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Proceedings of The 15th Annual Conference on Wetlands Restoration and Creation

May 19 - 20, 1988
Tampa, Florida

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Hillsborough Community College
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PROCEEDINGS OF THE FIFTEENTH ANNUAL CONFERENCE ON WETLANDS RESTORATION AND CREATION

May 19-20, 1988

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INTRODUCTION

The Annual Conference on Wetlands Restoration and Creation provides a forum for the nationwide exchange of results of scientific research in the restoration, creation, and management of freshwater and coastal systems. The conference is designed to be of particular benefit to governmental agencies, planning organizations, colleges and universities, corporations, and environmental groups with an interest in wetlands. These proceedings are a compilation of papers 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, fifteen years ago. We are grateful for his help and participation. Appreciation is also extended to Charles Deusner for providing administrative support for the conference.

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

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DIVERSIFICATION IN WETLAND MITIGATION:
A CASE STUDY - COPPERCREEK DEVELOPMENT PROJECT

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ABSTRACT

To offset the loss of 2.2 acres of persistent emergent wetland habitat impacted as a result of a 200 acre development project in Farmington Hills, Michigan, a 2:1 mitigation replacement ratio was required by the Michigan Department of Natural Resources (MDNR) under the Goemaere-Anderson Wetland Protection Act to assure "no net loss" of wetland habitat. To meet the MDNR permit requirements, a mitigation design was introduced which would replace and enhance the functional integrity of the wetlands impacted through diversification of wetland habitat. For the Farmington Hills development project, 4.4 acres of wetland habitat containing three individual wetland vegetational communities (persistent emergent, scrub-shrub, and open water) was designed and created in an upland area contiguous to an existing wetland system and stream. This was accomplished through manipulation of slope, soil, and hydrology, and the application of native and non-native wetland seed mixes. Vegetational succession, functional values, and overall habitat quality will be monitored to determine if diversification in wetland mitigation design/creation increases the diversity of wetland functional attributes and if the mitigation ratio requirements as proposed by the MDNR result in "no net loss" of wetland habitat.

INTRODUCTION

In 1980, the Michigan Department of Natural Resources (MDNR) was given authority to administer the Section 404 requirement of the federal Clean Water Act. Subsequently, the State of Michigan promulgated and adopted the Goemaere-Anderson Wetland Protection Act No. 203 P.A. 1979 to "provide for the preservation, management, protection, and use of wetlands, to require permits to alter certain wetlands; to provide for a plan for the preservation, management, protection, and use of wetlands; and to provide remedies and penalties." General conditions within the act which provide for state regulation of wetland habitat are summarized below:

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A wetland or portion of a wetland:

1) Greater than 5 acres in size in a county with greater than 100,000 population;

2) Contiguous to an inland lake or pond, river or stream, one of the Great Lakes or Lake St. Clair; and

3) Less than 5 acres in size in any county, not contiguous but determined by the MDNR that protection of the area is essential to the preservation of the natural resources of the state from pollution, impairment, or destruction.

Except as otherwise provided by this act or by a permit obtained from the department, a person shall not:

1) deposit or permit the placing of fill material in a wetland;

2) dredge, remove, or permit the removal of soil or minerals from a wetland;

3) construct, operate, or maintain any use or development in a wetland; and

4) drain surface water from a wetland.

The above provisions have caused some confusion as to the definition of certain terms used in the act. These terms include inland lake or pond, river or stream, and contiguous. Common issues which have received major criticism include the criteria for wetland determination, the permit application process, prudent and feasible alternatives, and mitigation considerations. Rules which are intended to clarify these and other items are currently proposed for review and comment by the MDNR. If adopted, the rules may benefit both the public and private sectors through a better understanding of the intent of the act.

Wetland Mitigation in Michigan

The MDNR's policy on wetland mitigation has changed dramatically since the state wetland protection act became effective October 1, 1980. Initially, acceptable mitigation consisted of purchasing land which contained regulated wetland habitat and merely preserving it or enhancing contiguous or off-site wetland habitats of low quality or in regression. When "no net loss" of regulated wetland habitat became an issue, replacement ratios were set which consisted of 1:1 (acre for acre), 2:1, or 3:1 replacement for scrub/shrub, persistent emergent and open water habitats, respectively. However, many forested wetlands were considered by the MDNR to be of relatively lower quality and more abundant (more than two-thirds of the wetland remaining in Michigan) with a required replacement ratio of only 0.5:1. The "no net loss" issue for any regulated wetland habitat became a necessity when goals set to accomplish successful wetland mitigation for 0.5:1 forested wetland mitigation projects (by creating "ponds" consisting of primarily open water wetland habitat considered by the MDNR to be of higher quality because of use by waterfowl) were not being achieved in accordance with the proposed mitigation plans.

With regard to the current status of the MDNR's wetland mitigation policy, it is the department's position that ". . . may impose conditions on a permit for a use or development if the conditions are designed to remove an impairment to the wetland benefits, to mitigate the impact of a discharge of fill material, or to otherwise improve the water quality." The proposed rules, which incorporate more detailed mitigation definitions and guidelines are, for the most part, currently followed by the MDNR and are summarized in the flow diagram shown in Figure 1. From this figure, it can be seen that the key elements in determining the necessity for potential mitigation efforts are the issues of prudent and feasible alternatives (to include avoiding the impact(s) altogether through "no action") and wetland dependency of the proposed project or activity. Only until these issues are addressed will unavoidable impacts and potential wetland mitigation alternatives be considered by the MDNR. At this point it is imperative that wetland mitigation become an acceptable alternative only when the wetland(s) impacted by the proposed development project or activity can be replaced in a practical and feasible manner with a "no net loss" of wetland habitat and, where possible, be provided on-site where practical and beneficial to the wetland resources. Despite these mitigation considerations, 95% of wetland permits issued by the MDNR require no mitigation efforts. Most impacts that are mitigated have not been successful, therefore, net loss of wetland habitat is still occurring. Because of the lack of success of mitigation projects thus far, the MDNR lacks confidence in reviewing and authorizing the implementation of mitigation design plans. However, it is our opinion, and the purpose of this paper to show, that mitigation of certain wetland habitats can be accomplished by utilizing specific design criteria intended to increase the functional values and overall habitat quality through diversification of habitat structure. We believe and attempt to demonstrate that this can be accomplished through manipulation of slope, soil, and hydrology along with the application of native and non-native wetland vegetation seed mixes. Incorporating diversification of wetland habitat in the mitigation design/creation plan will help to assure diversification of wetland functional attributes so that the mitigation ratio requirements, as proposed by the MDNR, are likely to be satisfied and result in "no net loss" of wetland habitat from the proposed development activity.

Justification for Diversification in Wetland Mitigation

There is currently a great deal of literature which addresses the importance of habitat diversity in increasing wetland functional values. Jacobs (1975) notes that, in general, the greater the diver-

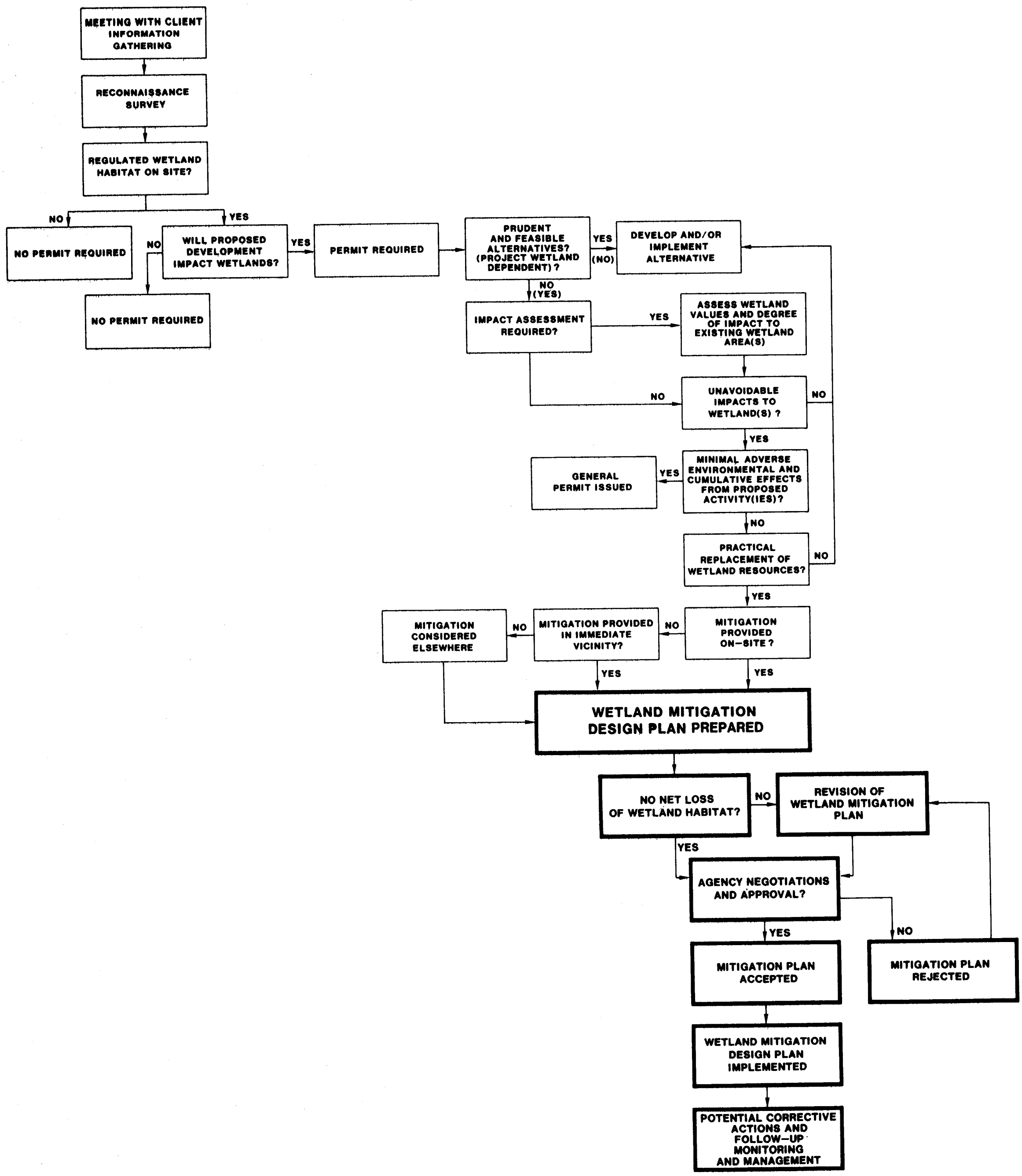


Figure 1. Flow diagram showing general wetland permit conditions and mitigation considerations under Michigan's Goemaere-Anderson wetland protection act (Act 203, P.A. 1979).

sity of wetland types, the greater the number of biotic niches, and therefore the more opportunities for species invasion. Larson and Neill (1986) state that heterogeneity of wetland types creates habitat diversity, which, in turn, results in high species richness of wildlife and that overall wildlife and ecological values will be enhanced by the induced habitat diversity. Beecher (1942) demonstrated that the number of bird nests was positively correlated with the number of plant communities in marshes. Individual plant species also appear to play an important role as specific food sources for a variety of wildlife (Weller 1978). Steel et al. (1956), Weller and Spatcher (1965), and Patterson (1974) all suggest that plant community interfaces or cover/water edges often are key habitat features to which birds respond and that few birds use dense and uninterrupted stands of vegetation. MacArthur and MacArthur (1961) and MacArthur et al. (1962) found that, in general, the more complex the habitat structure, the greater the number of bird species present and that habitats with permanent water have a higher diversity of bird species than do similar habitats without water (MacArthur 1964). Since life form of vegetation determines the structural character of the habitat for animals (Golet & Larson 1974), diverse vegetation forms attract a variety of faunal species (Jaworski & Raphael 1979) which is not limited to just avian use. Finally, it appears that wetland size may be of little consequence to diverse wildlife use provided that habitat diversity is retained and maintained (Weller 1978). Niering and Kraus (in Larson & Neill 1986) indicate that the overall goal of mitigation should be to recreate a wetland in a setting that is hydrologically sound and that will maintain a vigorous wetland plant community with high productivity. Establishing a wetland with a multi-functional role should be stressed, since the ability to create a wetland that serves a specific function with respect to vegetation is limited.

The aforementioned literature suggests that diversity of habitat yields diversity of wildlife and most likely other wetland functional values, and it is this widely accepted ecological principle upon which the Coppercreek wetland mitigation project is based.

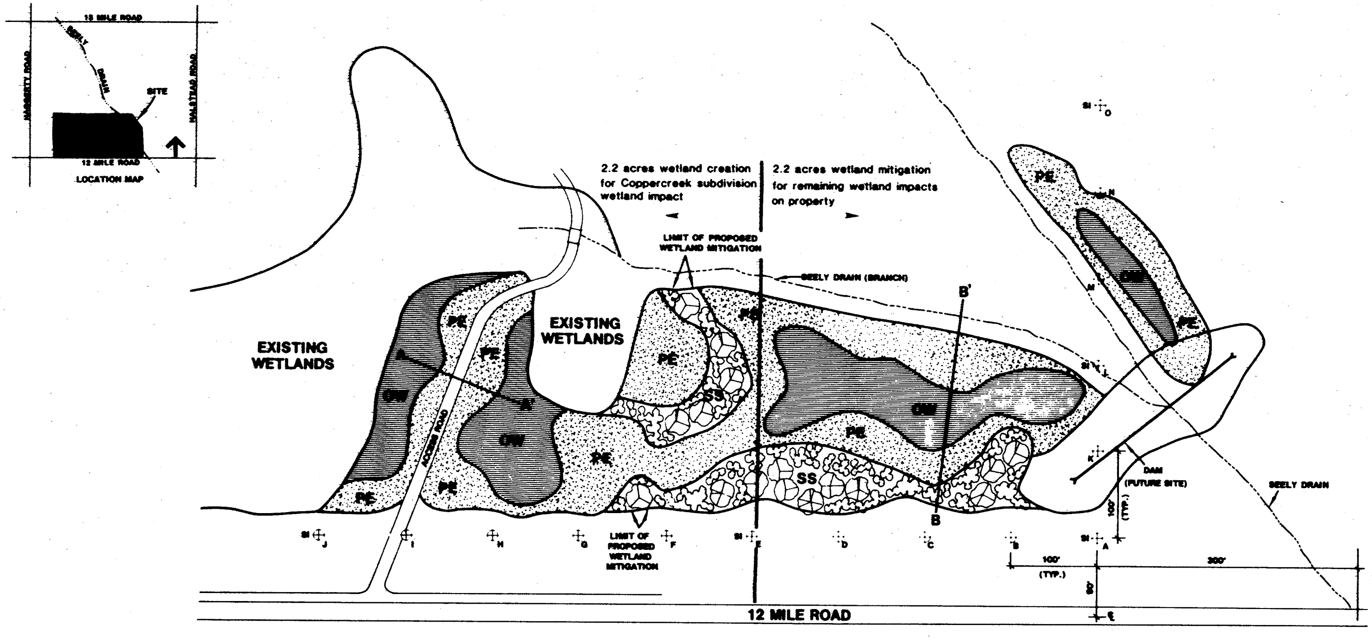
THE COPPERCREEK STUDY SITE

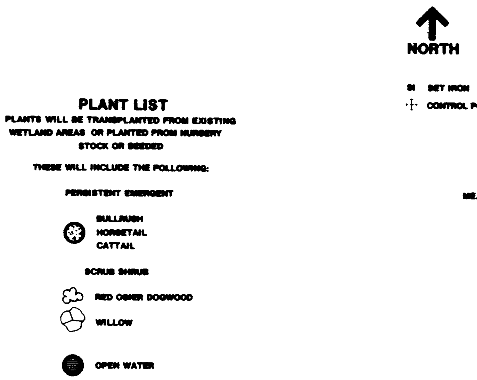
The Coppercreek study site consists of approximately 200 acres located in a residentially zoned area in Farmington Hills, Michigan (Oakland County, Section 28, T. 1 N., R. 9 E.). The property was purchased by Biltmore Properties Corporation, a major home developer in Michigan, for the purpose of developing a 9-hole golf course and both single family and cluster residential units. A wetland assessment performed on the property identified approximately 35 acres of regulated wetland habitat, including forested, scrub/shrub, wet meadow, persistent emergent, and open water areas. Prior to designing a development plan for the property, sensitive and relatively high quality (unique and functionally valuable) wetland habitats were delineated and proposed as "unmitigatable"; these included primarily forested, scrub/shrub, and open water habitats, which totaled approximately 50% or 17 acres of the total wetland habitat identified. The remaining wetland habitat on the property, mostly wet meadow and persistent emergent, was considered as potentially "mitigatable," as long as the impacted areas were limited in size (<5 acres) and encroached only along the perimeter of the habitats. The final approved development plan for the Coppercreek project avoided all but 2.2 acres of persistent emergent and wet meadow habitat. Construction impacts to existing wetland habitat consisted of a few single family lots, golf course tees and greens, and a bicycle path.

To offset the loss of the 2.2 acres of persistent emergent and wet meadow wetland habitat (qualitatively considered to be of relatively low to medium quality) to be impacted as a result of the development project, a 2:1 mitigation replacement ratio was required by the MDNR to assure "no net loss" of wetland habitat.

COPPERCREEK MITIGATION DESIGN PLAN

To meet the MDNR mitigation ratio requirements and replace and enhance the functional integrity of the wetland habitats impacted, a mitigation design was introduced which would enhance the functional quality of created wetland through diversification of habitat design (Figure 2). For the Farmington Hills development project, 4.4 acres of wetland habitat containing three individual wetland vegetational communities (i.e., persistent emergent, scrub-shrub, and open water) were designed to be created in an upland area contiguous to an existing wetland system and stream. This was accomplished through slope, soil and hydrologic manipulation, and the application of native and non-native wetland seed mixes. The purpose of the mitigation plan designed by the Coppercreek project was not only to prevent a "no net loss" situation, but also to replace and enhance the functional integrity of the wetland habitats impacted by expanding the functional quality of created wetland through habitat diversification. The final mitigation design plan approved by the MDNR consisted of creating 1.4 acres of open water, 2.2 acres of persistent emergent, and 0.8 acres of scrub-shrub wetland habitat. These habitats were intended to be created adjacent to a stream and existing forested, scrub-shrub, and persistent emergent wetland habitat. The existing contiguous wetland habitats indicated that this site could provide a suitable hydrologic regime as well as allow vegetational encroachment from adjacent native wetland plants and provide a potential seed source, buffer zone, and source of aquatic organisms and wildlife, desirable for a successful establishment and use of the created wetland habitats. The wetland habitats in the Coppercreek mitigation plan were designed with the intent to create habitat for waterfowl and avian feeding, nesting, resting, and rearing, as well as aquatic organisms such as macroinvertebrates, plankton, amphibians, and reptiles which would provide food chain support. In addition to the increased wildlife value, the created wetland habitats would provide additional flood storage, sediment trapping, water quality enhancement, and nutrient retention capacities to the existing contiguous wetland habitats.





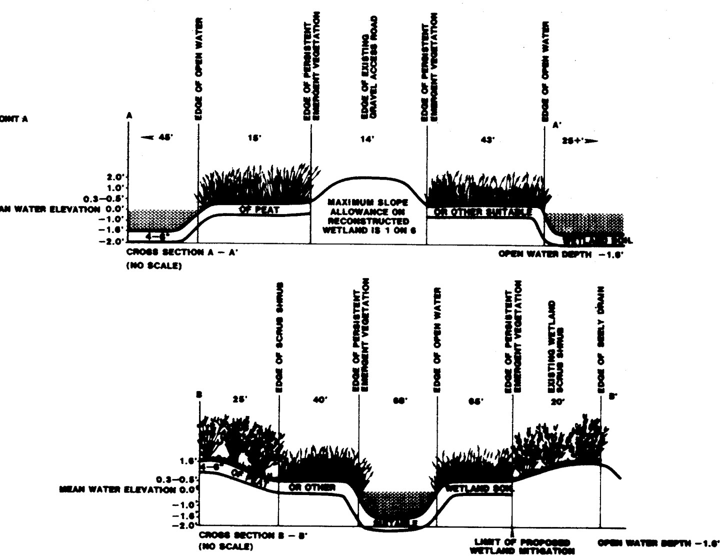


Figure 2. The Coppercreek wetland mitigation design plan illustrating diversification of proposed vegetational communities contiguous to existing wetland habitat.

According to Garbisch (1977) the two most important factors to be considered when preparing a site for marsh establishment are surface slopes and surface elevations; these factors were also considered to be of primary importance in designing wetland habitats to establish a potentially diverse habitat association, the implementation, construction, and development of which are described in the following section.

IMPLEMENTATION/CONSTRUCTION/DEVELOPMENT

Construction of the Coppercreek mitigation design was initiated on July 6, 1987 and actual physical construction was completed on September 9, 1987. In accordance with MDNR permit requirements, construction of the mitigation wetland was to be initiated before project development, with all construction activities supervised by experienced wetland ecologists.

Prior to construction, a filter-fabric fence was used to delineate created wetland boundaries and to prevent impacts from construction activities to the existing contiguous wetland habitats and stream. Surface ground elevations were surveyed at the proposed wetland edges and within the specific wetland habitats to determine excavation depths required to maintain continuous hydrologic infiltration from groundwater and surface water runoff. The proposed wetland site and surrounding area contained a near surface aquifer which maintained contiguous wetlands and streams and thus provided an excellent source of water for the created wetlands. In addition, subsurface soils consisted of silt, clays, and marl in which to help capture and retain surface water runoff.

Surface water elevations of the contiguous stream were used to determine the depth at which to excavate soil in order to create the various wetland habitats and to establish expected water elevations in the created areas. These stream elevations were measured during the assumed lowest level in July. Excavation depths in the created wetlands varied from 1 to 6 feet. Water depths in the created open water wetland areas were designed and excavated to range from no less than 1.5 feet during dry periods (summer) to no more than 3 feet during high periods (spring runoff) as required by the City of Farmington Hills. Persistent emergent areas were designed and excavated to contain fluctuating water levels from 0 to 1.5 feet in depth, with the low water period elevation being at the expected low groundwater level. Scrub/shrub areas were designed and excavated to contain no more than 0.5 feet of water and no less than 1 foot below the ground surface during dry periods.

Excavated depths in the created wetland areas were 6 inches below final grade elevations to allow for the placement of peat material removed from impacted wetlands. It is expected that the peat mix will provide a seed source for vegetative establishment in the created wetland areas. Larson and Neill (1986) indicate that the use of dredged wetland soils as a seed source avoids the problem of importing plants from unsuitable sources, reduces the potential of importing unwanted species and is far less expensive than planting. It is also suggested that rapid response of native plants appears to be common and this has been interpreted as strong evidence of replication success.

In an attempt to maintain diversity of habitats, the Coppercreek mitigation plan was designed to incorporate fluctuating water levels associated with seasonal surface and groundwater fluctuations. It is our opinion that this variation in water levels should eventually help establish and maintain a diverse vegetational association. The hydrogeology of the Coppercreek site provided an excellent opportunity for creating a variety of wetland habitats within a relatively small area.

Due to space limitations (property boundaries), slopes greater than 1:6 could not be used; however, slopes generally should be maximized where possible to greater than 1:6 with 1:10 as a potential guideline value. Slopes in the Coppercreek wetland mitigation plan were designed to allow for the establishment and transition of submerged/floating/persistent emergent/scrub-shrub vegetation. To minimize erosion and stabilize the soils after construction of the wetland habitats, slopes were hydroseeded with barnyard grass (*Echinochloa crusgalli*) and winterwheat (*Triticum* sp.).

Initial vegetational development of the created wetlands in late summer of 1987, included horsetail, cattail, and smartweed, which occurred only in the persistent emergent zone. No submerged or floating vegetation was noted in the open water areas; however, considering the season, this was not unexpected. Wildlife use consisted of mallard ducks (*Anas platyrhynchos*), Canada geese (*Branta canadensis*), and leopard frogs (*Rana pipiens*).

Due to the time of seeding (late September) and an early frost just after germination, much of the hydroseed did not grow the following spring (1988); therefore, the slopes were re-seeded to assure erosion control. Some erosion has taken place on the site, but it has been minimal and in fact it has created an even more diverse and erratic shore zone. The erosion has also created a varied sediment deposition which may assist in diversifying vegetational establishment.

In an attempt to augment wetland successional development, 10 pounds of a wetland sedge-wildflower seed mix obtained through Applied Ecological Services, Inc. in Juda, Wisconsin was purchased and applied to the persistent emergent elevations of the created wetland areas. The seed was dispersed by hand using wind and water currents as dispersal aids in May, 1988 after the threat of frost had passed. As Garbisch (1977) indicates, marsh establishment by seeding is considered feasible only in the spring. Approximately 6 of the 10 pounds were dispersed in the low lying persistent emergent zones which, at the time of seeding, were wet mud flats. The mud flats aided in the adhering of seeds and provided a moist environment for seed germination.

