. These problems include obtaining and transporting plant material, poor access along the shore of L-Lake, and acquiring skilled labor. Plant material was collected by boat at Par Pond, transported to L-Lake and then stored before being installed. Transporting plant material often took more time than installation and the planting crew often had to wait for material. The average planting rate was approximately 200-300 plants per day per person, collected and installed.

Obtaining skilled labor was a major problem. A temporary employment agency was used to obtain labor. Employees often found the work to be wet, cold and generally uncomfortable. Absenteeism was high and employment periods short. This resulted in consistently shorthanded planting crews and the need to hire and train new employees.

Phase one of the planting was completed in April 1987, three months after being initiated. Plant material that was unavailable during the spring months will be planted in July 1987. Areas where excessive plant mortality has occurred also will be replanted during this time.

RESEARCH

The introduction of wetland plant species into L-Lake provides an opportunity to study wetland establishment processes and determine the advantages of introducing transplanted species that accelerate plant succession in a lacustrine wetland. Research which addresses plant colonization, the establishment of seed banks, changes in shoreline geomorphology, and soil development is being implemented.

Successional changes in vegetation are being monitored in planted and unplanted areas. A species list of plants found and introduced around the shore is being developed. Wetland plant colonization will be monitored by photographs at permanent shoreline markers and by sampling vegetation in permanent transects and plots. At one end of each transect a permanent bench marker will be used to measure changes in shoreline morphology.

Introducing transplants into L-Lake may accelerate plant colonization by asexual means, but will also spread plants into unplanted areas by seed. A seedbank in L-Lake is presumed nonexistent because riparian wetland and terrestrial soils were disturbed during construction of the shoreline. A lacustrine seedbank will become established from the wetland vegetation. The development of the seedbank in L-Lake will be one focus of research.

Six research areas have been established in the southern portion of the lake which provide the opportunity to study some attributes of species biology. Two species, *Typha domingensis* and *Pontederia cordata*, have been planted along an elevational gradient. Above and below, ground biomass and seed production will be used to determine at which water depth these species grow best. Other research projects include the development of a phosphorous budget, larval fish associations with different vegetation structural types (e.g., floating-leaved vs. emergent stems) and bird use of developing habitat.

CONCLUSIONS

The establishment of wetland/littoral zone vegetation on L-Lake has resulted in the installation of more than 100,000 plants along 4270 m (14,000 ft.) of shoreline. Initial survivorship of installed plants has been high. However, it is difficult to evaluate long-term success only four months after project initiation. Success will be evaluated during the summer, 1987. In addition, research into wetland establishment and long-term success of wetland plantings has been initiated.

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