The seed mix was harvested from wetland habitats in Rock County, Wisconsin in early August of 1987 by combine. There are at least 15 species of sedges, grasses and wildflowers included in this mix. Plant species most prevalent in this seed mix are the first several listed; all other species are present in minor amounts and included the following:

SEDGES

Carex vulpinoidea	*C. annectens*	*C. stipada*
C. prairea	*C. Sartwellii*	*C. muskingumensis*
C. Crawfordii	*C. tenera*	*C. normalis*
C. brevior	*C. comosa*	

GRASSES

Leersia oryzoides	*Agrostis alba*	*Poa pratensis*
Elymus canadensis		

REEDS

Juncus dudleyi	*J. tenuis*	*Lythrum alatum*
Penthorum sedoides	*Lobelia cardinalis*	

The seed mix was used in an attempt to establish non-native species, although wetlands in Wisconsin could be considered in the same ecoregion as Michigan. Planting or transplanting of vegetational species (especially scrub-shrub and tree species) was not implemented primarily due to costs.

At the time of this writing (first spring since creation), vegetational encroachment into the created areas has been limited, although it is much too early to observe any obvious signs. However, wildlife use appears to be more diverse with use of the habitats by mallard ducks, killdeer (*Charadrius vociferus*), spotted sandpipers (*Actitis macularia*), bank swallows (*Riparia riparia*), great blue heron (*Ardea herodias*), green-backed heron (*Butorides striatus*), and nesting geese.

We would like to note that the final cost for the developer to create this wetland habitat, purchase the seed, and monitor the progress to date has exceeded approximately $250,000.00.

GOALS

To monitor the progression of wetland habitat development as proposed in the mitigation design plan, we will continue to qualitatively assess and photodocument successional development through vegetational changes and species use. The types of vegetation established in each area and at what time in the development they occur will also be noted.

Our goal is not only to establish a diverse high quality wetland habitat that results in "no net loss," but to establish a wetland that, although artificially created, proceeds along a natural wetland successional development path that requires no management. Through vegetational encroachment and seed dispersal from contiguous wetlands, seeds present in overlain peat, artificial seeding, and other seed invasion routes, we hope to establish the beginning of a diverse persistent emergent, submergent, and floating vegetational community within 2 to 3 years, with scrub/shrub habitat developing within 3 to 5 years, and finally the occurrence of tree species within 5 to 10 years. The created wetland should be an area that under future expectations and regulatory definitions be an unmitigatable wetland.

CONCLUSIONS/RECOMMENDATIONS

A key issue to be addressed regarding the necessity for wetland mitigation is to determine whether or not there are prudent and feasible alternatives to the proposed development project or if the project is wetland dependent; these issues are not easily resolved. It must also be realized by the developer, regulatory agencies, and the general public that, should mitigation be considered, not all wetland areas can be successfully mitigated to the point where habitat loss and functional values can be compensated for by simply increasing the mitigation replacement ratio. As Banner (1979) notes, the practicality of mitigating wetland losses decreases as habitat quality increases. A wetland considered to be "degraded" should be given the same protection as one that is pristine and operating at full ecological function (Clark 1985) if "no net loss" of wetland habitat is to be maintained. However, it should also be recognized that the degraded wetland possesses a much higher mitigative potential.

Where wetland mitigation is considered, habitat diversification should be incorporated into the design plan and as a general guideline for wetland creation projects. Although wetland mitigation and management means several possibilities depending on the goals of the wetland manager (Mitsch & Gosselink 1986), the overall goal of habitat diversification is to incorporate both in-kind and out-of-kind replacement by including the same type of wetland habitat (unless considered degraded) with other habitat types. This would provide an effort to increase both plant and animal species diversity along with abiotic ecosystem processes attributed to habitat diversity. This approach, in turn, attempts to create unmitigatable wetlands of much higher functional quality which, in their natural state, should not be impacted in the first place.

Due to the current uncertainties of wetland mitigation, mitigation projects should be limited to small scale attempts (<10 acres) but with a minimum creation requirement of 2 acres. We believe that to successfully incorporate diversification into mitigation and significantly enhance the functional values of created wetland habitat, not less than 2 acres should be required for any impact to a wetland exceeding 0.5 acres. If wetland creation of 2 to 5 acres is proposed in an upland area not contiguous to another wetland system or under the conditions previously described, then legislation in the proposed rules of Act 203 should provide for the avoidance of creating non-regulated wetland habitat.

Although the MDNR requires construction of wetland creation projects prior to development of a parcel to assure compliance, it is our opinion that wetland habitats should, where possible, be created after major earth moving activities have been completed around the proposed wetland creation site to avoid post-creation impacts (e.g., erosion and sedimentation). Another precaution to help avoid potential post-creation impacts would be to propose wetland creation projects as far away from the development as possible and maintain as much of an upland buffer zone as possible. A buffer zone would not only provide extra protection to the created wetland but provide cover and resting habitat for wildlife.

It seems reasonable that small scale success from wetland mitigation efforts must first be achieved at a small cost so that we will have increased confidence, success, and cost-effectiveness in future large scale projects. The establishment of a larger data base from follow-up monitoring on at least a qualitative basis should be required of all mitigation considerations to document that the proposed goals are achieved. The follow-up monitoring program should incorporate a reference wetland (on-site if possible) for the purpose of examining the effectiveness of mitigation efforts and to quantify wetland functions and values, thereby providing an objective means to define mitigation requirements (Hughes et al., in publication). The reference wetland, however, could not be a site similar to the mitigated wetland since the goal to off-set the impact(s) to this area would not be restricted to in-kind replacement.

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LITERATURE CITED

Banner, A. 1979. Mitigation Under the Corps Regulatory Program. In G. A. Swanson (technical coordinator), The Mitigation Symposium: A national workshop on mitigating losses of fish and wildlife habitats. General Technical Report RM-65, Rocky Mountain Forest and Range Experiment Station, Forest Service, U.S.D.A., Fort Collins, Colorado, pp. 369-399.

Beecher, W. J. 1942. Nesting birds and the vegetation substrates. Chicago Ornithol. Soc., Chicago, Illinois, 69 pp.

Clark, J. R. 1986. Assessment for Wetlands Restoration. In J. A. Kusler and P. Riexinger (Eds.), Proceedings of the National Wetland Assessment Symposium, June 17-20, 1985, Portland, Maine, pp. 250-253.

Garbisch, W. E., Jr. 1977. Recent and planned marsh establishment work throughout the contiguous United States: A survey and basic guidelines. Contract Report D-77-3, Waterways Experiment Station, U.S. Army Corps of Engineers, Vicksburg, Mississippi, 277 pp.

Golet, F. C., and J. S. Larson. 1974. Classification of freshwater wetlands in the glaciated northeast. Bureau of Sport Fisheries and Wildlife, U.S. Dept. of Interior, Resource Publication No. 116, 56 pp.

Hughes, R. M., D. P. Larsen, and J. M. Omernik. In press. Aquatic ecoregions and reference wetlands: A conceptual approach for study and management of the nation's wetlands. In R. R. Sharitz and J. W. Gibbons (Eds.), Proceedings of the Symposium on Freshwater Wetlands and Wildlife, March 24-27, 1986, Charleston, South Carolina.

Jacobs, J. 1975. Diversity, stability, and maturity in ecosystems influenced by human activities. In W. H. van Dobben and R. H. Lowe-McConnell (Eds.), Concepts in ecology. Dr. W.J.B.V. Publishers, The Hague, pp. 187-207.

Jaworski, E., and C. N. Raphael. 1979. Historical changes in natural diversity of fresh water wetlands, glaciated region of northern United States. In P. E. Greeson, J. R. Clark, and J. E. Clark (Eds.), Wetland functions and values: The state of our understanding. American Water Resources Association, Minneapolis, Minnesota, pp. 545-557.

Larson, J. S., and C. Neill. 1986. Mitigating freshwater wetland alterations in the glaciated northeastern United States: An assessment of the science base. Proceedings of a workshop held at the University of Massachusetts, September 29-30, 1986, Amherst, Massachusetts. Publication No. 87-1, 143 pp.

MacArthur, R. H. 1964. Environmental factors affecting bird special diversity. American Naturalist 98:387-397.

MacArthur, R. H., and J. W. MacArthur. 1961. On bird species diversity. Ecology 42:594-598.

MacArthur, R. H., J. W. MacArthur, and J. Preer. 1962. On bird species diversity II. Prediction of bird census from habitat measurement. American Naturalist 96:167-174.

Mitsch, W. J., and J. G. Gosselink. 1986. Wetlands. Van Nostrand Reinhold Company, Inc., New York, New York, 539 pp.

Niering, W. A., and M. L. Kraus. 1986. Science base for freshwater wetland mitigation in the northeastern United States: Vegetation. In mitigating freshwater wetland alterations in the glaciated northeastern United States: An assessment of the science base. Proceedings of a workshop held at the University of Massachusetts, September 29-30, 1986, Amherst, Massachusetts. Publication No. 87-1, 143 pp.

Patterson, J. H. 1974. The role of wetland heterogeneity in the reproduction of duck populations in eastern Ontario. Can. Wildl. Serv. Rep. Ser. No. 29, 31-32.

Steel, P. E., P. D. Dalke, and E. G. Bizeau. 1956. Duck production at Gray's Lake, Idaho, 1949-1951. J. Wildl. Manage. 20:279-285.

Weller, M. W. 1978. Management of freshwater marshes of wildlife. In R. E. Good, D. F. Whigham, and R. L. Simpson (Eds.), Freshwater wetlands: Ecological processes and management potential. Academic Press, New York, pp. 267-284.

Weller, M. W., and C. E. Spatcher. 1965. Role of habitat in the distribution and abundance of marsh birds. Iowa Agric. Home Econ. Exp. Stn. Spec. Rep. No. 43, 31 pp.

THE APPLICATION OF SCIENCE AND ENGINEERING TO RESTORE A SALT MARSH, 1987

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ABSTRACT

The concept and practice of enhancing wetlands has been well established in the profession of wildlife management. However, to date, the practice of such enhancement was largely carried out in relatively pristine, freshwater environments, whereas in tidal areas or on sites already heavily distributed by man, biological enhancement is in its infancy.

The primary subject of this study is a 63-acre parcel adjacent to the New Jersey Turnpike's Eastern spur just north of New Jersey Route 3 (3 miles from Manhattan). The mitigation site is tidal and located in an urban region. The parcel was formerly a monoculture of common reed-grass (Phragmites communis), 3.6 to 4.5 meters tall. Because of its physical dominance, it has long been suspected that the common reed crowds out desirable wildlife, their food plants and reduces the value of the habitat. The 63-acre parcel has been enhanced using horticultural and earthwork techniques. Horticultural efforts involved elimination of common reed and seeding of saltmarsh cordgrass (Spartina alterniflora) to create a better environment for wildlife. Earthwork involved moving 60,000 yards of material on-site. The renovated marsh now features three diverse habitats: 10-15 percent open channels/mud flats, 10 percent dry berms, and 75 percent cordgrass meadows. This study compares wildlife utilization of the mitigated marsh to that of an adjacent unenhanced, 131-acre common reed marsh.

Highly mobile wildlife species have responded dramatically. A total of 32 different bird species were observed utilizing the still to be completed 63-acre mitigation site; 16 different species were seen on the 131-acre control site. Confirmed bird observation on the mitigation site numbered 1,592 versus 204 in the unmitigated wetland. This represents a doubling of bird species and a seven-fold increase in bird numbers. On an acreage basis, the increases are again doubled. Benthic invertebrates demonstrated a tripling of numbers and doubling of species compared to adjacent control unmitigated wetlands (Kraus 1986).

Mammaliam population numbers have doubled (on an acreage basis) on the mitigation site.

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*Subconsultant to TAMS Consultants, Inc. and Hartz Mountain Industries, Inc.

No noticeable changes in herptilian and fish populations were noted. Some improved water quality occurred but data was not sufficient to statistically verify the finding.

INTRODUCTION

The concept and practice of enhancing or creating wetlands has been well established in the profession of wildlife management. For example, the U.S. Fish and Wildlife Service spends millions of dollars annually to create impoundments, dredge waterbodies, reestablish desirable plant species and eliminate pests for the purpose of increasing the environmental value of wetland properties under their management. However, to date, the practice of such enhancement was largely carried out in relatively pristine, freshwater environments, whereas in tidal areas or on sites already heavily disturbed by man, biological mitigation is in its infancy.

The subject of this study is a 63-acre parcel adjacent to the New Jersey Turnpike's Eastern spur just north of New Jersey Route 3 (only 4.8 kilometers from Manhattan). The mitigation site is both tidal and in an urban region. The parcel, located in Mill Creek Basin in the Hackensack Meadowlands, was formerly a monoculture of common reedgrass (*Phragmites communis*) 12 to 15 feet tall. The population density (average 200 plus stems per square meter) was so great that visibility through the stands was less than one meter. Because of its physical dominance, it has long been suspected that the common reed crowds out desirable wildlife, their food plants and reduces the value of the habitat. According to the U.S. Fish and Wildlife Service, encroachment by common reed can cause a decline in wildlife use of up to 80 percent (U.S. Department of Interior 1983). The species primarily affected by this reduced habitat include ducks, other waterfowl, several marsh mammals, and wading shorebirds (i.e., herons, egrets, and rails).

Common reed was most probably also a factor in causing a pre-mitigation average site elevation of +1.06 meters National Geodetic Vertical Datum (NGVD). This elevation is only 0.3 miles below the mean high tide of +1.09 meters NGVD. Thus, only 2.5 or 5.0 centimeters of tidal water flows on-site twice daily. This slight tidal inundation in conjunction with poor water quality (i.e., dissolved oxygen frequently below 1.0 part per million) and the slowly decomposing reed stems caused the site to have reduced marine values.

The 63-acre parcel was enhanced using horticultural and earthwork techniques intended to enhance both its estuarine and marine environmental values. The major effort involved the effective elimination of the well-established common reed, and ensuring plant or seedling survival of saltmarsh cordgrass (*Spartina alterniflora*). Cordgrass is a shorter, less densely packed grass which constitutes a more natural habitat, which encourages a greater diversity of wildlife. Common reed was eliminated by the use of a glyphosate, systemic herbicide. Cordgrass was seeded on the site where elevations were reduced by excavation to +.6 to +.9 meters NGVD (or .15 to .45 meters below mean high water). Cordgrass needs to be inundated daily by tidal waters. Consequently, canals were cut leading from the adjacent Mill Creek deep into the 63-acre parcel. Excavation of slopes with drainage sufficient to provide the cordgrass with the right amount of water (between elevation +.6 and +.9 NGVD) required extreme care and precision. The excavated material was then piled into raised berms that will eventually support shrubs and trees for nesting and roosting areas (TAMS 1985). The renovated marsh now features three diverse habitats: 15 percent open channels/mud flats, 10 percent raised berms, and 75 percent cordgrass meadows.

The purpose of this study was to compare the restored marsh to an adjacent unenhanced 131-acre common reed marsh. Bird and mammal observations were conducted, water quality measurements were taken ("in-situ" and samples collected for laboratory analysis) and the fish population was sampled in each marsh area over a nine month period.

METHODS

The restored site, located in Secaucus, New Jersey, is bounded by the New Jersey Turnpike (East Spur) on the east, Mill Creek on the west, lower Cromakill Creek and the Hackensack River on the north, and a line perpendicular to the Turnpike on the south, about 427 meters south of the Turnpike bridge over Cromakill Creek. The winding Mill Creek shoreline of the mitigation site is about 610 meters long.

The control marsh, hereafter referred to as the IR-2 site, is adjacent to the restored marsh, occurring just south of it, and it is also bounded by Mill Creek on the west. The site is 95 percent common reed monoculture with several steep-sided mosquito control channels making up the balance of the area.

A comparative study of the two sites was conducted regularly between October 1986 and August 1987. Bird and mammal counts were performed monthly, while seining and water quality measurements were conducted bi-monthly.

Bird counts were conducted at each site by walking or boating through the entire site for a one-hour period. Only birds seen actually utilizing the marsh (i.e., feeding, resting, stalking overhead) were counted. Birds were observed with binoculars and identified using Birds of North America (Robbins et al. 1966). Physical evidence, such as tracks or scat, were also noted.

Mammal traps, with bait, were interspersed randomly throughout each site for 24-hour periods to sample the mammal populations. In addition, other physical evidence was observed, such as animal tracks, burrows, or scat, to determine the presence of mammals.

Seining was conducted bi-monthly at each site. A three meter seine was pulled once for fifteen meters. Netted fish were then identified and counted.

Water quality measurements were also taken bi-monthly at each site. Dissolved oxygen was measured with a YSI meter. Water was collected in sterile jars and then transported, on ice and within 12 hours, to a certified water quality laboratory where biological oxygen demand, total coliform bacteria, and fecal coliform bacteria levels were measured.

RESULTS

In terms of both the number of species and individuals, far more birds were recorded on the restored site than the adjacent reed marsh (see Table 1 for species list). A total of 32 different bird species were observed utilizing the still to be completed 64-acre mitigation site. Sixteen different species were seen on the 131-acre IR-2 site (Table 2). Confirmed observations on the restored site numbered 1,592 and 204 on the IR-2 site. This represents a doubling of species and a seven-fold increase in bird numbers, at the restored site, which is less than half the size of the IR-2 site. Three active waterfowl nests and 10 confirmed fledglings were observed on the mitigation site. No breeding birds were observed on the IR-2 site.

No mammals were caught in any of the traps at either the restored marsh or the IR-2 marsh. Tracks of the domestic cat (*Felis domesticus*), Norway rat (*Rattus norvegicus*) and muskrat (*Ondatra zibethicus*) were observed on the restored site. Muskrat and Norway rat tracks were also seen on the IR-2 site, as well as those of the eastern cottontail (*Sylvilagus floridanus*). Muskrat burrows were counted on the mitigation site and the IR-2 site. A total of 39 burrows were counted on the various islands (only 50 percent complete) in the mitigation area. Forty burrows per 130 acres were counted on the banks of the channels at the IR-2 site. No hut dwelling muskrats were observed on either site.

An average of 29 (standard deviation +-30.7) mummichogs (*Fundulus heteroclitis*) were netted per seine at the restored site. A total of 10 silverside (*Mendia beryllina*) and one sunfish (*Lepomis macrochirus*) were also taken at this site. An average of 148 mummichogs were caught per seine at the IR-2 site but the standard deviation, +-201.3 was very high. A total of 4 silverside were caught at IR-2 over the six month period.

The water quality measurements were similar at both sites (Table 2). The dissolved oxygen (DO) level averaged 5.2 ppm at the restored site and 4.9 ppm at the IR-2 site. Biological Oxygen Demand (BOD) levels were also similar at both sites, with an average of 6.2 ppm at the mitigation site, and 6.0 at the IR-2 site. Total coliform bacteria levels at both sites always exceeded 2,400 most probable number (mpn), while fecal coliform levels were almost identical at the mitigation and IR-2 sites with 1766 mpn and 1726 mpn readings, respectively.

Table 1. Species list of birds observed at the restored and IR-2 sites (October, 1986 - August, 1987).

Latin Name	Common Name	Mitigation Site	IR-2 Site
Agelaius phoeniceus	Red-winged blackbird	90	1
Ammospiza maritima	Seaside sparrow	12	0
Anas crecca	Green-winged teal	6	0
Anas discors	Blue-winged teal	0	3
Anas platyrhynchos	Mallard duck	68	19
Anas rubripes	Black duck	6	0
Ardea herodias	Great blue heron	2	0
Botaurus lentiginosus	American bittern	2	1
Butorides straitus	Green heron	1	2
Calidris alba	Sanderling	2	0
Calidris minutilla	Least sandpiper	230	0
Calidris pusilla	Semipalmated sandpiper	230	0
Charadrius semipalmatus	Semipalmated plover	230	0
Charadrius vociferus	Killdeer	19	6
Circus cyaneus	Marsh hawk	4	1
Cistothorus palustris	Marsh wren	22	34
Corvus brachyrhynochos	American crow	5	1
Egretta thula	Snowy egret	90	1
Falco columbarius	Pigeon hawk	1	0
Gallinulla chlorpus	Gallinule	1	0
Hirundo rustica	Barn swallow	27	18
Iridoprocne bicolor	Tree swallow	3	1
Larus argentatus	Herring gull	496	27
Larus atricilla	Laughing gull	4	0
Larus delawarensis	Ring-billed gull	3	0
Larus marinus	Greater black-backed gull	1	0
Melanitta deglandi	White-winged scoter	0	2
Melospizza georgiana	Swamp sparrow	26	28
Nycticorax nycticorax	Black-crowned night heron	2	0
Quiscalus quiscula	Grackle	0	6
Sterna albifrons	Least tern	23	0
Sturnus vulgaris	Starling	1	0
Tringa flavipes	Lesser yellowlegs	8	0
Tringa melanoleuca	Greater yellowlegs	30	0
Zenaida macroura	Mourning dove	3	0
	TOTAL	1592	204

Table 2. Water quality measurements at the mitigation and IR-2 sites (October, 1986 - August, 1987*).

Parameter	Units	Mitigation Site	IR-2 Site
Dissolves Oxygen	(ppm)**	5.2	4.9
Biological Oxygen Demand	(ppm)	6.2	6.0
Total Coli. Bacteria	(MPN)***	always >2400	always >2400
Fecal Coli. Bacteria	(MPN)	1766	1726

* Average of bi-monthly samples
** Parts per million
***Most probable number per million

DISCUSSION

Healthy Spartina marshes provide an ideal habitat for a variety of bird species. They generally are teeming with macroinvertebrates such as worms and crustaceans which form an integral part of many bird's diets. They serve as a spawning ground for fish--another component of a predatory bird's diet. They provide the proper shading and protection from potential predators and yet have enough open space to allow birds and mammals freedom of movement throughout the marsh.

In a common reed marsh, the plant's stems grow so tall and densely (on the IR-2 site, stem counts average 200 per square meter with heights of 3.6 to 4.5 meters), that movement within the marsh is extremely limited. Avian species are affected both physically and behaviorally. It is physically impossible for a comparatively large bird such as a heron or egret to stalk or roost in a robust reed marsh. The stands of reed are simply too dense. Behaviorally, they cannot tolerate being "closed in." They are apparently unable to accept the limited sight distances which would shorten their escape response time when a dangerous intruder is detected.

The above data indicate that shorebirds in the Hackensack Meadowlands area are attracted to the newly restored marsh. A 100 percent increase in species and a 700 percent increase in numbers was found at the mitigation site. On an acreage basis, there was a 1,400 percent increase in the number of birds in the mitigation area. Many wading birds (two great blue herons and 64 snowy egrets) and a variety of shorebirds (sandpipers, plovers, least terns, and several species of gulls) were observed utilizing the restored marsh. Successful waterfowl breeding was also observed on the restored site due to the presence of the dry berm nesting sites above tidal flows.

At the IR-2 site, only birds small enough to go through the dense reed stems (i.e., sparrows, red-wing blackbirds, wrens), were observed utilizing the Phragmites marsh as well as some ducks which were swimming in the channels adjacent to the reed stands. In the unenhanced common reed marsh perhaps only 5 to 10 percent of the habitat was usable by estuarine wildlife, and then only at low tide. When the mitigation is conducted, thoroughly interspersed estuarine habitat is available over the entire site for wading, feeding, resting and nesting throughout the tidal cycle (Hanley 1987).

The fish population of both areas was composed almost exclusively of the mummichog (Fundulus heteroclitis). While greater numbers of mummichogs were caught in the unenhanced site, the data indicated that there is, in fact, a large number of mummichogs in the restored site--enough to provide an adequate food source for predatory wading birds in the region. Lack of improved species diversity on the site is due to the generally poor water quality condition of the adjacent river as discussed below.

The data seen indicate that there is no significant difference in the mammalian community of each area. However, an equal number of muskrat burrows were observed in each area. Since the restored site encompasses 63 acres, while the IR-2 site is more than double in size (131 acres) there appeared to be doubling of the muskrat population on the restored site. The lack of increased mammalian species diversity is probably due to the lesser mobility of these species and lack of suitable, proximate, natural mammalian populations.

The benthic invertebrate population was compared previously at both sites in a study conducted by the Hackensack Meadowland Development Commission. It was found that benthic invertebrates demonstrated a tripling of numbers and doubling of species by comparison to adjacent unmitigated wetlands (Kraus 1986).

The last comparison conducted, water quality, also reflected no significant difference between the waters of the mitigated marsh versus the waters of the control marsh. Dissolved oxygen and BOD levels in both areas were nearly identical. Both fecal and total coliform levels in each area were extremely high, as is typical of marshes in the Hackensack estuary.

These data lend to the preliminary conclusion that a simple, physical change in the marsh environment without accompanying detectable chemical changes can make a dramatic difference in wildlife utilization. This technique has been used by wildlife managers in relatively pristine wildlife refuges for many years. As a result of this project, it appears that the same techniques can be effectively applied to environmentally distressed urban sites.

It should be noted that the restored marsh is quite young. Work commenced in 1985, and still continues. Observations there will continue for some years and most likely will reveal an even greater diversity of wildlife. This further positive result is anticipated as disturbance due to construction and horticultural work subsides and the cordgrass matures, allowing the more sensitive marsh species (e.g., over-wintering teals, *Anas* spp.) to utilize the marsh.

LITERATURE CITED

Hanley, Robert. From man, a home for wildlife. New York Times, 3 January 1987, p. 25.

Kraus et al. October 1986. Benthic fauna in a mitigated versus unmitigated tidal wetland in New Jersey. Hackensack Meadowlands Development Commission, Lyndhurst, New Jersey.

Robbins, Chandler S., Berrel Bruun, and Herbert S. Zim. 1966. Birds of North America. Golden Press, New York.

TAMS. January 1985. Brackish wetland mitigation plan. U.S. Army Corps Permit No. 12746. New York, New York.

U.S. Department of the Interior, Fish and Wildlife Service. 1983. Environmental assessment marsh vegetation rehabilitation--chemical control of Phragmites Prime Hook National Wildlife Refuge. Smyrna, Delaware.

A SALTWATER WETLAND IN NORTHEASTERN KENTUCKY

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ABSTRACT

In the summer of 1987, the Kentucky Transportation Cabinet created a 1.5 hectare wetland as mitigation for the loss of a similar habitat that was to be destroyed as a result of highway construction. However, soon after construction, a water quality problem occurred that has not previously been addressed by wetland biologists. Although chloride levels had been 25-35 mg/l in the original wetland and adjacent Salt Lick Creek, they were measured as high as 1600 mg/l in the created wetland, which lies midway between the original wetland and the creek. EPA-established criteria for most aquatic life is 1200 mg/l. We had apparently intercepted an underground, salt-laden stream by the construction of a new wetland.

Planning and timing of the construction was designed to create a facsimile of the original wetland. Preconstruction sampling of the original wetland revealed it to be a nursery for fishes and an important habitat for several species of frogs, salamanders and turtles. The new wetland was dug in a fallow cornfield and was similar to the original in area and depth. Then a hydraulic connection was created between the two. The original wetland was seined and resident fishes and herpetofauna were transferred to the created slough. After drawndown in the original, the seed-bearing muck was moved from the original to the new wetland. Finally, the original wetland was filled, and the edges of the created were planted with trees and shrubs.

After considering several expensive solutions to our chloride problem, we decided to wait and see if flooding would alleviate the situation. January rains raised the stream level into the wetland bringing an abundance of young fishes and also diluting the salts. Through the spring and summer of 1988, chloride levels remained at or below 40 mg/l. Several species of amphibians, reptiles and fishes now use the new wetland, and the organic material distributed in the area has proved to be a rich source of diverse wetland vegetation.

INTRODUCTION

In far western Kentucky where the physiographic province of the Coastal Plain nears its northern limits, wetlands are not rare habitats. Cypress ponds and bottomland hardwood forests still persist despite continued encroachment by agriculture, silviculture and development. However, on the unglaciated Appalachian Plateau of eastern Kentucky, wetlands have never been common and often furnish important, but isolated, habitats for a variety of species.

The construction of the Alexandria to Ashland Highway impacted a 1 hectare, Palustrine, shrub swamp in Lewis County of northeastern Kentucky. As mitigation for a required 404 permit from the U.S. Army Corps of Engineers, a replacement wetland was created adjacent to the slough that was to be eliminated by highway fill slopes. Although biologically important communities of long replacement time such as old growth cypress and bottomland hardwood wetlands cannot be created in a lifetime and should be avoided, in this situation an on-site creation was a reasonable mitigative measure.

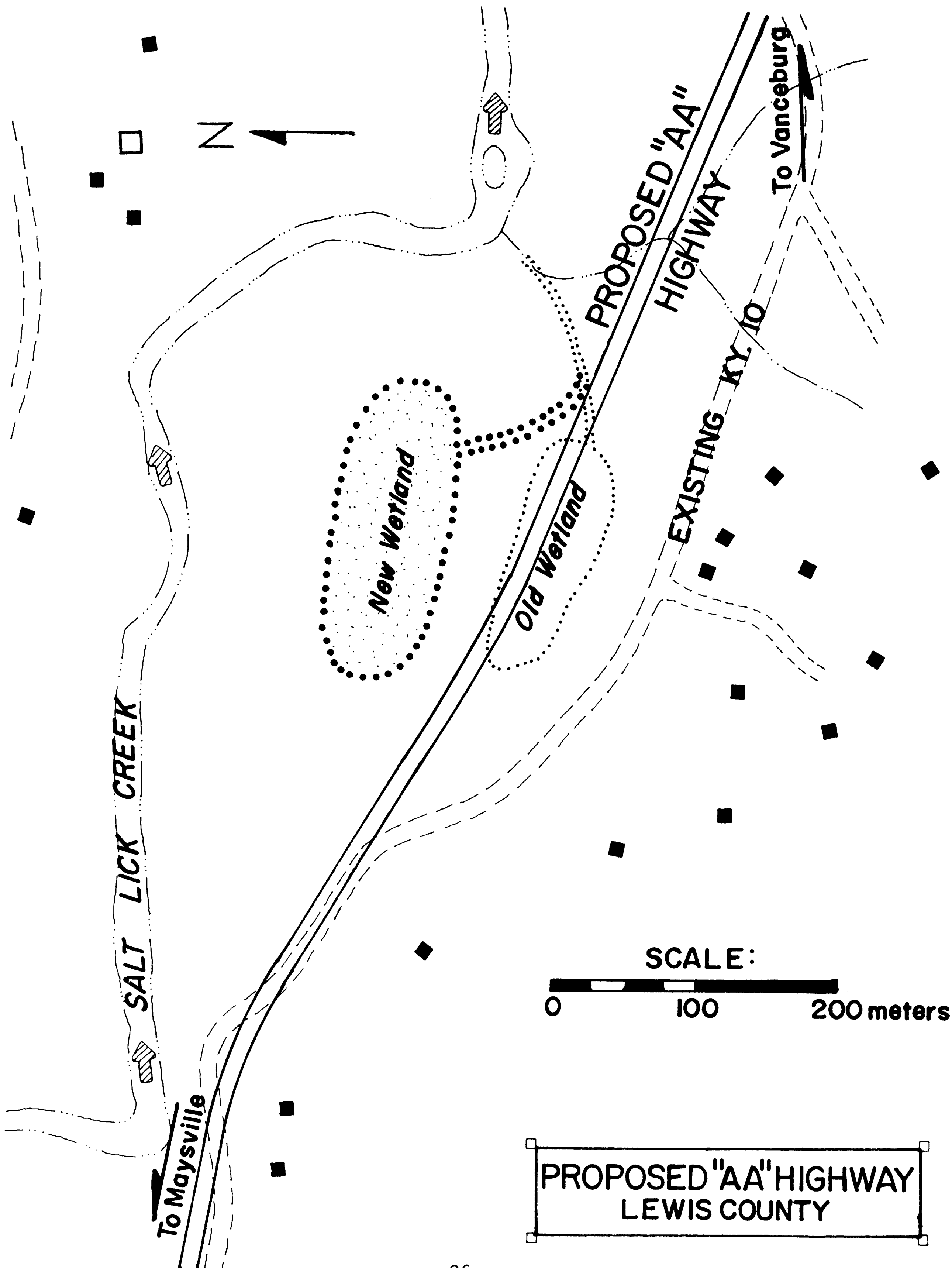
STUDY SITE

The project site is near Vanceburg, Kentucky approximately 1.5 km south of the Ohio River and 65 km west of Ashland. The shrub swamp in the project path was an abandoned channel of Salt Lick Creek with an open water area approximately one meter deep (see figure).

Although the area did not support any plants considered rare by state or federal agencies, it contained a diverse assemblage of native flora. Buttonbush, *Cephalanthus occidentalis*, was a dominant, but other woody species like alder, *Alnus serrulata*; black willow, *Salix nigra*; and red maple, *Acer rubrum*, ringed the shallow, open slough. Dominant herbaceous species included arrowhead, *Sagittaria latifolia*; water plantain, *Alisma subcordatum*; monkey flower, *Mimulus ringens*; sensitive fern, *Onoclea sensibilis*; water hemlock, *Cicuta maculata*; and false nettle, *Boehmeria cylindrica*.

Because the slough was an abandoned channel hydraulically connected with Salt Lick Creek, it also served as a nursery for several species of stream fishes. Fry of sunfishes, *Lepomis* spp., and spotted bass, *Micropterus punctulatus*, were common in the slough as were several smaller cyprinids, catfish and suckers. Several species of amphibians such as spotted salamanders, *Ambystoma maculatum*; bullfrogs, *Rana catesbeiana*; green frogs, *R. clamitans*; red-spotted newts, *Notophthalmus viridescens*; and spring peepers, *Hyla crucifer*, all laid their eggs in the slough. A trilling chorus of American toads, *Bufo americanus*, was deafening in early spring and the northern leopard frog, *Rana pipiens*, reached its southern distributional limits in this area. Snapping turtles, *Chelydra serpentina*, and midland painted turtles, *Chrysemys picta marginata*, were also common in the wetland.

Bat species documented feeding over the stream and slough included red bats, *Lasiurus borealis*; eastern pipistrelles, *Pipistrellus subflavus*; little brown bats, *Myotis lucifugus*; and Keen's bat, *M. keenii*. Two small mammals, short-tailed shrews, *Blarina brevicauda*, and white-footed mice, *Peromyscus leucopus*, were found in the wetland. Wood ducks, *Aix sponsa*, were seen in the slough several times.


N
PROPOSED "AA" HIGHWAY
To Vanceburg
EXISTING KY. 10
New Wetland
Old Wetland
SALT LICK CREEK
To Maysville
SCALE:
0
100
200 meters
PROPOSED "AA" HIGHWAY
LEWIS COUNTY

The species inventory revealed the existing slough to be an uncommon habitat that harbored many native wetland species, and therefore, a valuable ecological community (Bryan 1987).

METHODS

An investigation of the wetland slough to be lost was initiated to document what were its resident species and to gain insight into the creation of a similar system. Both floral and faunal inventory of the existing buttonbush slough was done in January, April, June and July of 1986. Plants were recorded and those species not identified in the field were collected for laboratory determination. The wetland was sampled in both January and June with seines and dipnets. Twenty-five pitfall traps as described in Caldwell and Bryan (1982) were set around the edges of the slough for small mammals and crayfish. Discarded bottles were also searched for remains of small mammals (Pagels & French 1987). Monofilament mist-nets were set over the stream and slough for two nights in mid-June to collect bats. Water quality parameters were determined in both the existing slough and adjacent Salt Lick Creek with a Hach kit.

Since engineers and contractors were not familiar with wetland creation, several on-site meetings were held in the summer of 1987 to discuss both the importance of the wetland and critical timing of the steps of the construction process. Although most of these considerations were detailed on the plans, these on-site discussions were essential to emphasize their importance to project success.

RESULTS

The site of the new wetland was a fallow cornfield located between Salt Lick Creek and the slough that was to be filled. The first step was to dig the new wetland a little larger (about 1.5 hectare), and at the deepest point, 0.7 meters deeper than the existing. This would replace slightly more area than that lost and allow for the inevitable silting of the new wetland. Outflow elevations were critical to wetland function as the existing wetland was both supplied and flushed by creek flooding. In step two of construction, elevation of the original hydraulic connection with the stream was maintained. A connector ditch was then constructed to drain the old wetland into the new. Water movement between the wetlands ceased when levels in the original wetland fell to an approximate depth of 0.7 meters. Water in the created wetland was supplied mostly by groundwater and only slightly augmented by the hydraulic connection.

In step three of the process aquatic fauna, such as fishes, turtles, amphibian larvae and macroinvertebrates, were translocated from the old to the new wetland with seines and dipnets. The next phase was to remove the seed-bearing, organic muck from the old wetland and distribute it around the borders of the created wetland to a depth of approximately 0.3 meter. After removing as much of the muck and aquatic fauna as possible, the original buttonbush slough was covered under 10 meters of highway fill. Seeding and planting of the new wetland was to begin immediately.

However, in late October of 1987, a local resident informed us we had constructed the wetland on the site of an old "salt mine." Kentucky pioneers often traveled many miles to where salt-laden springs would emerge and valuable salt could be collected by boiling. However, no springs were visible in this area when construction began. After being alerted to this possibility, we conducted water quality sampling in the new wetland which showed chloride levels of 1250 mg/l and specific conductivity at 2000 micromhos/cm. We had apparently intercepted a salt-laden aquifer with new wetland excavation. For chloride, the EPA-established one-time maximum criterium for aquatic life is 1200 mg/l (EPA 1987). Perhaps the name of the creek should have alerted us to the possibility of a problem, but we had not noticed elevated salt levels in either the stream or the original slough. Preconstruction chloride levels in the slough and adjacent Salt Lick Creek were 35 and 10 mg/l, respectively. The remote chance of hitting an unsuspected subterranean stream would have made groundwater sampling impractical.

Solutions to our problem were either expensive or nonexistent. We considered methods of sealing subsurface flows and/or diluting the salts by pumping water from Salt Lick Creek. At least one engineer suggested we simply use crabs and cordgrass instead of crayfish and cattails in our eastern Kentucky salt marsh. However, in spite of chloride levels that reached 1600 mg/l in the next couple of months, the translocated fauna continued to survive in the new wetland.

Then in January 1988, winter rains raised the stream level into the wetland and accomplished salt dilution without the aid of engineers or biologists. Perhaps siltation also reduced the influx of salt-laden groundwater. After the flood, chloride levels returned to about 40 mg/l and have not risen above this level despite drawdown conditions caused by the summer drought of 1988 (Table 1).

Elevations of the hydraulic connections to the adjacent stream were essential to the functioning of the new wetland. Flooding also brought an abundance of young fishes into the wetland. In the spring of 1988 thousands of American toad larvae were using the created slough. The spreading of topsoil and muck from the original wetland was a success. Wetland species such as sensitive fern; buttonbush; fog-fruit, *Lippia lanceolata*; spike-rushes, *Eleocharis* spp.; smartweeds, *Polygonum* spp.; and sedges, *Carex* spp., are flourishing in and around the new wetland. Vegetation cover averaged 85% in May of 1988, where the muck was distributed and less than 10% (mostly garden weeds) in a small area that was not so treated. Seeding was not needed where the muck was spread. It will, however, require several years of succession to create sufficient habitat structure in the water to provide cover for aquatic invertebrates, fishes and herpetofauna. Presently, this is the limiting factor in the success of the wetland as a faunal habitat.

Table 1. Chloride levels (mg/l) and specific conductance (micromhos/cm) in Salt Lick Creek, original shrub swamp and created wetland before and after highway construction in Vanceburg, Lewis County, Kentucky (Cl/sp. cond.).

Date	Salt Lick Creek	Original Shrub Swamp	Created Wetland
JAN 8, 1986	5/400	-	pre-construction
APR 4, 1986	-	35/550	pre-construction
SEP 3, 1987	N.D./420	filled	1250/3200
OCT 5, 1987	-	-	1400/3400
NOV 18, 1987	10/380	-	1575/3800
JAN-FEB	floods	-	floods
MAR 31, 1988	-	-	35/1000
MAY 12, 1988	-	-	30/1250
JUN 27, 1988	-	-	40/1700
AUG 7, 1988	-	-	40/1200
OCT 26, 1988	-	-	35/1000

N.D. = no data

CONCLUSIONS

Despite careful preconstruction data collection, a problem arose in wetland construction that could not have been foreseen by wetland planners. It took winter flooding and presumably sealing by silt to reduce chloride levels in the new wetland.

This project demonstrated a need for close monitoring of project initiation, construction and completion. Information and instruction transmitted from biological wetland planners and engineers to construction supervisors are sometimes not fully understood.

Qualitative observations of flora and fauna and measurements of water quality of the created wetland will continue for several years. Early monitoring revealed the high chloride level and recently has indicated another potential problem that we are attempting to mitigate. Fill slopes of the new road were constructed of acidic shale that threatens to contribute excessive iron and sulfur into the wetland. These slopes were covered with fill material and a herbaceous cover was attempted despite spring and summer drought conditions. Ironically, we are designing other small wetlands underlain with limestone to mitigate acidic runoff on other projects, but had not anticipated the problem here.

These situations have illustrated the need for project monitoring not only to indicate unexpected problems, but also to provide information for the next project. However, on most highway projects funds are not available for such monitoring unless they are set aside during the project planning or design phases. Natural resource agencies should require monitoring as an essential part of the permit conditions so that early provisions can be made for funding beyond project completion.

ACKNOWLEDGEMENTS

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LITERATURE CITED

Bryan, H. D. 1987. Important habitats and quantitative environmental assessment. In W. R. Byrnes and H. A. Holt (Eds.), Proceedings of the Fourth Symposium on Environmental Concerns in Rights-of-Way Management (pp. 517-221). Indianapolis, Indiana, October 25-28.

Caldwell, R. S. and H. D. Bryan. 1982. Notes on distribution and habitats of Sorex and Microsorex (Insectivora: Soricidae) in Kentucky. Brimleyana No. 8:91-100.

Environmental Protection Agency. 1987. Water Quality Standards for the Surface Waters of the Commonwealth of Kentucky; Final Rule. Federal Register 52(54):9102-9105.

Pagels, J. F. and T. W. French. 1987. Discarded bottles as a source of small mammal distribution data. Am. Midl. Nat. 118(1):217-219.

CUMULATIVE IMPACT ASSESSMENT--
A WATERSHED APPROACH

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ABSTRACT

The Florida Department of Environmental Regulation (FDER) is required to review cumulative impacts under Section 403.919, Florida Statutes (1985). The Tampa Bay Regional Planning Council, with funds from the FDER Coastal Zone Management Program, sought to develop a program to assess the needs, opportunities and constraints for management of cumulating developmental impacts.

An inventory of existing conditions and a habitat trend analysis was accomplished for two minor tidal tributary watershed undergoing a variety of development pressures. Results of the program define the problems with cumulative impact analysis, current statutory requirements and the recommended management scheme.

INTRODUCTION

Tampa Bay is one of the largest estuaries in the world (400 square miles) with 1.7 million people now living in the three counties bordering its shores. This represents a 55 percent population increase since 1970 (TBRPC 1987). Rapid urban and industrial development have radically changed the character and ecology of Tampa Bay and adjacent estuarine systems. For example, recent studies have indicated that 44 percent of the original 25,000 acres of mangroves and marshes has been destroyed, and 81 percent of the original 76,500 acres of seagrasses has disappeared. 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, a complete collapse of such fisheries as those for scallops and oysters and major declines of the bait shrimp, red drum, and spotted sea trout fisheries.

Rivers and tidal creeks are vulnerable to numerous impacts which also become evident downstream in terms of decreased estuarine productivity. Examples of the impacts include (1) hydroperiod alterations through excessive drainage or impoundment, (2) loss of corridor by damming, (3) changes to stream loads by increasing runoff or discharging pollutants, and diverting or preventing flows, (4) increased relief and habitat losses through dredging and filling, and (5) contamination through disposal of toxic materials. As rivers and creeks deteriorate, their ability to buffer cultural shocks to the estuary is also lost.

The destruction or modification of coastal and estuarine wetland vegetation has been occurring at an alarming rate in the Tampa Bay area, and has been identified in a report to the Florida Legislature by the Tampa Bay Study Committee in 1983 as the most serious problem affecting the ecological stability of the bay. Historically, however, local governments have acted independently in regulating the development of their natural resources. Consequently, the effects of habitat destruction have generally been evaluated on a parcel by parcel basis with little concern for the cumulative effects on the entire Tampa Bay system.

The Tampa Bay Regional Planning Council, with funds from the Florida Department of Environmental Regulation (FDER), Coastal Zone Management Program, sought to develop a program to assess the needs, opportunities and constraints for management of cumulative developmental impacts to tidal creek watersheds. The information provided in this report has been condensed from Assessing Cumulative Impacts on Tidal Creek Watersheds prepared by the Tampa Bay Regional Planning Council (1987).

STUDY SITE

Delaney Creek in Hillsborough County represents a tributary through an industrialized urban and agricultural area with rapid urbanization taking place (Figure 1). Frog Creek in Manatee County represents a system through an agricultural-rural watershed with little alteration in the estuarine portion of the creek (Figure 1). Delaney Creek has been classified in stressed condition, while Frog Creek is considered a natural tidal tributary to Tampa Bay (TBRPC 1986).

The two tidal tributaries to Tampa Bay were surveyed with respect to hydrographic features, biology and chemistry, and physical and chemical alterations (provided in TBRPC 1986 & 1987). A land use inventory for 1950 and 1982 time periods with an associated land use trend analysis was accomplished within each watershed to evaluate development changes.

METHODS AND MATERIALS

The watershed inventories, trend analysis and statistical information was provided by the Marine Resource Geographic Information System (MRGIS) owned and operated by the Florida Department of Natural Resources (FDNR) Bureau of Marine Research. The data generation and queries were accomplished on Earth Resources Laboratory Application Software (ELAS) developed by NASA. The information was then downloaded for printing in Earth Resources Data Analyses System (ERDAS). The land use information for the MRGIS was provided by the U.S. Fish and Wildlife Service with refinement by FDNR.

The background information and recommended cumulative assessment

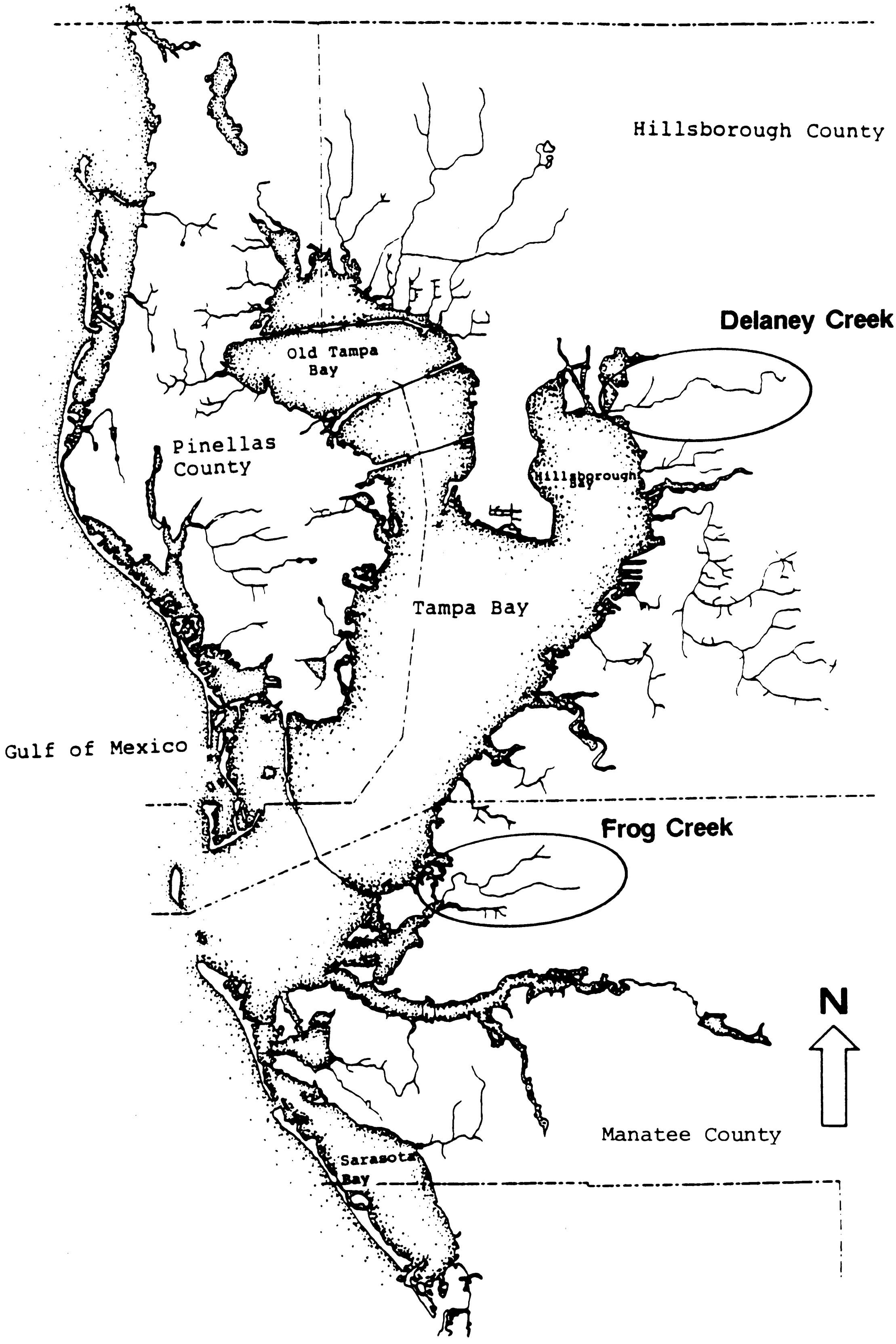


Figure 1. Location of Delaney Creek and Frog Creek in the Tampa Bay area.

procedure was accomplished by extensive literature surveys and data gathering relating to cumulative impact analysis throughout North America, irrespective of application to environmental systems.

RESULTS

Delaney Creek is located in Hillsborough County north of the Alafia River and south of the City of Tampa. The watershed contains industrial areas near the mouth with residential, commercial and agricultural activities in the remainder of the watershed. Due to construction of Interstate-75 through the middle of the drainage basin, rapid urbanization is currently occurring.

Delaney Creek is a first order stream as defined by the Florida Land Use and Cover Classification System (1977). The creek flows westward from the Brandon area and empties into the Hillsborough Bay subsection of Tampa Bay. The creek is designated as Class III water as defined by the Florida Administrative Code (FAC), Chapter 17-3. The creek, however, does not meet these standards (TBRPC 1986). The waters of Delaney Creek often exhibit low dissolved oxygen values, extremely high nutrient concentrations, and has exhibited phytoplankton blooms in the past. Delaney Creek has been classified as a stressed tidal tributary by TBRPC (1986).

Review of the 1950 inventory of land uses for the Delaney Creek watershed identified 71 percent (%) as upland range (including upland forest areas) with urban uses totaling only 4.0% (Figure 2). Agricultural activities occupied 12% with freshwater wetlands constituting 7.4% of the drainage basin. The freshwater wetland category includes all freshwater marsh grasses, forest and scrub wetland vegetation unless otherwise noted. Table 1 identifies the inventory and trend analyses statistic for the Delaney Creek watershed.

Figure 3 represents the 1982 land use inventory for Delaney Creek. Upland range diminished to 15% while urban land uses increased to occupy 61% of the watershed area. Agricultural areas increased slightly, to 17%, while freshwater wetlands were reduced to almost half, 4.0% of the watershed.

To further identify the land use changes, a trend analysis was accomplished between the two time periods. Figure 4 represents the 1982 inventory on top while the bottom depicts 1950 land uses occurring under the 1982 urban category. This allows the reader to visualize the areas displaced by 1982 urban uses.

The vast majority of 1982 urban uses were derived from upland range areas as expected and indicated on Figure 4. In addition, 328 acres (133 hectares) of freshwater wetlands were converted to urban uses.

The reverse can be illustrated by reviewing the change of 1950

Figure 2. Land use inventory for Delaney Creek - 1950.

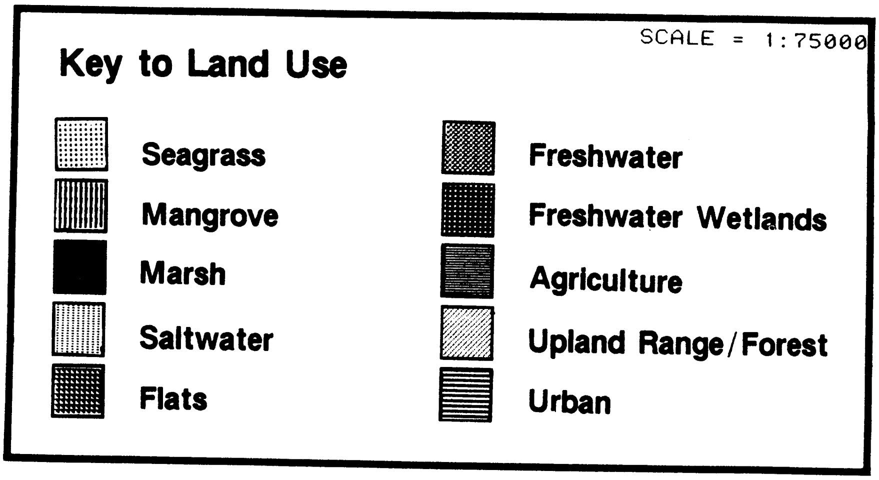


Figure 3. Land use inventory for Delaney Creek - 1982.

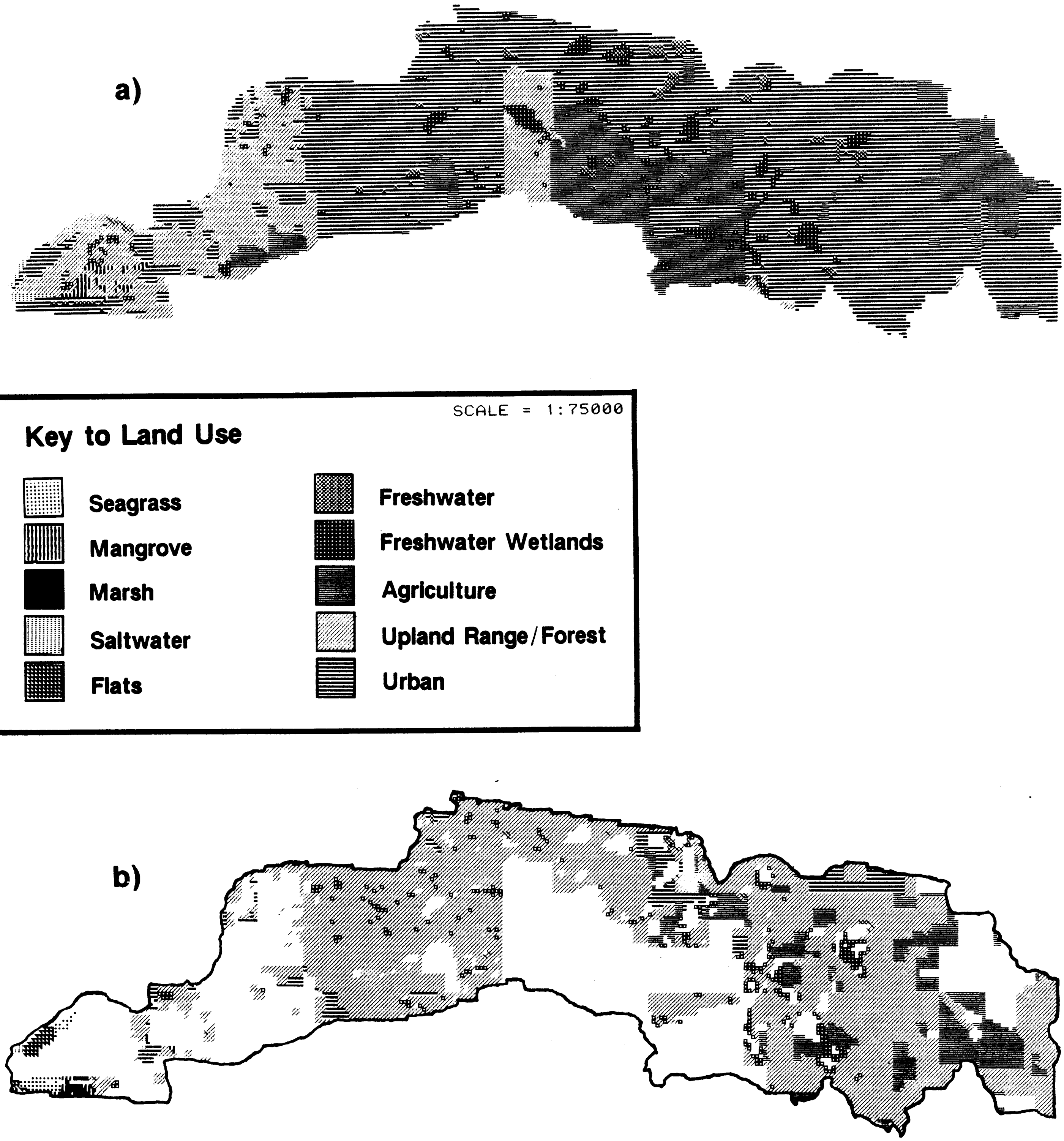


Figure 4. 1982 inventory (a) and 1950 land uses displaced by 1982 urban uses (b) for Delaney Creek.

upland range to other uses by 1982 (Figure 5). While urban uses acquired the vast majority of 1950 upland range, agricultural activity additionally consumed previous upland range area through the middle of the drainage basin.

Alterations to historic 1950 freshwater wetland coverage are represented in Figure 6. Total freshwater wetlands declined by 48% to 1982 levels with only 35% of the freshwater marsh component maintained in the watershed over the trend analysis, mostly due to urban and agricultural activities.

Frog Creek is located in Manatee County south of the County line and north of the Manatee River. The lower creek west of U.S. 41 remains relatively unaltered, maintaining a meandering course outfalling to Terra Ceia Bay, a subsection of Tampa Bay. East of U.S. 41, however, Frog Creek splits into what has become essentially two large agricultural drainage systems. Frog Creek has been classified as a natural tidal tributary to Tampa Bay (TBRPC 1986).

No significant baseline water quality assessment has been accomplished for Frog Creek. Considering its predominantly agricultural watershed, non-point sources potentially discharge high loads of nutrients, pesticides, herbicides, and sediment into the creek drainage system. Septic tank discharges may also contribute to bacterial and nutrient contamination.

The lower pristine estuarine segment has maintained extensive mangrove wetlands and is highly productive for oysters and blue crabs. Fishery collection near the saltwater/freshwater interface indicate utilization as a nursery area by important fish species such as tarpon (Megalops atlantica) and spot (Leiostomus xanthurus) (TBRPC 1986). In addition to fishery habitat, the mangrove wetlands of Frog Creek provide important bird habitat, including yellow crowned night heron (Nyctanassa violacea) and roseate spoonbill (Ajaia ajaja).

As would be expected, the 1950 inventory for Frog Creek identified 80% of the watershed in either agricultural or upland range land uses (Figure 7, Table 2). Freshwater wetland area represented 18 percent of the drainage basin which total 2,100 acres (931 hectares).

The 1982 inventory (Figure 8) portrays a doubling of area in agricultural activity. This change is also reflected by the trend analysis for 1950 upland range (Figure 9) where 60% of the land use was converted to agricultural practices.

Over the trend analysis period limited urban expansion within the Frog Creek drainage basin occurred, which includes the construction of Interstate 75 through the middle of the watershed and aerial expansion of the small town of Parrish in the eastern portion of the drainage basin (Figure 9).

Freshwater wetlands received extensive development pressure,

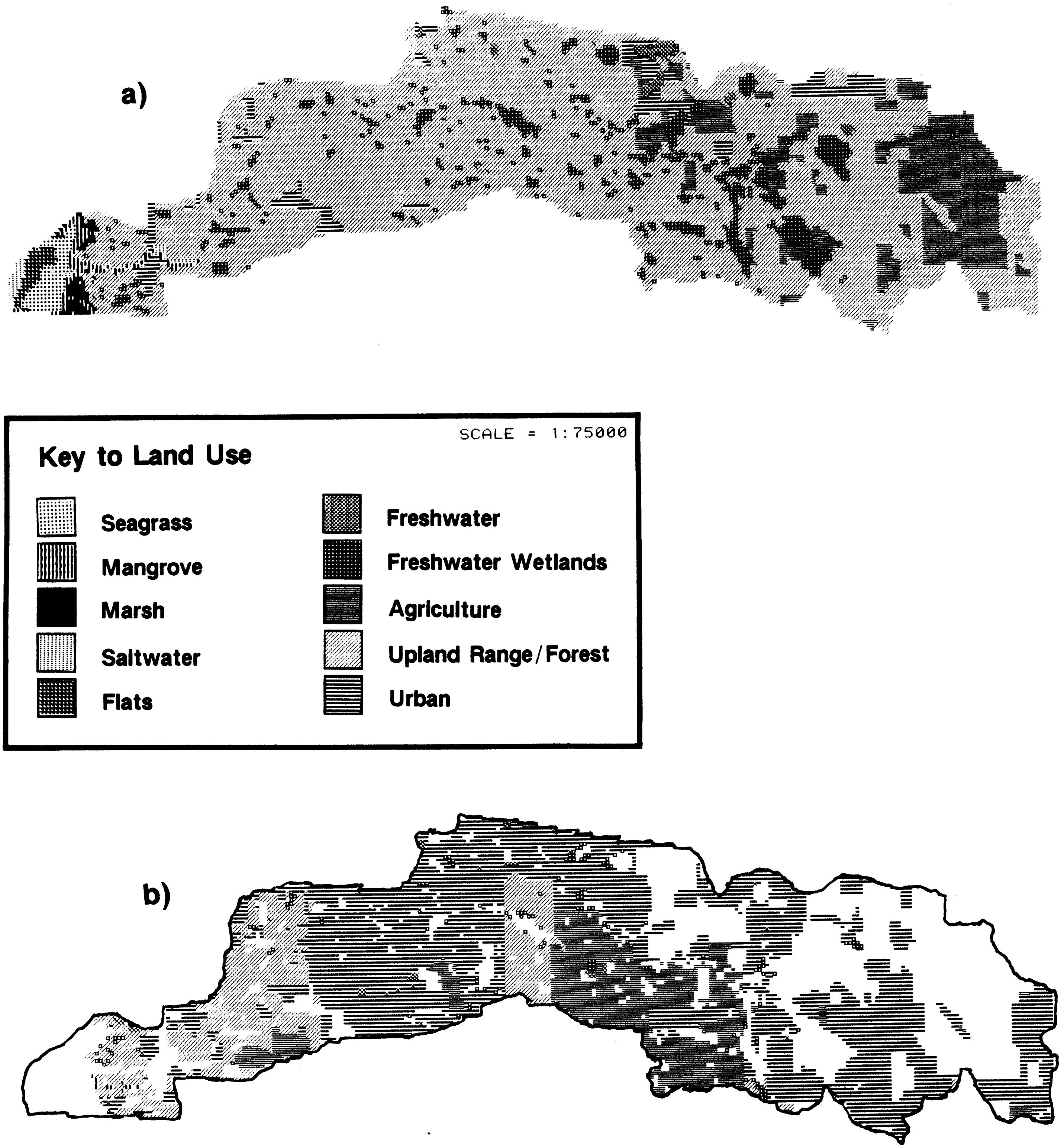


Figure 5. 1950 inventory (a) and 1982 land uses displacing 1950 upland range (b) for Delaney Creek.

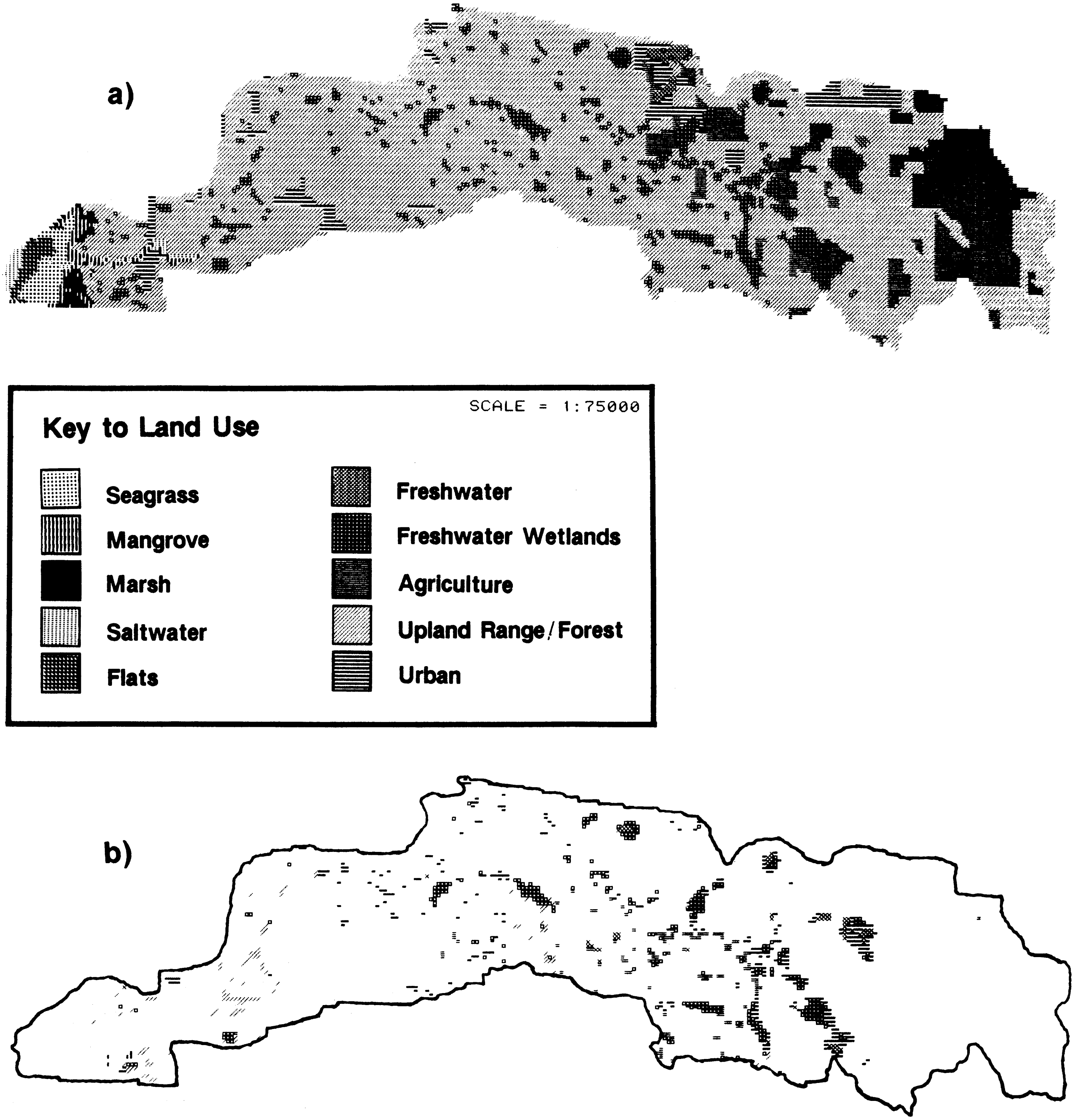


Figure 6. 1950 inventory (a) and 1982 land uses displacing 1950 freshwater wetland areas (b) for Delaney Creek.

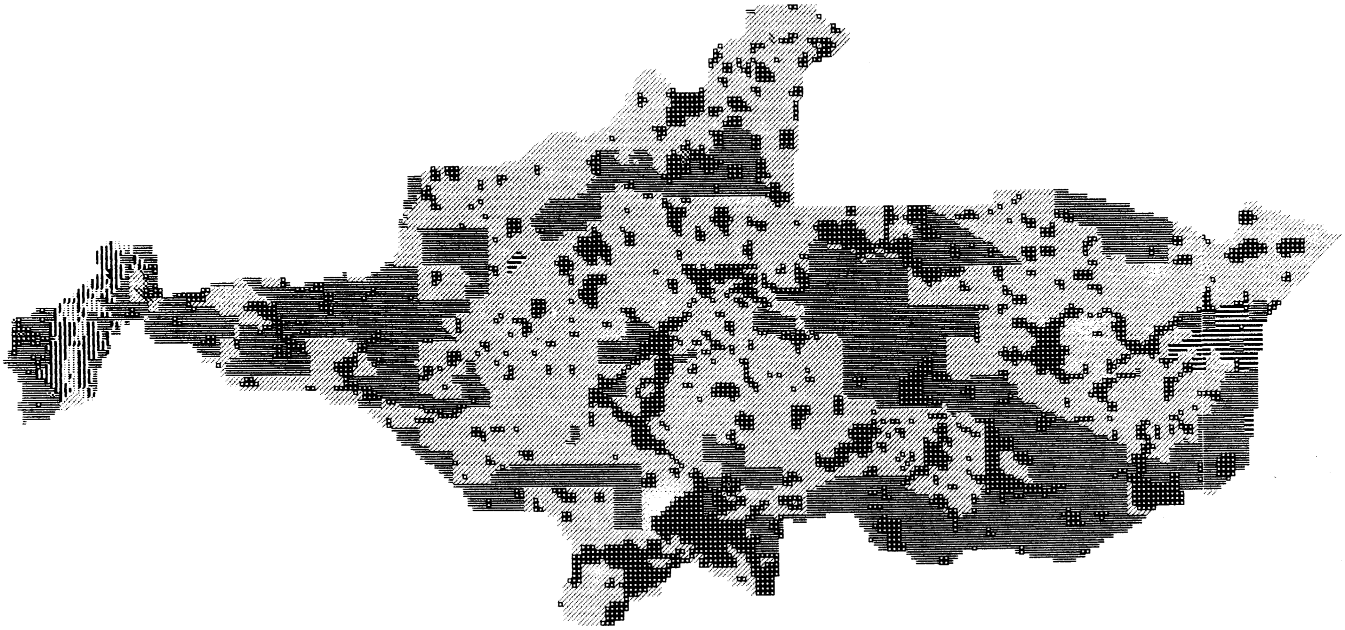
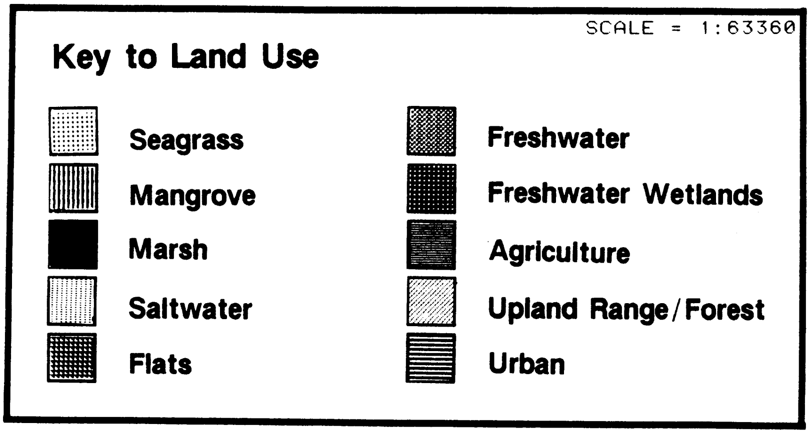


Figure 7. Land use inventory for Frog Creek - 1950.



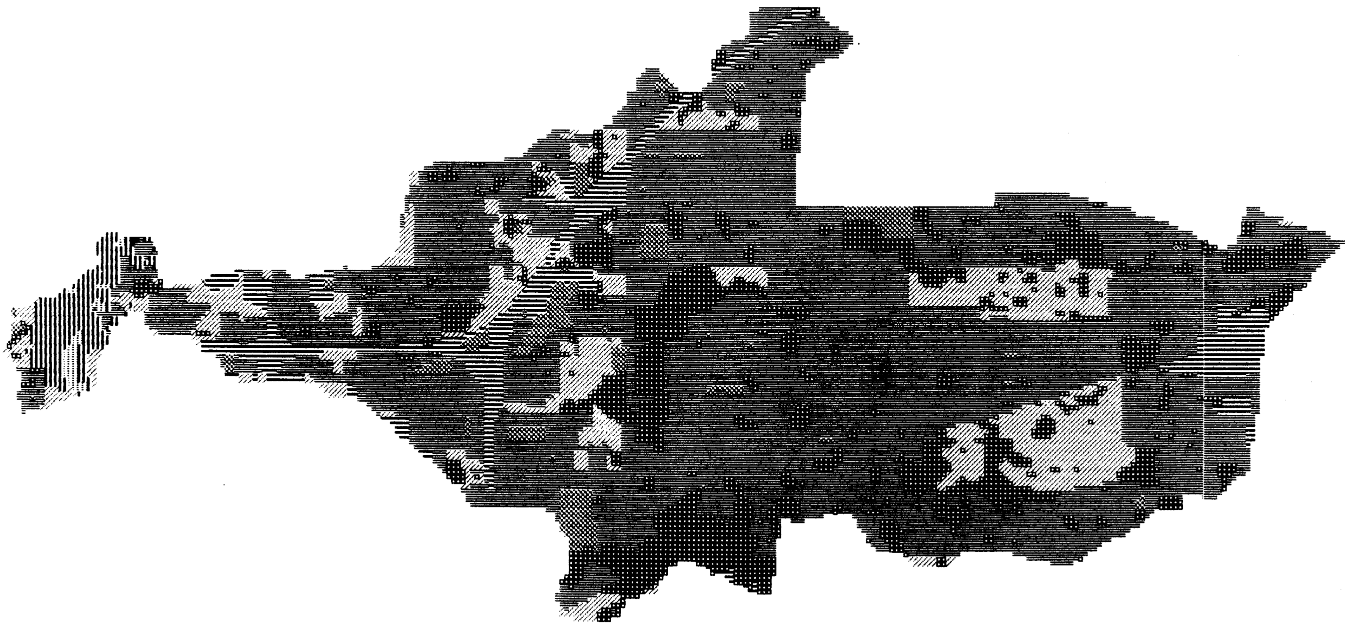


Figure 8. Land use inventory for Frog Creek - 1982.

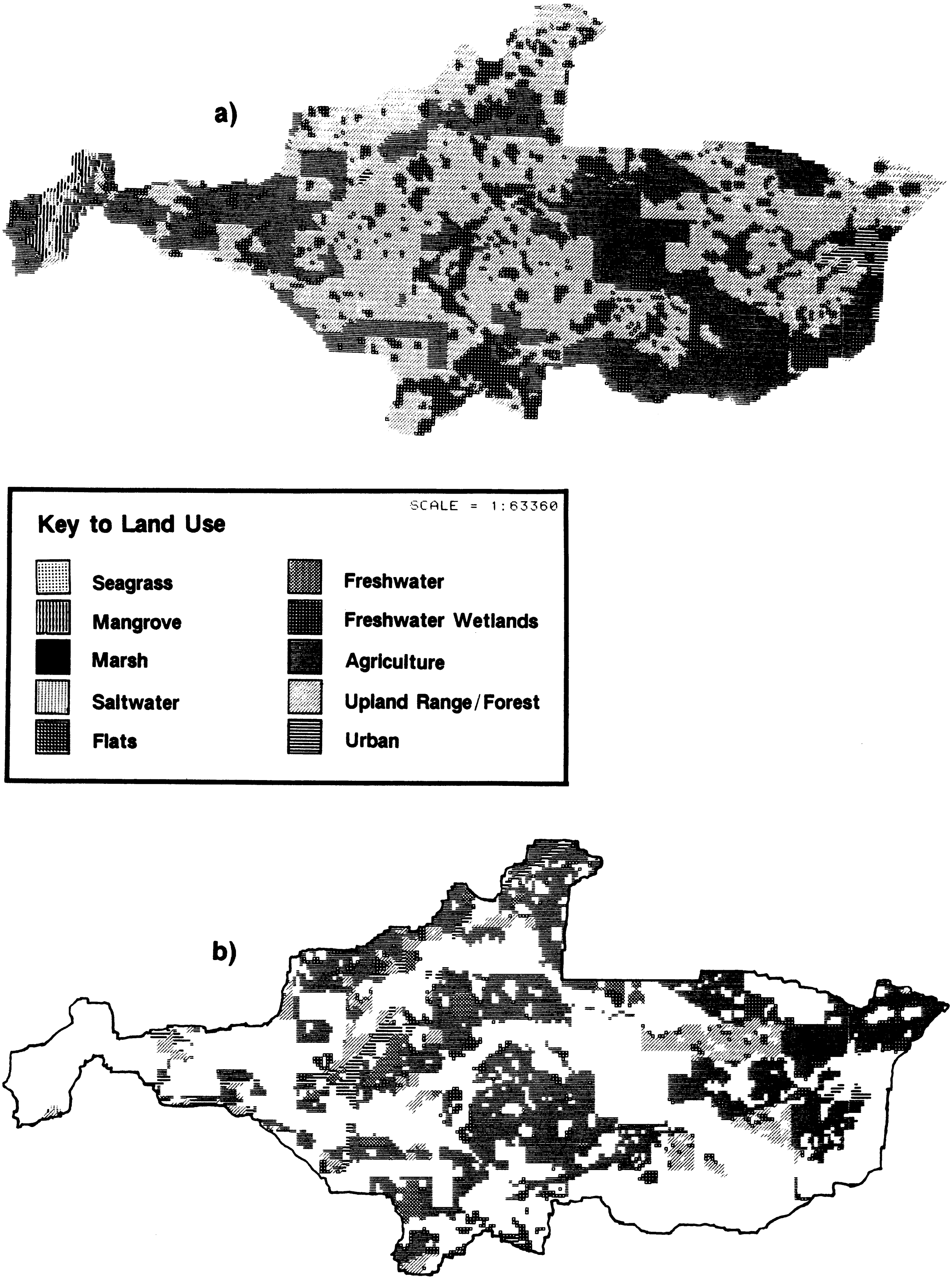


Figure 9. 1950 inventory (a) and 1982 land uses displacing 1950 upland range (b) for Frog Creek.

Table 1. Delaney Creek watershed net land use changes.

LAND USE	1950 HECTARES	1950 % OF WATERSHED	1982 HECTARES	1982 % OF WATERSHED	CHANGE IN AREA (+,- HEACTARES)	% CHANGE IN AREA (+,-)
Water	124.83	2.53	88.83	1.80	-36.00	-28.8
Seagrass	59.58	1.21	0.00	0.00	-59.58	-100.0
Mangrove	42.39	0.86	25.74	0.52	-16.65	-39.3
Saltmarsh	4.50	0.09	0.00	0.00	-4.50	-100.0
Fw. wetlands	367.02	7.43	189.81	3.84	-177.21	-48.3
Agriculture	617.94	12.51	835.38	16.91	+217.44	+35.2
Barren Upland	0.00	0.00	11.88	0.24	+11.88	+100.0
Upland Forest	216.18	4.38	243.81	4.93	+27.63	+12.8
Upland Range	3307.68	66.95	500.31	10.13	-2807.37	-84.9
Urban	200.25	4.05	3044.70	61.63	+2844.45	+1420.4

Table 2. Frog Creek watershed net land use changes.

LAND USE	1950 HECTARES	1950 % OF WATERSHED	1982 HECTARES	1982 % OF WATERSHED	CHANGE IN AREA (+,- HEACTARES)	% CHANGE IN AREA (+,-)
Water	30.42	0.57	156.42	2.96	+126.00	414.2
Seagrass	0.00	0.00	0.00	0.00	0.00	0.0
Mangrove	69.03	1.30	110.79	2.09	+41.76	60.5
Saltmarsh	0.00	0.00	0.90	0.02	+0.90	100.0
Fw. wetlands	930.96	17.59	804.87	15.21	-126.09	-13.5
Agriculture	1663.65	31.44	3275.82	61.90	+1612.17	96.9
Barren Upland	0.00	0.00	47.43	0.90	+47.43	+100.0
Upland Forest	240.75	4.55	176.04	3.33	-64.71	-26.9
Upland Range	2308.32	43.62	434.07	8.20	-1874.25	-81.2
Urban	48.60	0.92	285.39	5.39	+236.79	487.2

primarily from agricultural development and upland range encroachment (Figure 10). A net loss of 311 acres (126 hectares) occurred basin-wide for freshwater wetland vegetation.

Evaluation of the developmental trend analysis for both watersheds identify that wetlands not only suffer from intensive urbanization activities such as in Delaney Creek, but also from agricultural practices as in the Frog Creek watershed.

DISCUSSION

For the purposes of this report, cumulative impacts are defined as the total interactive impacts over time, i.e., the sum incremental synergistic effects on fish and wildlife populations and habitat caused by all current and future action over time and space (Cline et al. 1983).

Currently, the Florida Department of Environmental Regulation (FDER) is required to review cumulative impacts in wetlands under section 403-919 Florida Statutes, which reads:

> The department, in deciding whether to grant or deny a permit for an activity which will affect waters, shall consider:
>
> (1) the impact of the project for which the permit is sought.
>
> (2) The impact of projects which are existing or under construction or for which permits or jurisdictional determinations have been sought.
>
> (3) The impact of projects which are under review, approved, or vested pursuant to s. 380.06, or other projects which may reasonably be expected to be located within the jurisdictional extent of waters, based upon land use restrictions and regulations.

Clearly, this section gives FDER the authority during permit assessment to review the potential for cumulative impact without providing the methodology necessary for such assessment (Hamann 1986).

The need for assessment of specific cumulative environmental impacts of land and water development has long been recognized, but seldom been adequately addressed for a variety of social, economic, legal, institutional and technical reasons. Development of a methodology for watershed level cumulative impact assessment requires review of existing programs currently in place.

The FDER often conducts a detailed analysis of existing waterbodies to determine the effects existing and potential effluent discharges can produce, termed a "wasteload allocation." A wasteload allocation is more specifically defined by Canter (1986) as "an amount

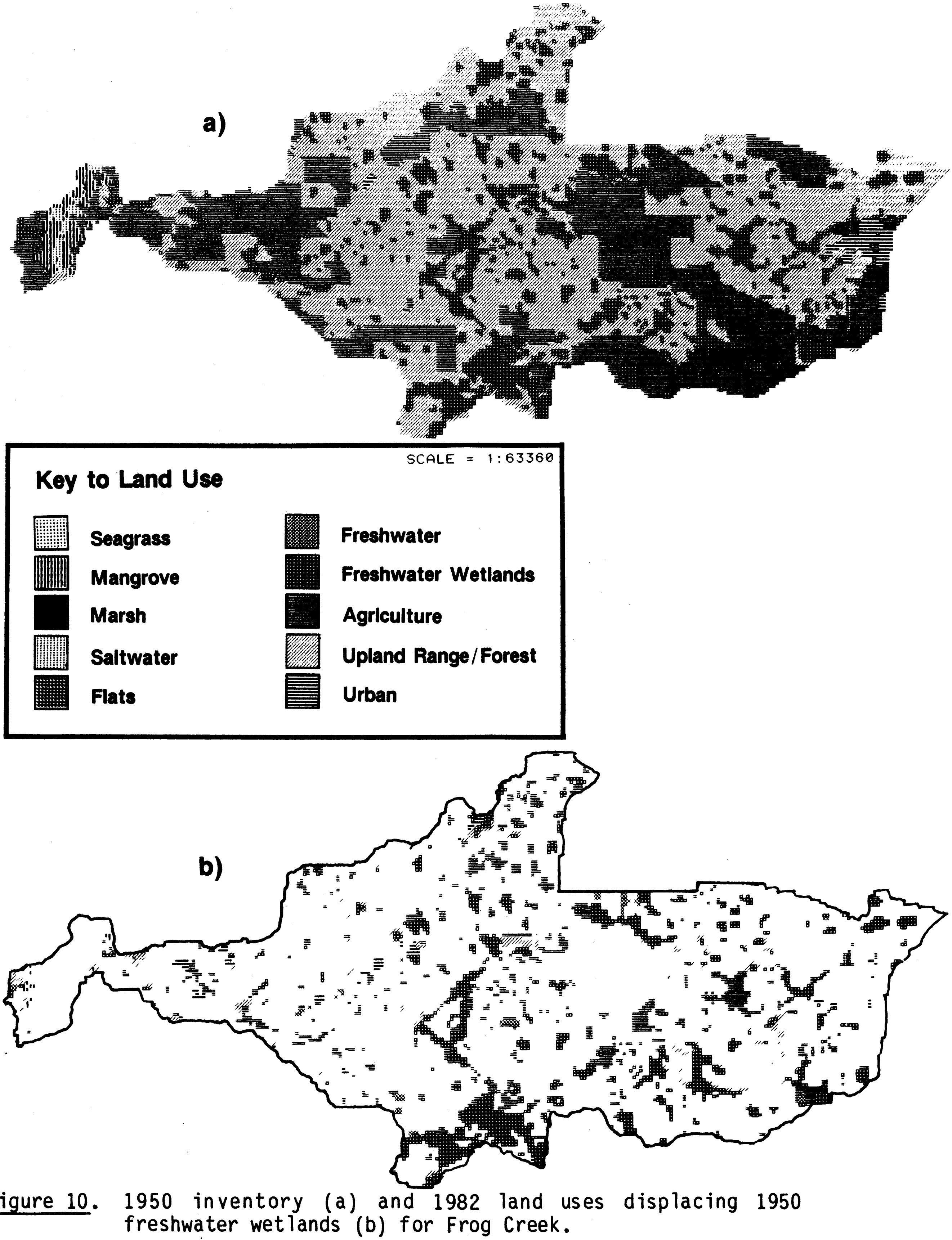


Figure 10. 1950 inventory (a) and 1982 land uses displacing 1950 freshwater wetlands (b) for Frog Creek.

of a particular pollutant, measured in volume or concentration over time, that can be discharged to the applicable water quality standard." A wasteload allocation is a form of cumulative impact assessment by establishing threshold levels--in this case the FAC water class designation--and determining the effects of multiple discharge scenarios on the background receiving water quality. However, the procedure requires extensive analytical analysis and computer modeling, and often cannot consider all of the factors involved with natural system dynamics. In addition, the analysis only includes the direct implications without further identifying the synergistic and indirect impacts created by single or multiple discharges.

Air quality analysis uses a technique similar to that employed in wasteload allocation determinations. The Environmental Protection Agency has established six "criteria pollutants" nationwide and specific localities are designated as "attainment," "non-attainment" or "unclassified" with respect to each of the criteria pollutant parameters. The regulatory strategy used in the classification and permitability of areas places emphasis on cumulative effects of all existing emissions in review of proposed additional air emissions (Canter 1986).

Water management authorities are also required to consider cumulative effects in the regulation of water consumption. Consumptive Use Permits (CUPs) consider background water conditions, existing water consumption and anticipated results of additional withdrawals. In addition, threshold values have been implemented to prevent excessive water consumption which would hasten saltwater intrusion, wetland dehydration or reduction of lake elevations.

The first priority of cumulative impact management is the shift of emphasis by the regulatory community from the "piece-meal" analysis to the systems approach, or the review of developmental impacts on environmental function rather than a review of loss of acreage. Protection of riverine and coastal resources, such as wetlands, requires improved methods for determining the cumulative impacts derived from land development. The reliance on mitigation measures or direct land acquisition programs will not, by themselves, be enough to ensure long-term wetland protection. Similarly, project-by-project impact assessment does not provide a basis for estimating the watershed wide impacts of land development (Dickert & Tuttle 1985).

Due to the hydrological processes, the use of the watershed delineation as the management boundary is necessary. The watershed can further be divided into sub-basins for planning and management purposes. Within the watershed, initial analysis can include:

- current level of land development impacts
- ecological integrity of the ecosystems
- range of important species utilizing the watershed at any point in the species' life history
- individual wetland status, and

- characterization of abiotic factors influencing the watershed (hydrology, topography, soils, meteorology).

This particular task element will require intensive data gathering and analysis to provide sufficient information on which intelligent and equitable decisions can be based. In addition, available historic data for similar parameters is required to determine natural and developmental trends.

The information can then be entered into a data base (e.g.: geographic information system) to be overlayed or drawn from. It is imperative that the information be in a form that can be easily accessed by individual reviewers, since the majority of permit reviews occur within a very short period of time.

The most difficult aspect of cumulative impacts assessment is the determination of acceptable threshold levels of perturbations to natural systems. The evaluation should be based on the historic trend analysis wherever possible. This feature potentially avoids the problems inherent in deriving intrinsic ecosystem thresholds by substituting a historic trend target based on extrapolation of known levels of land use impact (Dickert & Tuttle 1985).

In addition, since the information has been overlayed within the data base, the analysis of trends can easily be acquired and implemented in the permitting process. The development of the threshold levels, however, will require: agreement within the scientific community; public hearings to inform and acquire necessary feedback for implementation; and incorporation into comprehensive plans, coordination of drainage plans and inclusion into regulatory rules and procedures.

Finally, the use of mitigation can be employed to prevent additional degradations and to reverse the trend of cumulative impacts. Drainage basins with adequate quantities of vegetation and habitat, for example, can be replaced equally. Degraded areas require higher replacement ratios to restore historic watershed functions. Efforts to purchase, preserve and restore natural or degraded areas can additionally aid in minimum threshold level achievement.

An important consideration of management needs must include the analysis of existing laws concerning management of agricultural lands, since such drainage and channelization practices have impacted the majority of tributaries to Tampa Bay. Regulations are necessary to prevent additional perturbations without restitution. Incentives are also imperative for private landowners to restore the function of tributaries and drainageways.

The use of the recommended cumulative impact management scheme can be generally applied to the watersheds of Delaney and Frog Creeks. The trend analysis for Delaney Creek identifies a complete loss of seagrass and saltmarsh estuarine habitat and a major decline in freshwater wetland systems (48.3%). Through a series of technical seminars and public workshops, threshold values can be established to identify required habitats needed to improve conditions within the stressed tributary, on a subbasin level. Once threshold values are established, individual permit applications can be reviewed within the context of the entire subbasin and watershed inventories. Mitigation can be used in incremental ratios, to increase wetlands and habitats toward the minimum threshold value. In addition, keystone and unique habitats would receive protection from development that would alter their character or function.

The freshwater segment of Delaney Creek is entirely channelized. Future development can have a positive impact on the system by creating meanders, littoral shelves and other habitat components to improve the natural system while maintaining drainage requirements. The lower estuarine segment has already experienced extensive urbanization and will require restoration actions to supplement any form of mitigation enhancement undertaken.

To a lesser extent, Frog Creek has lost quantities of freshwater wetlands and has actually experienced a gain in acreages of mangrove (+41.8 hectares) (Table 2) between 1950 and 1982. Using subbasin level detail, threshold values can be established while required mitigation ratios to achieve threshold levels will be respectively lower. In subbasins where habitats are above identified threshold values, the mitigation ratios would be at least one-for-one replacement-for-impact, to compensate only for proposed environmental perturbations.

Managing cumulative impacts in tidal tributaries classified in natural conditions should place emphasis on preservation of existing conditions and improvements where applicable. Tidal tributaries in restorable condition should have the highest priority to prevent further losses of environmental systems and a focus of energies on improvement. The stressed tidal tributaries will require a long-term commitment for improvement and an immediate effort to prevent further impacts to the receiving water body.

SUMMARY

The condition of tidal tributaries is dependent on the extent of development activities and natural systems. Continued permitting of "piece-meal" development normally cannot fully consider the impact to the entire watershed unit. A means of assessing impacts, both existing and future, of proposed projects is necessary to prevent degradations and to improve conditions where feasible. The use of geographic information systems and the associated query of data base information is the first step in providing readily available watershed level information. The establishment of threshold levels for habitat, vegetational communities, water quality and consumptive uses will determine target values for improvements. Mitigation standards and public restoration/preservation programs are the basic tools needed to reverse the trend of declining tidal tributary conditions.

ACKNOWLEDGEMENTS

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LITERATURE CITED

Canter, B. 1986. Models for cumulative effects management in non-wetland regulatory programs. In E. D. Estevez, J. Miller, J. Morris and R. Hamman (Eds.), Proceedings of the Conference: Managing Cumulative Effects in Florida Wetlands (pp. 205-296). October 1986. New College Environmental Studies Program Publication No. 37. Omnipress, Madison, Wisconsin.

Cline, E. W., E. C. Vlachos, and G. C. Horak. 1983. State-of-the-art and theoretical basis of assessing cumulative impacts on fish and wildlife. Prepared for U.S. Department of the Interior. Dynamac Corporation, Fort Collins, Colorado, 69 pp.

Dickert, T. G., and A. E. Tuttle. 1985. Cumulative impact assessment in environmental planning - A coastal wetland watershed example. Environ. Impact Assess. Rev. 1985, 5:37-64.

Florida Department of Administration, Division of State Planning and Bureau of Comprehensive Planning. 1977. The Florida Land Use and Cover Classification System: A Technical Report.

Hamann, R. G. 1986. Summary of legal issues, history and rules. In E. D. Estevez, J. Miller, J. Morris, and R. Hamman, Proceedings of the Conference: Managing Cumulative Effects in Florida Wetlands (pp. 219-221). New College Environmental Studies Program Publication No. 37. Omnipress, Madison, Wisconsin.

Tampa Bay Regional Planning Council. 1983. Tampa Bay management study. Tampa Bay Regional Planning Council, St. Petersburg, Florida, 129 pp.

Tampa Bay Regional Planning Council. 1986. Ecological assessment, classification and management of Tampa Bay tidal creeks. Tampa Bay Regional Planning Council, St. Petersburg, Florida, 164 pp.

Tampa Bay Regional Planning Council. 1987. Assessing cumulative impacts on tidal creek watersheds. Tampa Bay Regional Planning Council, St. Petersburg, Florida, 225 pp.

THE EVOLUTION OF MARSH MANAGEMENT PRACTICES IN SAINT LUCIE COUNTY, FLORIDA

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ABSTRACT

Salt-marsh management for mosquito and sandfly control in Saint Lucie County began in 1927 with a massive (475 kilometers) explosive and hand-ditching program. This effort was an attempt to control the salt-marsh mosquito, *Aedes taeniorhynchus*. In 1935, an experiment initiated in Saint Lucie County studied diking of 335 hectares as a method to control sandflies. Initially the experiment was designed to dewater the marsh. However, when this failed, the pumps were rotated and used instead to flood the marsh. Successful control of both sandflies and mosquitoes was achieved. However, further source reduction work did not resume until 1958, and the District relied on chemical control alone (primarily DDT), during the early 1950s. When the permanent control program was reinstated, diking (impounding) of the remaining 2025 hectares of salt-marshes in Saint Lucie County began, and was completed by 1967. However, the diking segregated the salt-marsh habitat from the lagoon, and the flooding and trapping of rainfall resulted in the decimation of the natural high marsh flora (which could not sustain the state of continuous flooding nor the elevations at which the flooding waters were maintained). In this early period, solitary culverts were installed to allow some connection during the fall months. However, in Saint Lucie County, this limited exchange came to an end in the mid-1970s. In 1983, the first Rotational Impoundment Management Plan (RIMP) was drafted, which outlined a management scenario calling for approximately four months of management and 8 months of tidal exchange. Since that initial development-related project, the Saint Lucie County Mosquito Control District and the Florida Department of Health and Rehabilitative Services, have combined to restore 1336 hectares of managed impoundments to the RIM system in Saint Lucie County. Currently, 101 perimeter dike culverts have been installed in managed impoundments, and permits are in various stages of processing for 34 more. Current best management practices include the use of aerial larvicides such as *Bacillus thuringiensis israelensis* (Vectobac), and Altosid (Altosand) Liquid Larvicide (methoprene) for the control of mosquito larval stages during the unmanaged period.

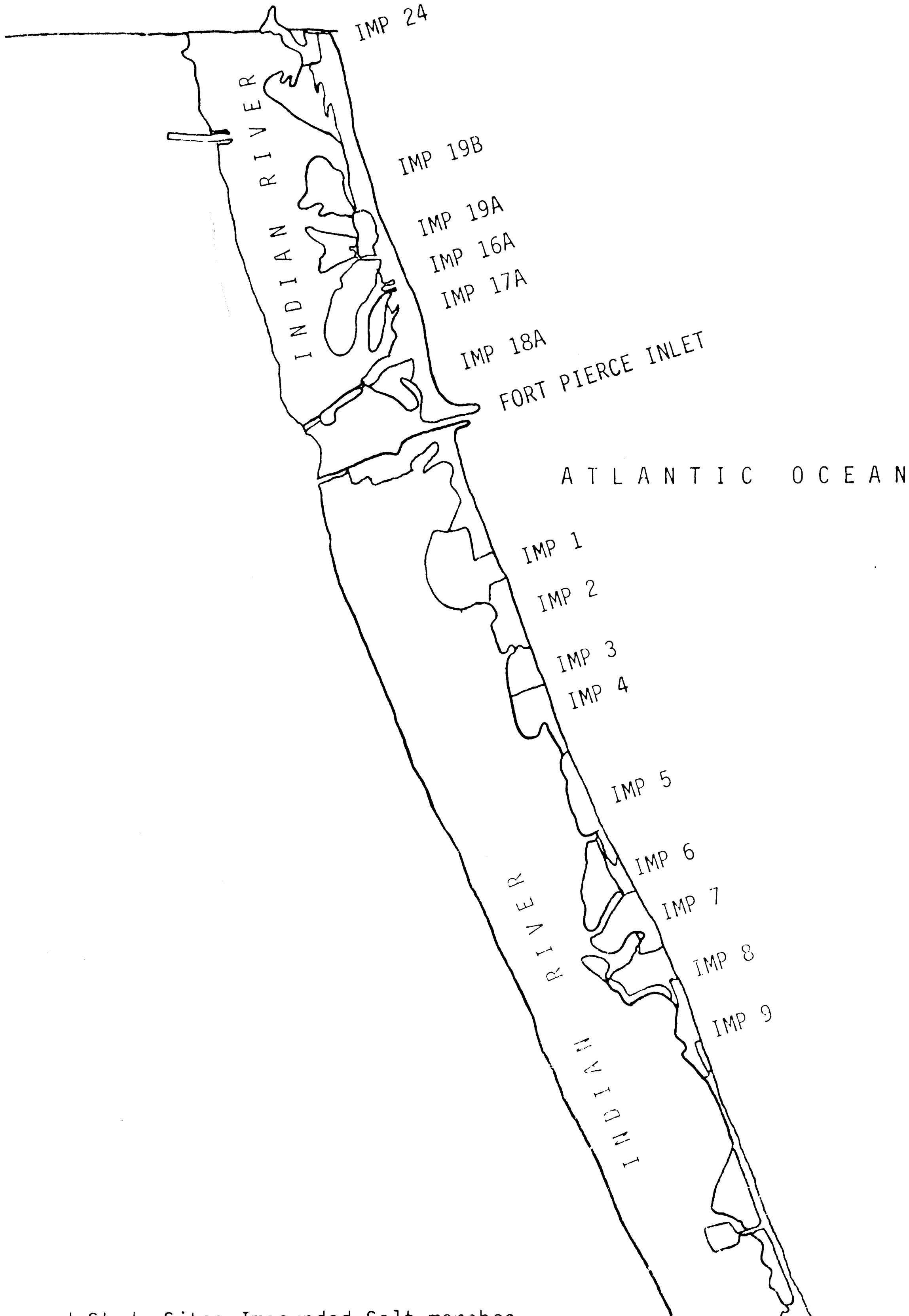
INTRODUCTION

The History of Ditching and Diking in Salt-marshes Located on a Barrier Island Adjacent to the Indian River Lagoon in Saint Lucie County, Florida

Approximately 2430 hectares of high marsh habitat was present in Saint Lucie County, when the thrust for development began in the 1920s. In 1925, Col. Wm. D. Wrightson was solicited for his advice on how to eradicate the mosquito problem. Col. Wrightson suggested three possible courses, filling, ditching and diking. The County decided to begin a ditching program in 1927, and by 1935, had completed 475 kilometers of ditches. At that point an unforeseen complication arose. The ditching brought permanent water into the marshes and provided optimum conditions for repeated sandfly broods. Ditching was also shown to concentrate 95% of the sandfly breeding within 0.4 m of the ditch banks (Hull & Dove 1939). Studies by Linley (unpublished) showed that for 0.6 m wide ditch banks (vertical ditch banks), up to 123,200 salt-marsh sandflies (per day per 0.83 km of ditch), Culicoides furens (the fire biter), can be produced. For 1.51 m wide ditch banks, that number increases to 308,100/day/0.83 km of ditch. Multiplying the lower of the two values (123,200) by the number of kilometers of ditches (475), gives the value of 70,506,024 sandflies produced per day (all of which were capable of migrating 3.3 - 8.3 kilometers). Multiplying the high value (308,100) by 475 kilometers results in a total of 176,322,289 sandflies per day. The above values reflect averages over the entire year. However, the salt-marsh sandflies produce synchronous broods on 30-60 day cycles during the summer and fall. This means that upwards of 5-12 billion sandflies could be produced during each of the five (approximately) summer and fall synchronous broods produced each year.

The amount of sandfly control necessary depends upon the amount of sandflies present. In Jamaica, 3,000 bites per hour have been recorded, and it has been reported that over 99.5% control would have to be achieved for tourists to support the area (approx. 5 bites/hr) (Linley & Davies 1971). This is a level of control that cannot easily be achieved through the use of pesticides. In addition, those pesticides that were found to be effective, included ingredients such as creosote, which is extremely toxic to the marine environment, and dieldrin, which was responsible for a massive fish kill when it was tested (Harrington & Bidlingmayer 1958).

In 1935, an experimental diking program for the control of sandflies was begun in Saint Lucie County (Hull & Dove 1939). Pumps were installed after a dike was constructed out of dredged marsh material, and an attempt was made to drain the marsh. Significant sandfly control was achieved after the pumps were reversed and used to pump water onto the surface of the marsh to create an artificial lake (Hull & Dove 1939). Similar results were reported by Hull and Shields (1943) and by Rogers (1962). It was discovered that this artificial flooding method also achieved 99% control of the salt marsh mosquitoes



Managed Study Sites Impounded Salt-marshes
Saint Lucie County, Florida

as well. Further work on impounding did not take place until the 1950s because of the advent of World War II. In 1954, the Brevard County Mosquito Control District began construction of impoundments primarily for mosquito control (sandflies were apparently more of a problem for counties near inlets). Shortly thereafter, Indian River County, Volusia County and Saint Lucie County (in 1958), also began salt-marsh impounding programs. By the end of 1966, 30 impoundments had been constructed in Saint Lucie County.

Past Vegetation Impacts

The natural vegetation of the preimpoundment era was predominantly scattered black mangrove trees, *Avicennia nitida*, separated by open fields of saltwort, (*Batis maritima*), glasswort, (*Salicornia* spp.), and salt-barren. A thin fringe (approximately 50' wide) of red mangroves, *Rhizophora mangle*, occurred along the Indian River edge of most of the marshes, and stands of white mangroves, *Laguncularis racemosa*, and buttonwood, *Conocarpus erectus*, grew along the upland edges and in the high elevation portions of the salt-marshes (1944 aerial photos; Hull & Shields 1943; Hull & Dove 1939; Provost 1973; Provost 1957).

Natural Tidal Regime

Tidal water rarely penetrated into the salt-marshes of Saint Lucie County except for 4-6 weeks in the fall, usually beginning with the fall equinox (Clements & Rogers 1964). This resulted in the growth of what Provost (1977) referred to as high marsh vegetation and habitat (the terminology describing the vegetation types and natural flooding periodicities employed by Provost are still in use today). Salt-marsh mosquitoes breed in areas of the salt-marsh which are not inundated more than 4 days per month, 2 days on each spring tide (Provost 1977). In the 1957 report by the Entomological Research Center in Vero Beach (now the Florida Medical Entomological Laboratory affiliated with IFAS), the mosquito breeding areas were described as consisting of the entire high marsh area, which they stated lay between MHW (mean high water) and Max. HW (maximum high water). Provost (1977) stated that where the tidal range is approximately 0.4 m or less (such as that found in the Indian River) over 65% of the total marsh area is high marsh habitat. Provost (1973) also described intertidal profiles in Florida, and stated that although much of the high marsh substrate may lie below the elevation of MHW, a natural levee is created by wave-action at the waterward limit of the high marsh, cutting it off from tidal connection at water levels less than MHW (Provost 1973).

Marsh Natural Fish Utilization

Mr. William Bidlingmayer (retired), a co-worker of Harrington, the fishery biologist at the Entomological Research Center who studies the salt-marsh fishes in the pre-impoundment era, has reported that the marshes were never sampled for fishes during the summer months because there was no permanent water to sample (personal communication). This lack of marsh fishery during the summer months was due to the lack of tidal inundation of these areas, and the lack of regular tidal inundation resulted in significant mosquito breeding.

The salt-marsh mosquito breeding season in the natural marshes normally began in March or April and continued throughout the summer and fall until the middle of October. Thus, breeding closely follows the range of seasonal high tide fluctuation in the Indian River.

Environmental Impacts of the Original Marsh Impounding Practices and the Influence of Current Research on Present Best Management Practices in Saint Lucie County

Prior to impoundment of the salt-marshes of the Indian River, salt-marsh fishery research was performed by Harrington and Bidlingmayer (Entomological Research Center). As stated previously, sampling was not performed during the summer mosquito breeding season, because there was no permanent water in the marshes at this time (Bidlingmayer, personal communication). This is a major point of contention was some agencies, since fishery data does not exist for the summer months for the pre-impounded natural marshes. However, the Mosquito Control Districts' management programs are based on the principle that the seasonally low water period experienced every summer in the Indian River, generally from May to August, is a period in which no significant tidal input ever occurred in the natural marshes. Therefore, closure during that period should have reduced impact on marine organism migration (related to tidal exchange).

However, the impacts of the original salt-marsh impounding upon the natural vegetation in the marsh were significant. The natural salt-marsh vegetation was destroyed because the dikes were kept closed year-round, and flood levels exceeded those that the salt-marsh flora could sustain on a permanent basis. In addition, fishery data available on the seasonal high-water period in the preimpoundment and early postimpoundment era, documented a decline in species diversity in the fall-flooding period from 16 species to 5, following impounding (Harrington & Harrington 1982).

The marshes remained denuded and permanently flooded in Saint Lucie County until the early 1970s. At that time, red mangroves began to encroach upon the high marsh because the area was flooded nearly year-round, and because the red mangroves had gained a foothold along the ditches dug prior to the impounding. There was also almost no other source of vegetative competition, since the natural vegetation had been eliminated. The red mangroves then began to build-up peaty substrate, consisting largely of leaf-litter and fibrous root systems, and spread throughout the marsh surface to create nearly 100% coverage. This high percent coverage apparently results in a restriction in the amount of light penetrating to the marsh water, which can severely limit the oxygen production by phytoplankton and algae during the closed managed period.

In 1979, Gilmore (Harbor Branch Oceanographic Institute), began fish sampling in the same marsh sampled by Harrington, 23 years earlier. Gilmore (1981) found 12 species of marine fishes in impoundments which were closed. Gilmore (1981) also studied a second impoundment in connection with the marsh. This second marsh was open to the Indian River through a single 75 cm diameter culvert. In this open, culverted marsh, Gilmore found 38 species of fish, and also found that the high marsh vegetation was recovering from the flooding conditions of the original impounding. Furthermore, transient fishery use during the summer lower high tide periods was comparatively reduced (Gilmore 1981), being primarily limited to the man-made perimeter ditch and natural tidal creeks which existed within the impoundment.

As a result of these data collected by Gilmore on fishery use of culverts and natural marsh revegetation, a culvert installation program was begun in Saint Lucie County (initially as part of developer mitigation).

Theoretically, culverts would allow unrestricted use of the impoundments by the marine fishes for 8 months per year, and still accomplish mosquito control during the 4-month-long mosquito breeding season. Two fish research projects (of 2 years duration each), have since been completed in two multi-culverted impoundments (#2 and #16A) in Saint Lucie County. A third project involving 5 years of data collection has been performed in Impoundment #12 (in Indian River County) (Gilmore, unpublished).

Gilmore (unpublished) found that in a red mangrove marsh which was studied (Impoundment #2, Green Turtle Beach), 94 species of fishes and macrocrustaceans were using the impoundment culverts after sufficient new culverts (1 culvert/10.9 hectares) were installed to approximate the tidal range. The red mangrove impoundment substrate was found to be used almost exclusively by marsh resident species of small minnows of the family Poeciliidae, *Gambusia affinis*. The remaining 73 resident fishes and transient fishes of commercial and sport value were found to use the man-made perimeter ditches, as well as the natural ponds having perimeter ditch connection.

In marshes that were not dominated by the red mangroves (impoundments #16A and #12), a closely related (to *Gambusia affinis*) marsh resident species of minnow of the family Cyprinodontidae, *Cyprinodon variegatus*, dominated the marsh surface collections. The remaining 50 species of marsh residents and transients of commercial and sport value collected in these multi-culverted marshes, were primarily captured in the man-made perimeter ditches (similar to the collections made in the red mangrove impoundment study). However, there were 6 species of marsh residents and marsh associates that apparently also used the marsh surface in the non-red mangrove dominated high marsh.

In addition, Gilmore (unpublished) found that the three most numerically abundant species of fishes in all of the marsh studies, were consistently the same three species of marsh residents, *Cyprinodon variegatus*, *Gambusia affinis* and *Poecilia latipinna* (the sheepshead minnow, mosquito fish and the sailfin molly respectively). The percent abundance in each marsh of these top three species was 69.6% in impoundment #2 (red mangrove vegetation), 94.2% in impoundment #16A (mixed vegetation), and 92.9% in impoundment #12 (unvegetated) (Gilmore, unpublished).

In terms of overall abundance, the addition of multiple culverts resulted in increased use of the marshes by transient fishes of commercial and sport value, far exceeding that which occurred under the permanent flooding management strategy. The transients primarily were collected in the marsh perimeter ditches (Gilmore, unpublished).

The marsh residents apparently remain associated with the marshes throughout their lives, and spread over the entire marsh surface when it floods (artificially or naturally) (Gilmore, unpublished). Population sizes of marsh resident fishes collected by Gilmore et al. (unpublished), in 1984, revealed increases of up to 34-fold from the beginning of closure, to the fall drawdown. In 1985, resident fishery population sizes increased 29-fold over the same period (Gilmore, unpublished).

Current Vegetation Impacts

Natural high marsh plants appear to be revegetating those impoundments which are not completely invaded by red mangroves and which remained denuded to some degree. Impoundment #16A was partially denuded in the southwest cell at the start of RIM management in 1982. Now, after three years of RIM, black mangroves, white mangroves, saltwort and glasswort are all revegetating the area (approximately 20% coverage at this time). Impoundment #18A was completely denuded in 1978. Now, after three years of RIM management, it has achieved approximately 80% coverage by white and black mangroves.

Lahman (unpublished ph.D. University of Miami) completed a study comparing red mangrove growth in a managed marsh, unmanaged marsh and in a fringing marsh along the Indian River. Results of his data show that the managed marsh studied produced similar quantities of litter compared to the unmanaged marsh studied. Tree growth, according to Lahman (unpublished) was not different between the managed and unmanaged marsh. Lahman (unpublished) also found no appreciable buildup of substrate elevation in the managed marsh studied. This last agrees with unpublished data from impoundments sampled for subsidence by the Saint Lucie County Mosquito Control District.

Current Water Quality

Water quality research by Montgomery et al. (unpublished) found that dissolved oxygen in an unmanaged marsh was not significantly different from a managed marsh during the closed, managed period, and was significantly better in the multi-culverted perimeter ditch of the managed marsh during the open period, than in an unmanaged marsh perimeter ditch fed by a breach in the dike.

In addition, Montgomery et al. (unpublished) also found no significant net daily transport of dissolved and particulate organic compounds from the managed and unmanaged marshes (prior to the initiation of continuous bottom-water release).

Ongoing research by Dr. Paul Carlson of the Department of Natural Resources, has quantified the production of hydrogen sulfide in the managed marshes. This work will contribute to the refinement of the hydrogen sulfide removal technology which Dr. Carlson has helped develop in the impounded marshes in the past two years. The hydrogen sulfide removal technique uses laminar flow to displace pore water and ditch-bottom water from the marshes, during the period in which they are closed. The impact on impoundment water quality is significant, in that dissolved oxygen (determined by a Leeds and Northrup model 8500 portable dissolved oxygen meter) can be maintained at levels greater than 3-4 ppm in a closed impoundment as a result of the bottom-water removal. Dissolved oxygen generally remained at or below 1 ppm during the daytime, prior to the bottom-water release system initiation.

The Effect on the District's Pesticide Use Since the New Impoundment Management Program was Initiated in 1984

Each year, from 1984 through 1986, the District was able to reduce ground ULV (ultra low volume) adulticiding by 81,000 hectares/yr., while providing a significant increase in (and much more pleasant) recreational use of the beaches and Indian River along the barrier island, as a result of improvements in the impoundment management program.

Ground adulticiding treatments along South Hutchinson Island, from Ocean Village to the Nuclear Power Plant, were numerically reduced from 87 treatments in 1984, to 57 treatments in 1985, to 7 in 1986, to 1 for 1987. It should be noted that ground ULV adulticiding costs are extremely expensive (approximately $55.00 per hour of fogging), and such reductions in cost are very important budgetary considerations.

In addition, without the ability to manage the impoundments, aerial adulticiding would have to be performed; 4,000 hectares would require aerial adulticide application each time treatment is required, at a cost of $5,000 - $10,000 per flight. No aerial adulticiding has been performed in Saint Lucie County since the start of the rotational impoundment management program in 1984.

Greater amounts of aerial larviciding would also be required if the impoundments were not managed. Approximately 1,620 hectares would have to be repeatedly larvicided 10-20 times per year, at a cost of approximately $20.62/hectare/treatment (with DNR-approved BTI). Thus, larviciding alone would range in cost from $334,000 to $668,000 per year (based on values for low and normal rainfall and tide years). 1988 aerial larviciding costs in the Rim managed marshes with minimum acres/culvert ratios of 50:1, were 12% of the minimum aerial larviciding cost (calculation as above) required if the impoundments were unmanaged.

Impoundment electric pump operation and maintenance costs for 14 impoundments (1250.4 hectares) are approximately $20,000/month (over a duration of approximately 4 months). These costs do not include purchase and installation of the initial culverts and pumps necessary to perform the Rim Management and restoration.

DISCUSSION

Impoundment management is still in its infancy. Further studies of the impacts of this work are necessary to refine our current techniques. For example, impoundment mariculture and stock enhancement practices may be able to be performed in such a manner as to elevate marsh estuarine energy linkages (by pumping enough water to allow the impoundments to remain open year-round).

Thus far, the Saint Lucie County Mosquito Control District has restored, or is in the process of restoring, 1619 hectares of managed salt-marsh impoundments, and has performed (or is in the process of performing) restoration projects in another 233 hectares of unmanaged salt-marshes. This has resulted in the installation of 101 perimeter dike culverts, with 34 additional perimeter culverts in various stages of permitting (for the managed impoundments). An additional 52 culverts have been installed or are in the permit process, which are internal impoundment culverts, or have been installed/proposed in perimeter dikes of unmanaged salt-marshes.

The goals of the current management are: 1) to continue to seek ways to improve water quality; 2) to improve marine fish and macrocrustacean access; 3) to participate in studies comparing natural systems with our artificial ones; 4) to study the potential benefits of mariculture within the impoundments. We must continue to support and stay abreast of current wetland research, and we must continue to attempt to apply these basic research findings to our management strategies. We believe that there are alternatives and refinements which can continue to be implemented as our knowledge grows from the study of these sensitive wetland areas.

LITERATURE CITED

Clements, B. W., Jr., and J. A. Rogers. 1964. Studies of impounding for the control of saltwater marsh mosquitoes in Florida. Mosquito News 24:265-276.

Dickinson, J. 1697. Johnathan Dickinsons Journal or God Protecting Providence. Yale Press.

Gilmore, R. Grant, D. W. Cooke, and C. J. Donohoe. 1981. A comparison of the fish populations and habitat in open and closed salt-marsh impoundments in east-central Florida. N. Gulf Sci. 5:25-37.

Gilmore, R. Grant, B. McLaughlin, and D. Tremaine. 1987. Fish and macrocrustacean utilization of an impounded and managed red mangrove swamp with a discussion of the resource value of managed mangrove swamp habitat. Unpublished.

Gilmore, R. Grant, P. B. Hood, R. W. Brockmeyer, and D. M. Scheidt. 1987. Final Report to the Florida Department of Health and Rehabilitative Services, Office of Entomology. Effects of increased culvert density on utilization of marsh impoundments by fishes and macrocrustaceans. Unpublished.

Harrington, R. W., Jr., and W. L. Bidlingmayer. 1958. Effects of dieldrin on fishes and invertebrates of a saltmarsh. J. of Wildlife Management 22:76-82.

Harrington, R. W., Jr., and E. S. Harrington. 1982. Effects on fishes and their forage organisms of impounding a Florida salt marsh to prevent breeding by salt marsh mosquitoes. Full. Mar. Sci. 21:523-531.

Hull, J. B., and W. E. Dove. 1939. Experimental diking for the control of sandfly and mosquito breeding in Florida salt water marshes. J. Econ. Ent. 32:309-312.

Hull, J. B., and S. E. Shields. 1943. Diking as a measure for sandfly control in salt marshes. J. Econ. Ent. 36:405-409.

Hull, J. B., and S. E. Shields. 1943. Diking and pumping for control of sandflies and mosquitoes in Florida salt marshes. J. Econ. Ent. 36:409-412.

Lahman, E. J. 1987. Fifth quarterly progress report: Effect of water management regimes on the productivity of red mangroves. Unpublished.

Linley, J. R., and J. B. Davies. 1971. Sandflies and Tourism in Florida and the Bahamas and Caribbean area. J. Econ. Ent. 64:264-278.

Miller, T. W. 1955. Twenty-ninth annual report of the Saint Lucie County Sanitary District. Unpublished.

Montgomery, J. R. 1987. Determination of the impact of flapgate controlled mosquito impoundments on litter and carbon fluxes using a breached impoundment as a control site. Final report to the Florida Department of Health and Rehabilitative Services, Office of Entomology. Unpublished.

Provost, M. W. 1958. Facts about the salt marsh and its mosquitoes. Entomological Research Center Report. Unpublished.

Provost, M. W. 1973. Mean high water mark and use of tidelands in Florida. Fla. Sci. 36:50-66.

Provost, M. W. 1977. Source reduction in salt-marsh mosquito control: Past and future. Mosquito News 37:689-698.

Rogers, A. J. 1962. Effects of impounding and filling on the production of sandflies (culicoides) in Florida salt-marshes. J. Econ. Ent. 55:521-527.

Rogers, A. J., B. W. Clements, Jr., and C. G. Witherington. 1962. Progress report on water management studies to control mosquitoes in impounded salt marshes in Florida. Report of 33rd Annual Meeting of FAMA. Unpublished.

Shingleton, R. D. ______. John Taylor Wood. Univ. Georgia Press. Athens, Georgia.

RESTORATION OF THE CHANNELIZED
CYPRESS CREEK SWAMP BY CONTROL
STRUCTURE INSTALLATION
(SUN CITY, FLORIDA)

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ABSTRACT

In August 1986, the Florida Department of Environmental Regulation permitted construction of a dam on the Cypress Creek Swamp at Sun City, Florida. The goal of the project was to restore the natural floodwater storage capacity of the system to prevent downstream flooding and reestablish the historical hydroperiod. The objectives of this study were: (1) to demonstrate that dam installation raised water levels, and (2) to characterize the vegetative community to determine if the plant community reflected an extended hydroperiod. Three transects with six shallow wells (piezometers) and six 1-m^2 vegetation plots were established. Ground water levels, plant species composition, and percent areal coverage were recorded quarterly. Six monitoring events occurred before, and five events after dam installation. Analysis of the piezometer data revealed that ground water levels increased on two of the three transects. Vegetation analysis using Czekanowski's Index of Percentage Similarity indicated that the plant community changed after dam installation. Further vegetation analysis using hydroperiod categories for plant species did not clarify the nature of the change in the plant community. This study demonstrated that long-term monitoring is necessary for accurate assessment of ecological change in wetland communities.

INTRODUCTION

Channelization of creeks, streams, and rivers historically was performed throughout Florida as a means of alleviating flooding, and improving transportation and trade (Blake 1980). With heightened environmental concern beginning in the early 1970s, attempts were made to reverse the negative effects of channelization on water quality, vegetation, and wildlife. The Cypress Creek Swamp at Sun City, Florida (Lat 27° 44' N, Long 82° 21' W) was channelized in 1962 and is an example of such a system. The Cypress Creek Swamp is roughly 243 hectares in extent, draining a 30.3-km^2 watershed. At flood stage, Bullfrog Creek to the north contributes an additional 38.9-km^2 to this drainage basin (Heidt & Associates 1980). In August 1986, the Florida Department of Environmental Regulation (DER) permitted construction of a dam on the southern extreme of the Cypress Creek system (DER Permit #290661453). The goal of the project was to restore the natural floodwater storage capacity of the Cypress Creek Swamp to prevent downstream flooding south of S.R. 674 (Figure 1). As a result of this action, the historical hydroperiod of this system would be restored. The objectives of this study were:

1. to demonstrate that dam installation did raise water levels on the Cypress Creek Swamp and,

2. to characterize the vegetative community before and after dam installation to determine if the plant community reflected an extended hydroperiod.

MATERIALS AND METHODS

As specified in the DER permit, three east-west transects were established to record changes in ground water levels and plant species composition (Figure 1). Each transect had six piezometers (shallow ground water wells) and six 1-m^2 vegetation plots adjacent to the piezometers. Piezometers consisted of PVC pipe with a screened bottom and cap that was driven into the ground. Each piezometer extended 1.2 meters above ground level. Water level measurements were taken from the top of the PVC pipe and converted to absolute elevations.

Plant species composition and percent areal coverage of herbaceous vegetation were recorded on 1-m^2 plots. Estimates of areal coverage by species were recorded to the nearest 5% from a vertical vantage point. Occasionally, total percent cover exceeded 100% when vegetation was stratified. Plots were located no closer than 30.5 m to each other along transects and usually were more than 76.5 m apart.

The DER permit specified that ecological monitoring was to be conducted at least once before dam installation, and to continue quarterly for three years thereafter. Monitoring began in 1984; six quarterly sampling events (March, June, September and December, etc.) occurred prior to dam installation in August of 1986. Five quarterly sampling events occurred after the dam was built.

Rainfall data were gathered from the National Weather Service station in Ruskin, Florida. This station is located approximately 4.0 km from the project site.

Quarterly piezometer data were analyzed using t tests to compare shallow ground water levels before and after control structure installation. Vegetation data were analyzed using two methods. First, the percent areal coverage of herbaceous species before and after dam installation was compared within plots using Czekanowski's Index of Similarity. Similarity indices were calculated for each plot within each transect by comparing the areal coverage of vegetation before and after dam construction. For example, an index was calculated by comparing the percent cover of vegetation in March for plot 1 Transect A in 1984 (before dam construction) to vegetative cover in March of

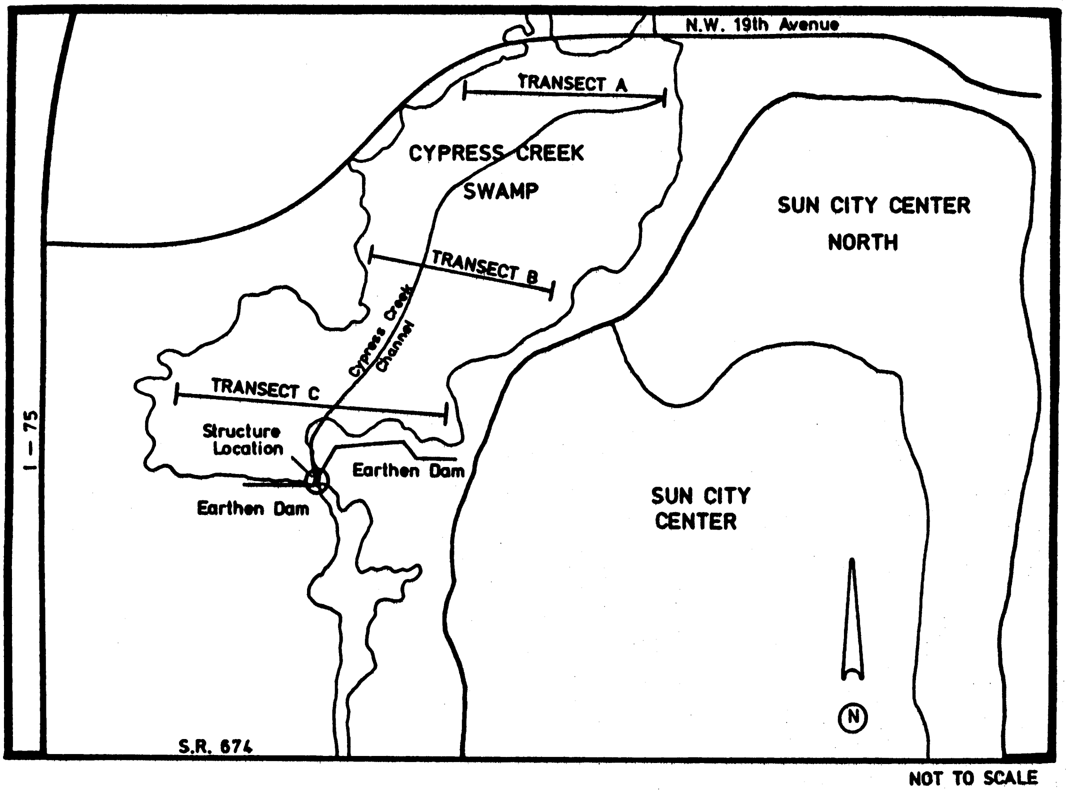
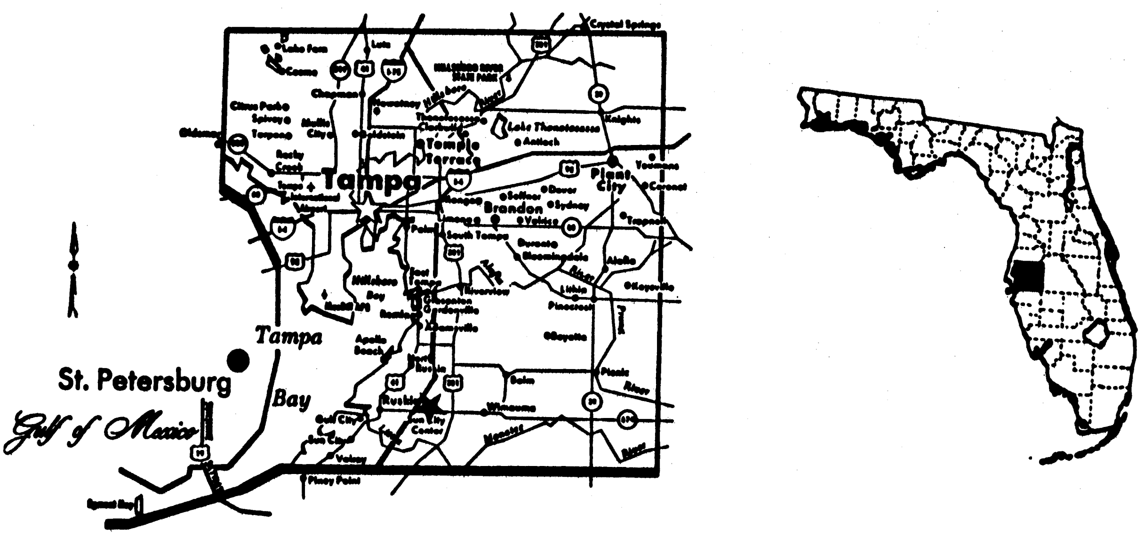


Figure 1. Cypress Creek Swamp and study area.



1985 (also before construction). Similarly, an index was calculated comparing the percent cover in March on this plot before construction to the cover in 1986 or 1987 (after construction). A series of indices for all plots on all transects was calculated. Indices calculated from samples taken entirely before dam construction were compared to indices derived from pre- and post-dam installation sampling events. Tests for significance were accomplished using t tests. Czekanowski's Index (CI) is defined as:

$$CI = \frac{200 \times \sum_{i=1}^{i=s} \min(Xi_1, Xi_2)}{\sum_{i=1}^{i=s} (Xi^1, XI_2)}$$

where s = the number of species found in one or both plots, and Xi_1 and Xi_2 = the percent cover of species i in plots 1 and 2, respectively. The values of CI can range from 0-100%; the higher the percentage, the greater the similarity between plots.

The second method used to evaluate changes in the plant community at Cypress Creek involved comparison of hydroperiod characteristics of plants present before and after dam installation. To accomplish this, all plants in the 1-m^2 plots were assigned a category according to hydroperiod as defined in the Wetland Plant List prepared by the National Wetlands Inventory (USFWS 1986). The proportion of obligate hydrophytes in each plot before control structure construction was compared to the proportion after the control structure was installed; these proportions were compared using t tests. All proportional data were normalized prior to analysis by arcsin transformation.

RESULTS

Analysis of the piezometer data revealed that shallow ground water levels increased significantly on transects A and B (Figures 2 and 3), but not on Transect C (Figure 4) after control structure installation. The variance in piezometer levels was much lower for Transect A (Figure 2) than for either transects B or C (Figures 3 and 4) both before and after installation of the dam. Transect A also was less variable and lower in piezometer elevation (and therefore 1-m^2 plot elevation) than either transect B or C (Figure 5).

Comparisons of the similarity in vegetation on the transects before and after control structure installation showed the same pattern as that observed with the piezometer data. On transects A and B, the composition and percent areal coverage of herbaceous vegetation were significantly different after control structure installation (Figure 6). The herbaceous vegetation on Transect C did not differ relative to dam construction (Figure 6).

Evaluation of the proportion of obligate hydrophytes on each

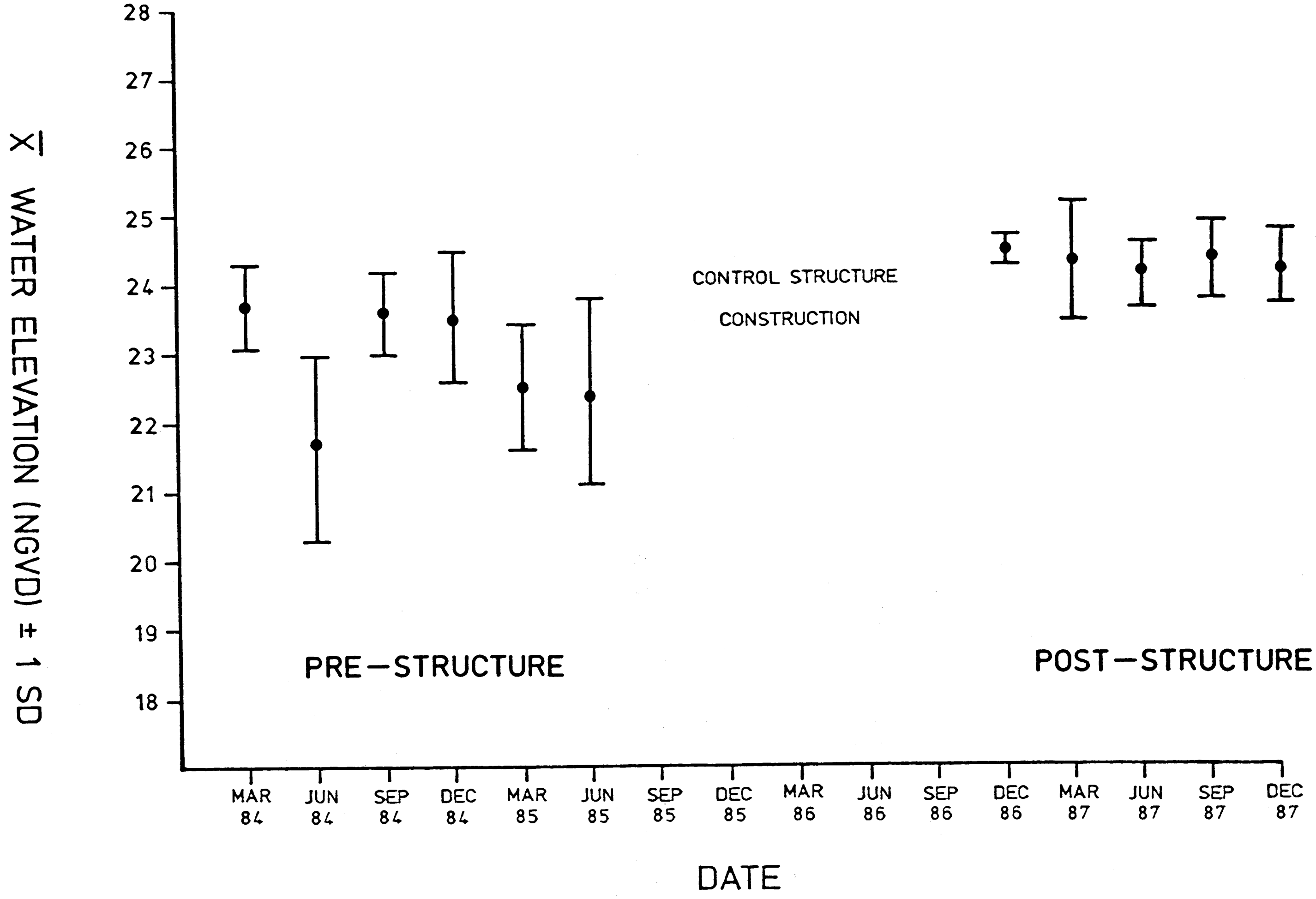


Figure 2. Piezometer levels on Transect A during the study period. Piezometer levels before structure installation were lower than those after structure installation ($P<0.05$).

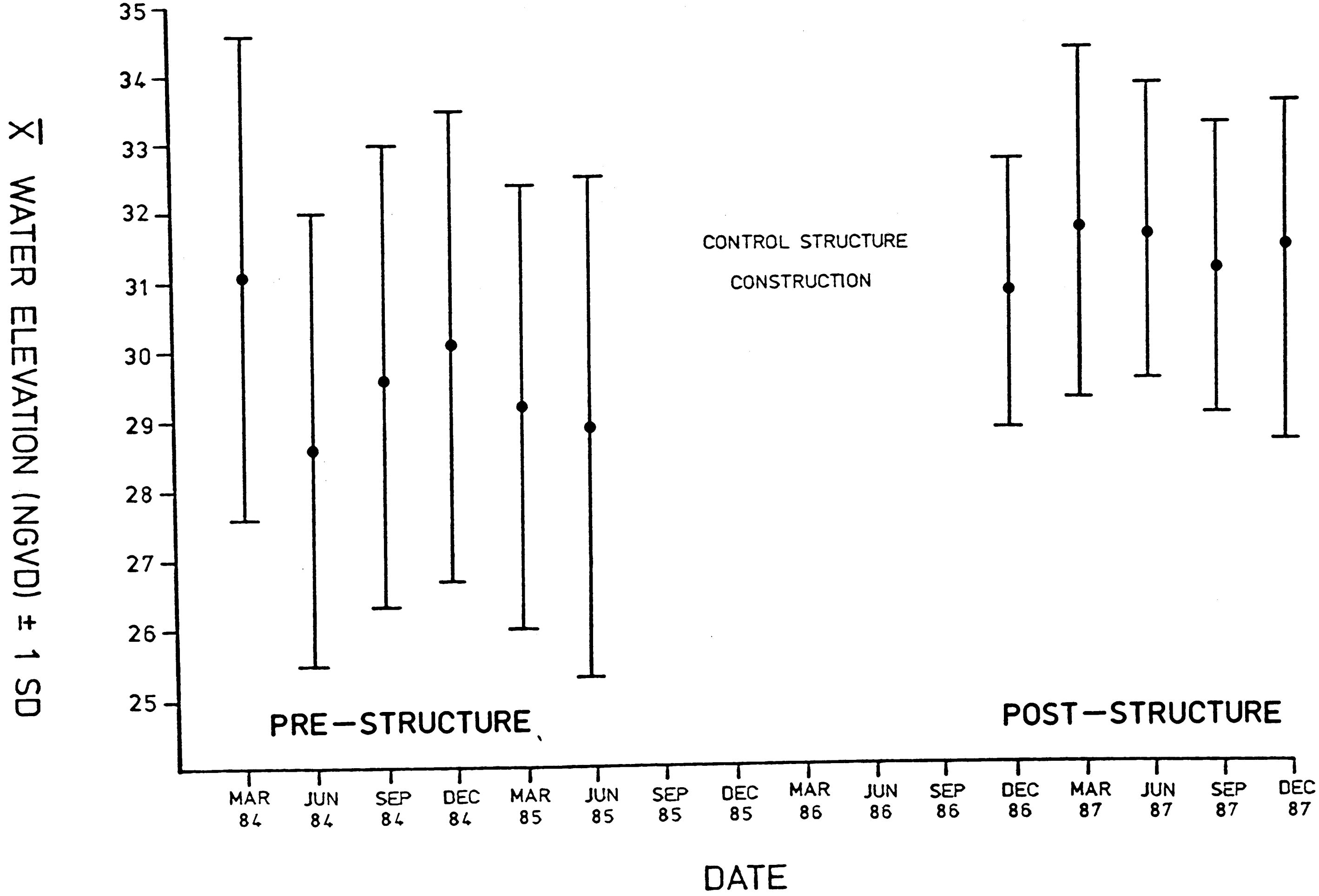


Figure 3. Piezometer levels on Transect B during the study period; piezometer levels before structure installation were lower than those after structure installation ($P<0.05$).

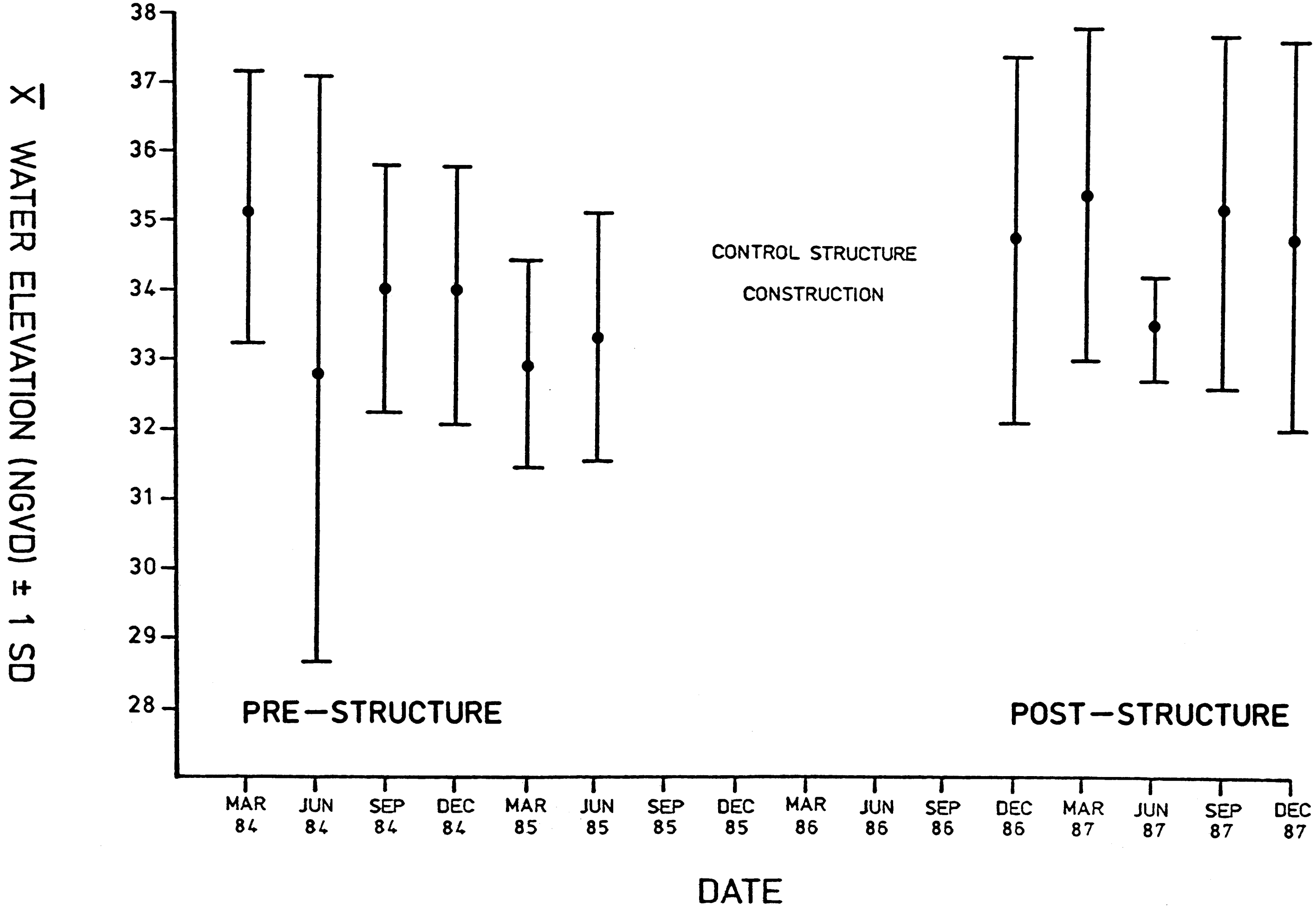


Figure 4. Piezometer levels on Transect C during the study period; levels did not differ before and after structure installation (P=0.088).

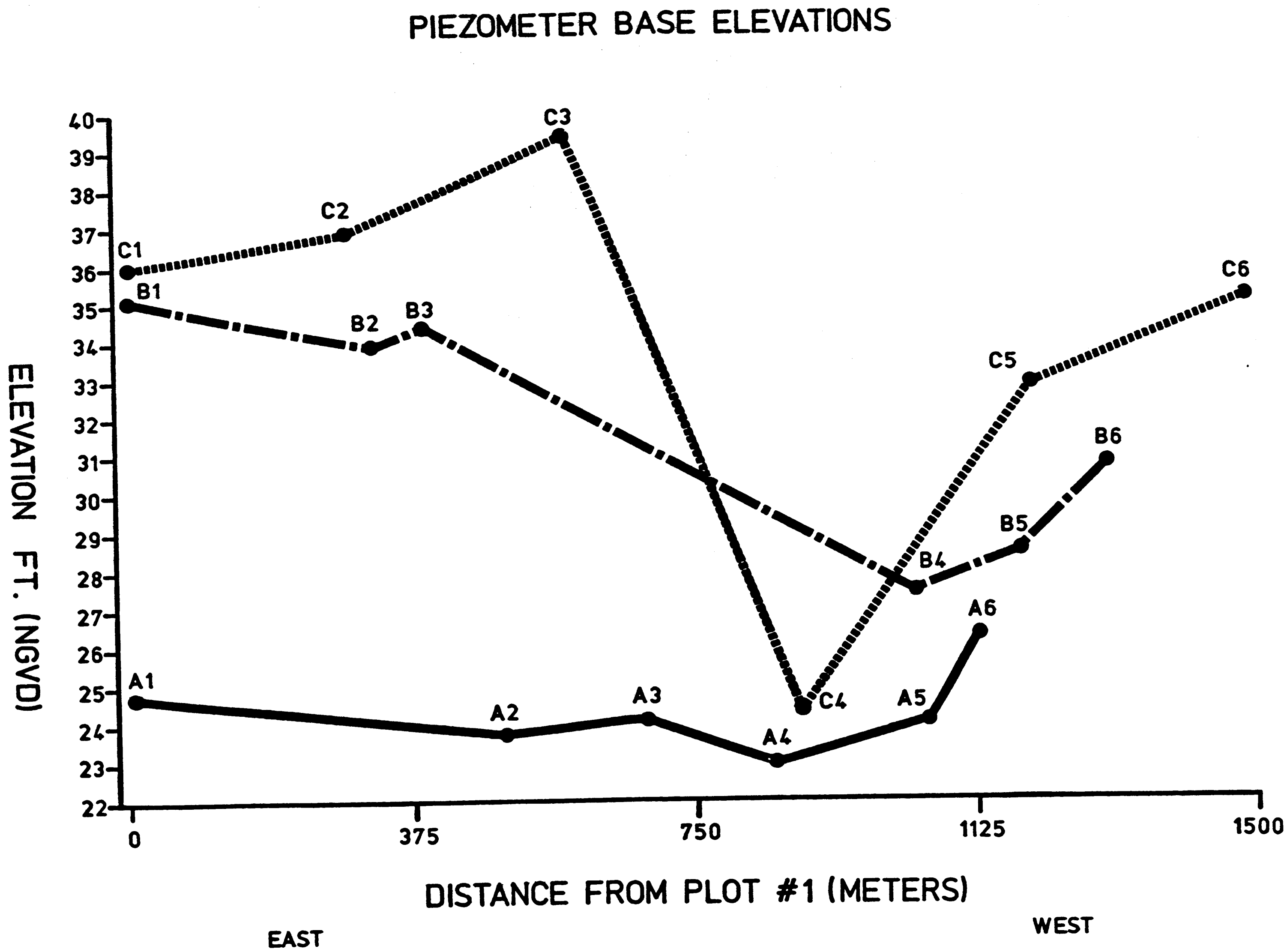


Figure 5. Piezometer base elevations and relative positions of meter-square plots along each transect.

Transect A

	Before	After
$\bar{x}$	47.4	21.5
SD	26.8	22.9

$t = 3.039 \quad p = 0.005$

Transect B

	Before	After
$\bar{x}$	48.1	20.9
SD	16.2	19.8

$t = 4.112 \quad p = 0.000$

Transect C

	Before	After
$\bar{x}$	42.6	28.2
SD	20.5	22.2

$t = 1.767 \quad p = 0.086$

Figure 6. Comparisons of percentage similarity of herbaceous vegetation on meter square plots before and after control structure installation.

transect provided ambiguous results. On Transect A, three of six plots showed a change in representation of obligate hydrophytes after the dam was installed (Figure 7). However, the proportion of obligates increased on Plot 3, and decreased on plots 5 and 6. None of the plots on Transect B changed significantly in the proportion of obligate hydrophytes (Figure 8). On Transect C, Plot 1 showed a significant decrease while Plot 4 showed an increase in the proportion of obligate hydrophytes (Figure 9).

DISCUSSION

It is evident from the data presented and our overall observations of the Cypress Creek Swamp that this restoration project achieved the major objective of restoring the hydroperiod to the system. Water levels in piezometers on transects A and B did increase after structure construction. Transect C showed piezometer water level change that approached significance (P=0.088).

The Cypress Creek Swamp is a complex system to evaluate, complicated by elevational change within and between monitored transects (see Figure 5). Not surprisingly, the results from the vegetation data were ambiguous, probably because of the complexities in this system. The percentage similarity analyses implied that changing water levels did affect the plant community. However, further analysis using the hydroperiod categories showed no pattern. Perusal of individual plant lists for each plot and monitoring event does suggest that change is occurring. For monitoring events in 1986 (immediately after structure installation) herbaceous species composition was dominated by floating aquatics. In 1987, many of those species were replaced by non-floating obligate hydrophytes. It is probable that the plant community in this system is continuing to become more hydrophytic and that our monitoring program was terminated too early to detect these changes. It was evident from our gross observations that many of the transitional wetland species such as wax myrtle (Myrica cerifera), blackberry (Rubus sp.), and laurel oaks (Quercus laurifolia) were dying and being replaced by an understory of buttonbush (Cephalanthus occidentalis) and a variety of herbaceous obligate hydrophytes. This system was still in a state of change at the end of the monitoring program.

From the perspective of the regulatory community, this study is of interest because it demonstrates the need for long-term monitoring programs to evaluate effects of water control structures that alter the hydrology of natural wetland systems. The permit required monitoring of the site for only one quarter prior to dam installation. Fortuitously, construction was delayed and we were able to collect six quarters of data prior to installation and five quarters after construction. Eleven monitoring events represent a considerable financial expense for the permittee, yet even with this amount of data collection, little could be demonstrated except change on a very gross level. Improvements to the monitoring regime would include evaluation of the overstory in addition to the understory, an increased number of study

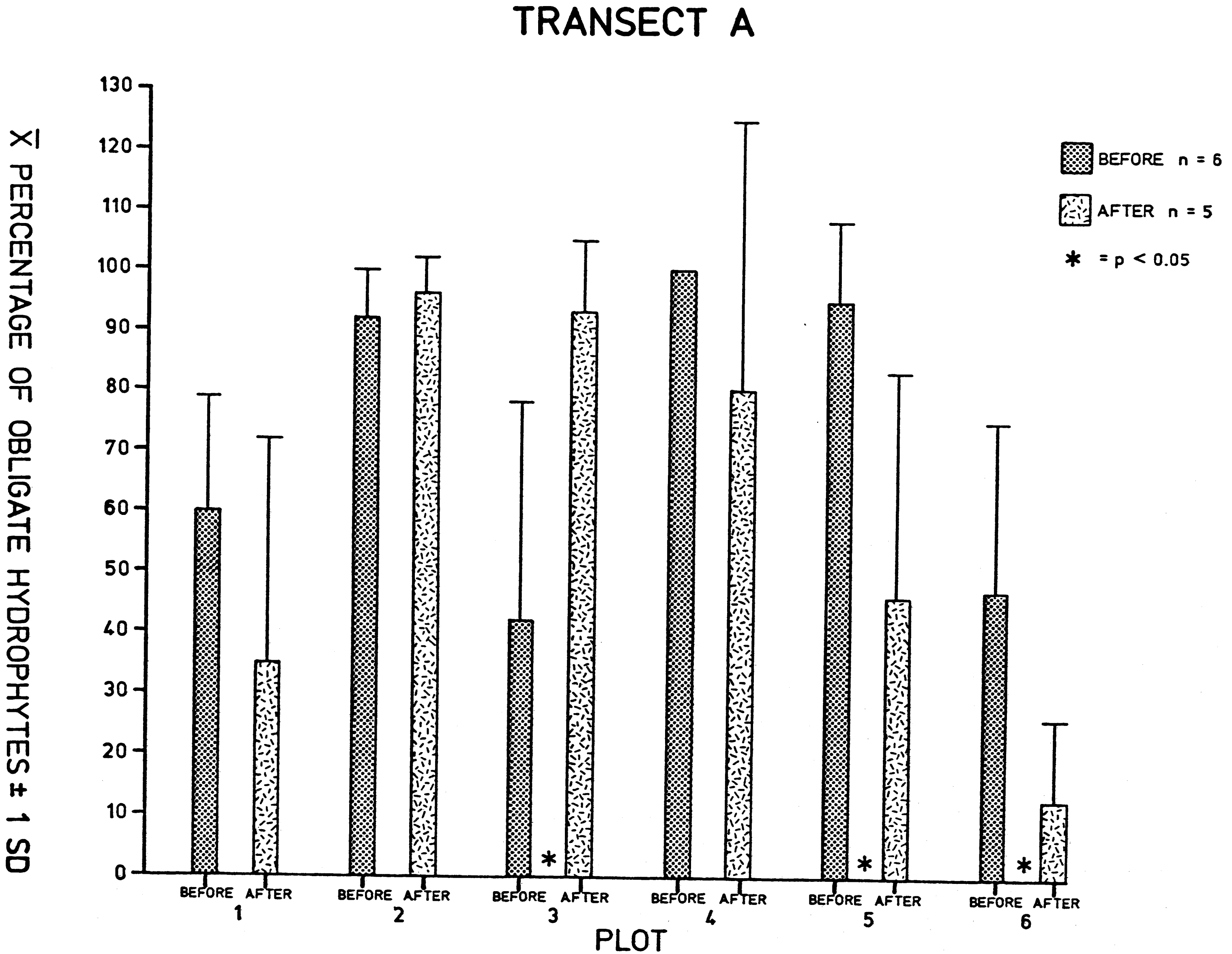


Figure 7. Comparison of obligate hydrophytes on Transect A before and after control structure installation. Line above bar indicates standard deviation.

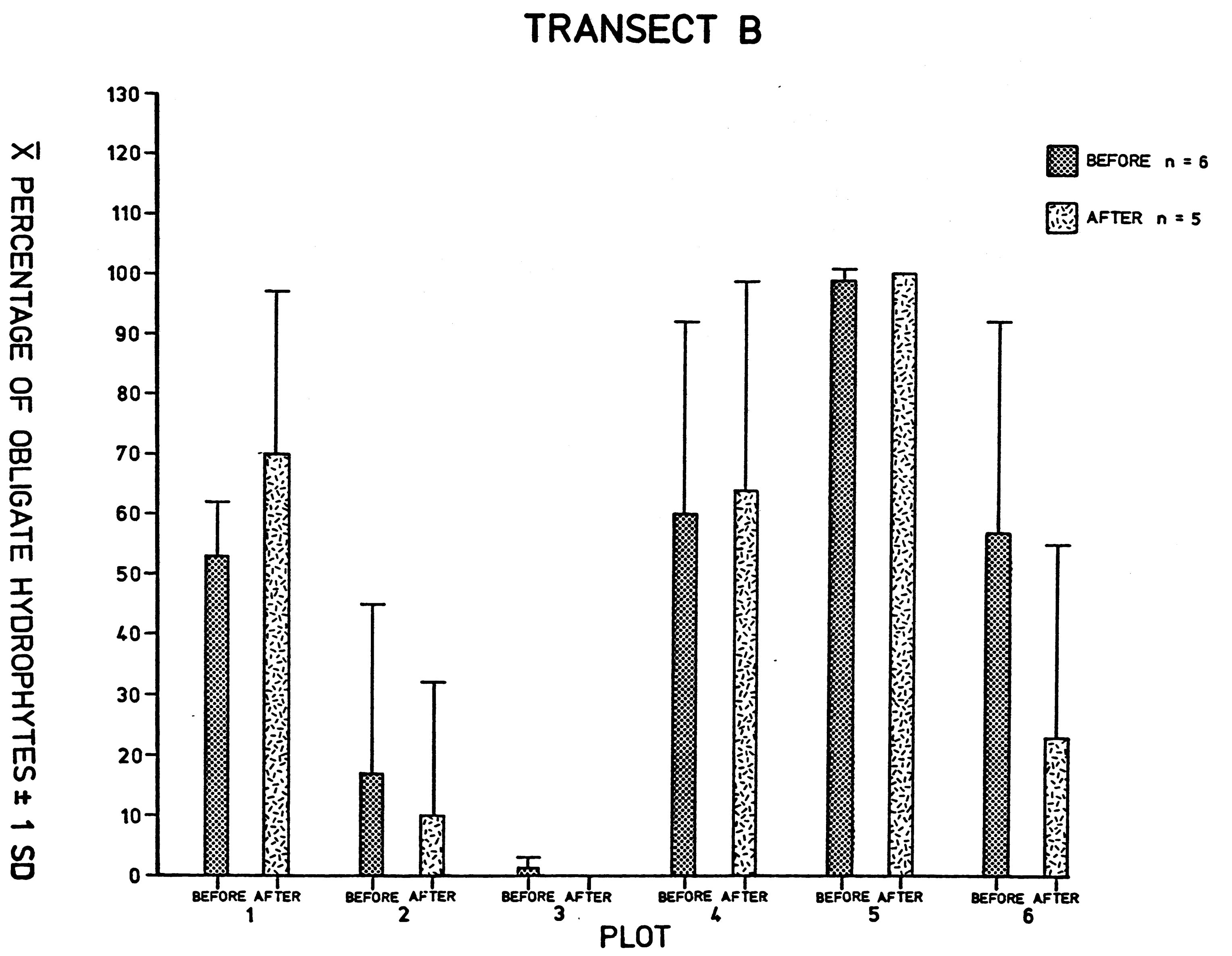


Figure 8. Comparison of obligate on Transect B before and after control structure installation. None of the comparisons were significant. Line above bar indicates standard deviation.

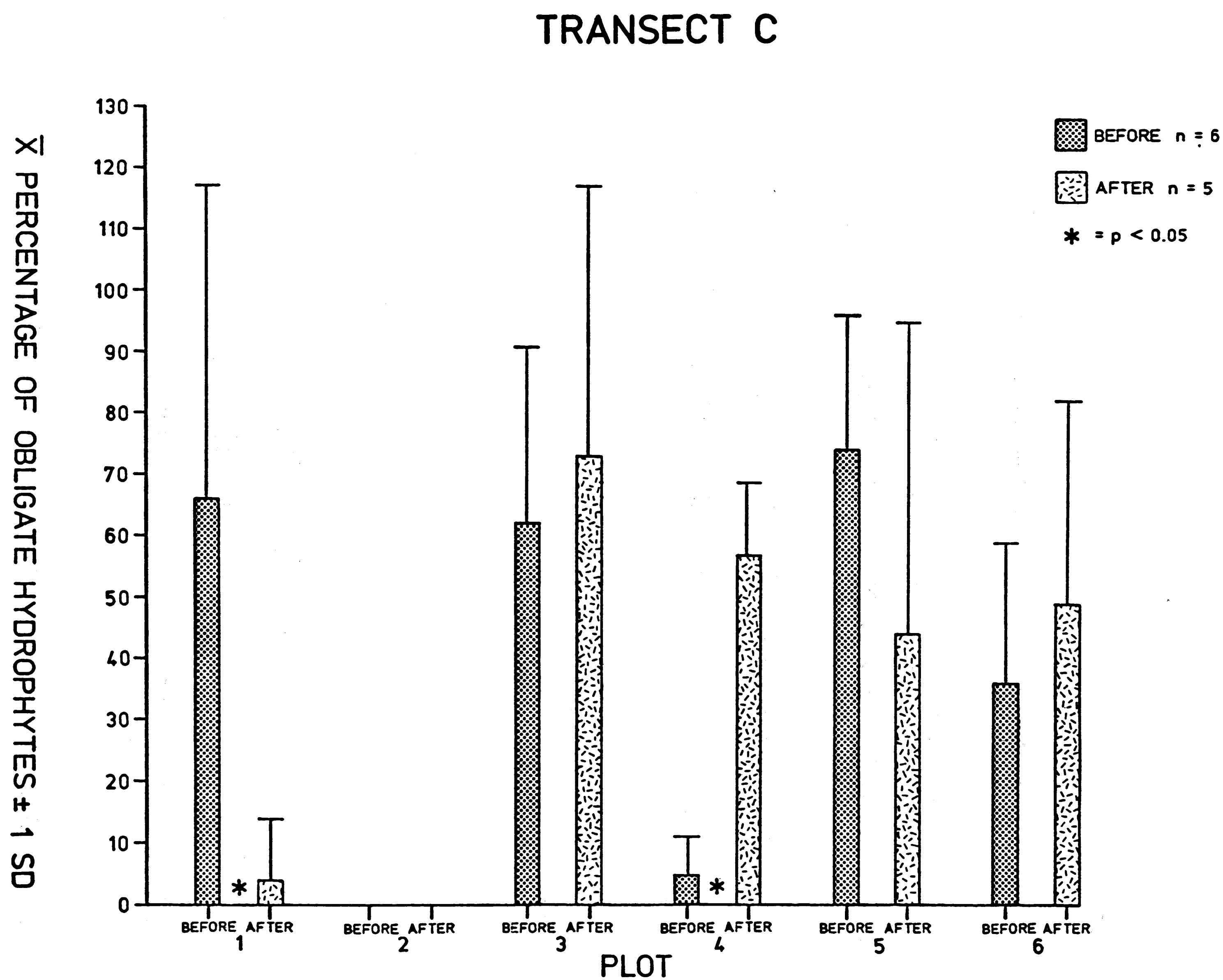


Figure 9. Comparison of obligate hydrophytes on Transect C before and after control structure installation. Line above bar indicates standard deviation.

plots, and continued monitoring of the system at least until similarity indices within plots no longer changed significantly.

LITERATURE CITED

Blake, N. M. 1980. Land into water - water into land. Tallahassee, University Presses of Florida.

Heidt and Associates, Inc. 1980. Cypress Creek Stormwater Management Plan, Tampa, Florida.

USFWS. 1986. 1986 wetland plant list: Southeast region. National Wetlands Inventory, U.S. Fish and Wildlife Service, St. Petersburg, Florida

THE COMPARISON OF SURVIVAL AND GROWTH OF PONDCYPRESS AND BALDCYPRESS IN FIVE DIFFERENT POST-MINING SOILS

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Gainesville, Florida

ABSTRACT

The growth and survival of pondcypress were compared to those of baldcypress, on five mined soils. Sites were chosen on three abandoned phosphate clay settling ponds and two limerock mines. Soils ranged from essentially pure clay to sand and gravel. Early results from paired field plots revealed negligible growth differences between species but large differences in survival did occur with baldcypress outperforming pondcypress. Large inter-site growth variations were noted for both species. Early analyses indicate a correlation of growth with soil particle size and nitrogen availability.

INTRODUCTION

Recent interest and legislation concerning the reclamation of lands disturbed by mining have produced considerable research in restoration techniques. It is generally held that one of the most effective and efficient programs of restoration involves the re-establishment of the naturally occurring, pre-mining, plant associations.

The succession of wetland areas often becomes arrested, however, at the shrub or willow stage due to a lack of seed source availability (Rushton 1983; McClanahan 1986). In order to move beyond this arrested stage and accelerate the return of pre-disturbance hardwood and conifer species, seeds or seedlings must be transferred in (Gunderson 1984; Rushton 1987). Cypress, as a wetland species, has been widely planted, however, few investigations have directly compared the two taxa, _Taxodium distichum_ and _Taxodium ascendens_. Comparisons that have been done often lack reference to edaphic factors, and/or produce conflicting results.

In a study about light intensity, baldcypress consistently outgrew pondcypress in all light treatments except 100% (Neufield 1983). In a study using soils with identical conditions, pondcypress showed a 50% greater height growth than baldcypress, but the stem diameters were roughly equal (Murphy et al. 1974).

When survival, height and growth of planted baldcypress were compared with pondcypress in several post-mining sites, no significant differences in growth or height appeared. However, survival rate of baldcypress was significantly greater than pondcypress (Rushton 1987). Both pond and baldcypress performed best when planted on sites influenced by clay. Hydrologic factors appear to play a vital role in their survival and growth (Rushton 1987; Miller 1987).

In naturally occurring situations, pondcypress generally predominate in low nutrient soils, while baldcypress are more commonly associated with higher nutrient conditions (Odum 1984; Monk 1965).

In order to successfully utilize both species as transfer plants for wetland restoration, a more complete comparison of the trees would seem necessary. The purpose of this study was to directly compare baldcypress with pondcypress on five different post-mining soils. Since mining reclamation techniques create different mixtures of sand and clay, the soils chosen cover the spectrum from pure sand to pure clay and contain varying concentrations of macro- and micronutrients. The study involves both field and greenhouse replications.

METHODS AND MATERIALS

During July 1987, 220 pond and baldcypress seedlings were planted in five post-mining sites. Sites were chosen for their variation from almost pure sand to almost pure clay as well as differing levels of nutrients. Two sites were in Alachua County and three sites were in Polk County. The Alachua sites were mined for foraminiferous limestone (Crystal River Formation) and the Polk sites were mined for phosphate. All sites had standing water and large growths of willows.

All seedlings were planted under willows to simulate natural successional processes (Brown & Montz 1986; Wharton 1977) and to provide optimal light conditions (Neufield 1983). Seedlings were planted on 1 meter centers in paired 10 meter lines. When ground elevations and water levels were relatively homogeneous, each column was planted with a single species. When conditions did not favor single species lines, seedlings were alternated to balance exposure to water levels (see diagrams).

X X X X X X	X 0 X 0 X 0 X 0 X
single species lines	alternate species lines
0 0 0 0 0 0 0	0 X 0 X 0 X 0 X 0

X = Bald 0 = Pond

Two replicate paired plots of 22 trees each were established at each site. The trees, all greenhouse-grown tubelings, were planted using a KBC planting bar. Water depth and tree height (ground surface to apical bud) were measured at planting.

Soil samples were taken from each site using a mud auger and placed in plastic bags for later analysis. A portion of each sample was analyzed (Wallace Lab, University of Florida) for macro- and micronutrients. Other portions were used (in the Center for Wetlands Lab) for determining sand, silt, and clay content using the hydrometer method (Klute 1986) and sieving. Several liters of soil were also removed from each site for use in greenhouse replication studies.

Particle sizes varied widely from site to site. Clay percentage ranged from a low of 0% at IMC-H9 to a high of 71.7% at Tenoroc. Sand percentages showed a reverse trend and went from a high of 100% at IMC-H9 to a low of 3.4% at Tenoroc (Figure 5).

With the exception of Tenoroc, all of the sites demonstrate a basic pH (Table 1). While all of the sites have large quantities of calcium, the Polk County sites have significantly larger amounts of phosphorus, magnesium, and aluminum due to the fact that they are phosphate rather than limerock mines. Nitrogen as NH4 and NO3 were highest at Hollingsworth, Hashknife, and Tenoroc.

Fifty seedlings were planted in 20 centimeter diameter pots in the CFW greenhouse for more controlled study. Five pond and 5 baldcypress seedlings were planted in material removed from each field site. The pots were arranged in a random block design (Little & Hills 1978). Measurements were made of each seedling's height above soil surface at planting.

SITE DESCRIPTIONS

The Hollingsworth site (Figure 1) is in an abandoned limerock mine near High Springs. The area was abandoned in 1967, however active mining is taking place on adjacent land. All of the properties are owned by E. V. Hollingsworth. Transects are located along the edges of water filled mining cuts. The soil is a thin layer of overburden on limestone and is periodically flooded.

Hashknife (Figure 2) is the oldest site. It is an abandoned limerock mine, west of Gainesville, off Highway 241 North. The mine was last active in 1948 according to the Buchanan families, the landowners. Trees were planted along the banks of shallow ponds lined with willows.

Tenoroc (Figure 3), in Polk County, is an old clay settling pond that is currently part of a State Reserve maintained by the Department of Natural Resources. The area was abandoned in 1972 and is surrounded by spoil piles. The transects were located in an intermittently flooded area. The eastern plot (I) was wetter, being on the edge of a small pond, while the western plot (II) was noticeably drier with no standing water nearby.

IMC-H9 and IMC-H9-A (Figure 4) are part of a reclamation project by the International Mineral and Chemical Corp. Both sites are

Table 1. Soil nutrient analysis.

SITE NAME	PPM Soil*									Concentration, mg N/L**	
	pH	P	K	Ca	Mq	Al	Zn	Cu	Mn	NH4-N	NO3-N
Hollingsworth	8.5	1	8	2000	56	4	<1	<1	<1	4.377	4.241
Hashknife	8.3	5	12	2000	72	4	<1	<1	1	5.882	4.970
IMC-H9	8.6	200	4	2000	72	60	3	<1	3	2.189	2.462
IMC-H9-A	7.9	200	8	2000	228	144	2	<1	7	3.055	2.417
Tenoroc	6.2	200	28	2000	522	-	-	-	-	10.382	3.648

* Soil analysis by IFAS Extension Soil Testing Laboratory, University of Florida
**Soil analysis by Department of Soil Science, University of Florida

Hollingsworth Alachua County

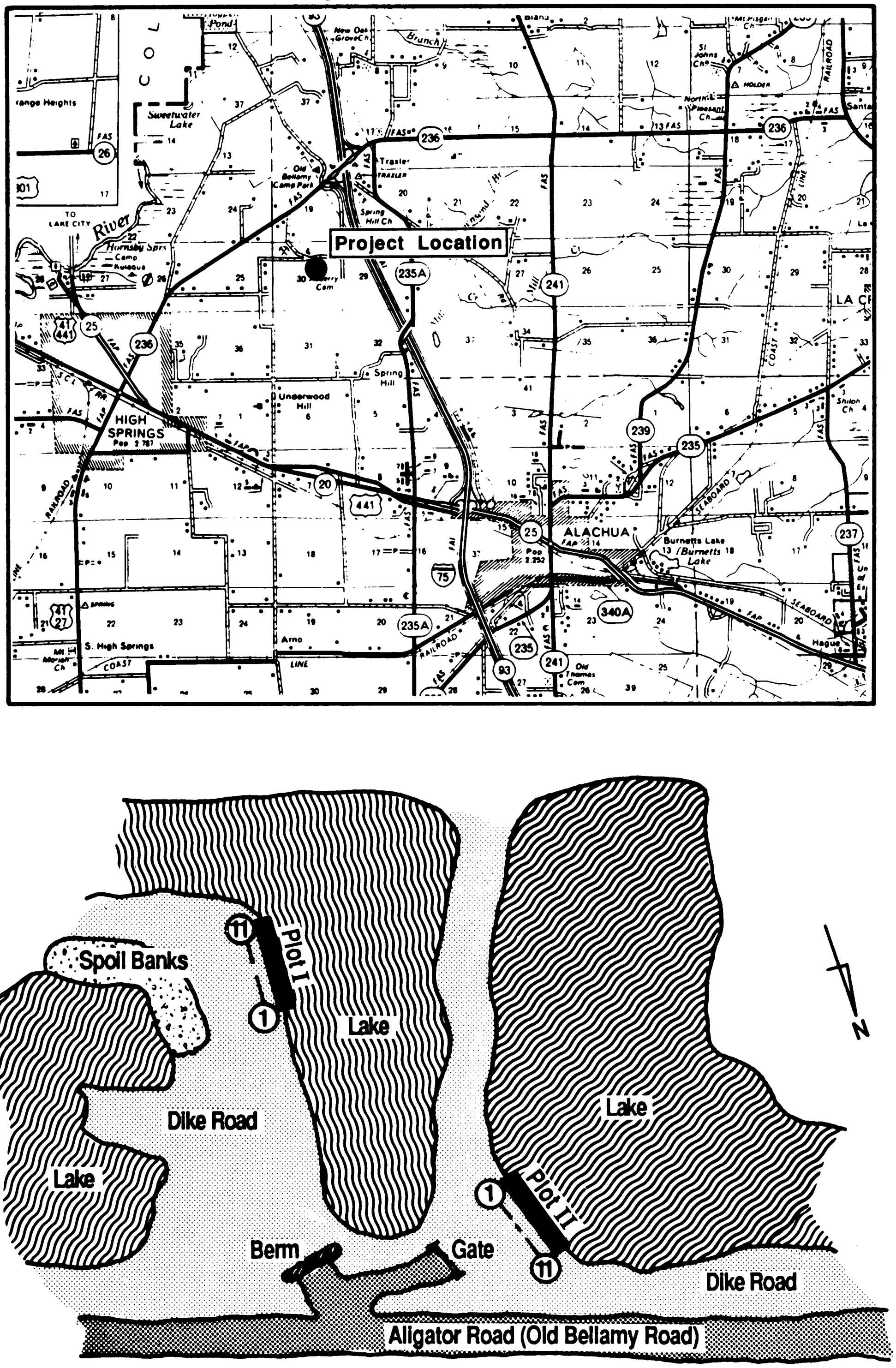
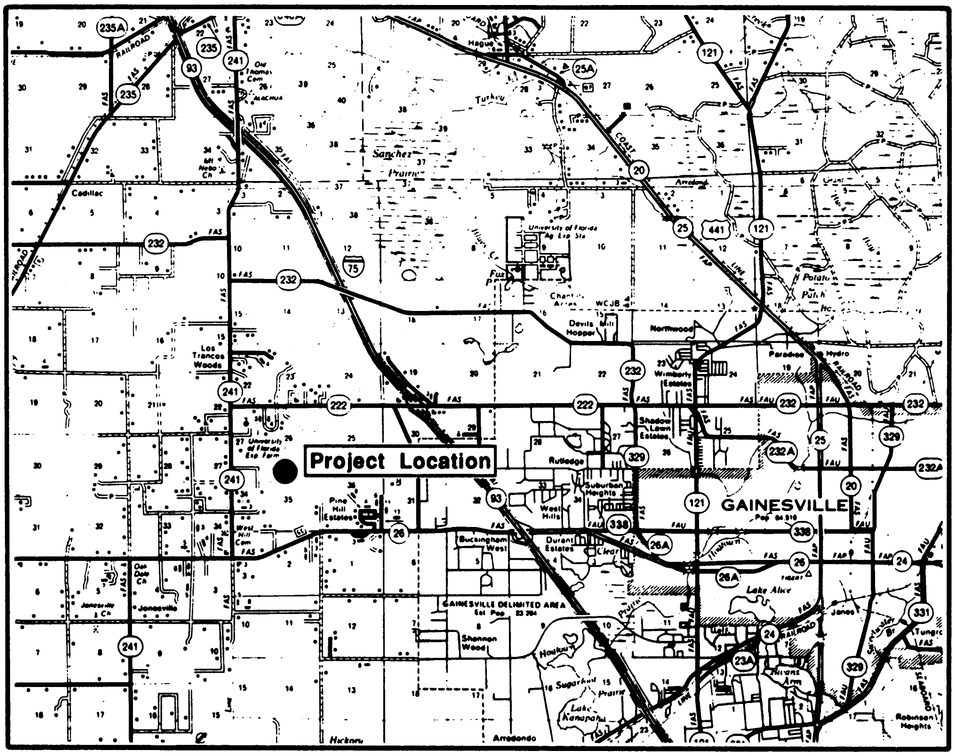


Figure 1.

Hashknife Alachua County



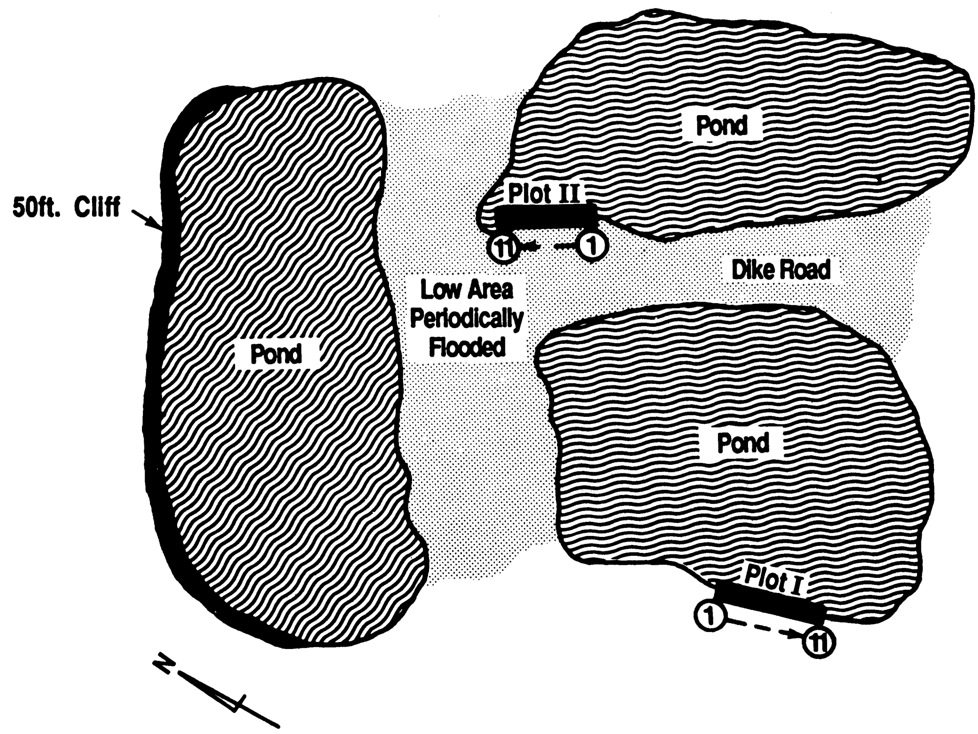


Figure 2.

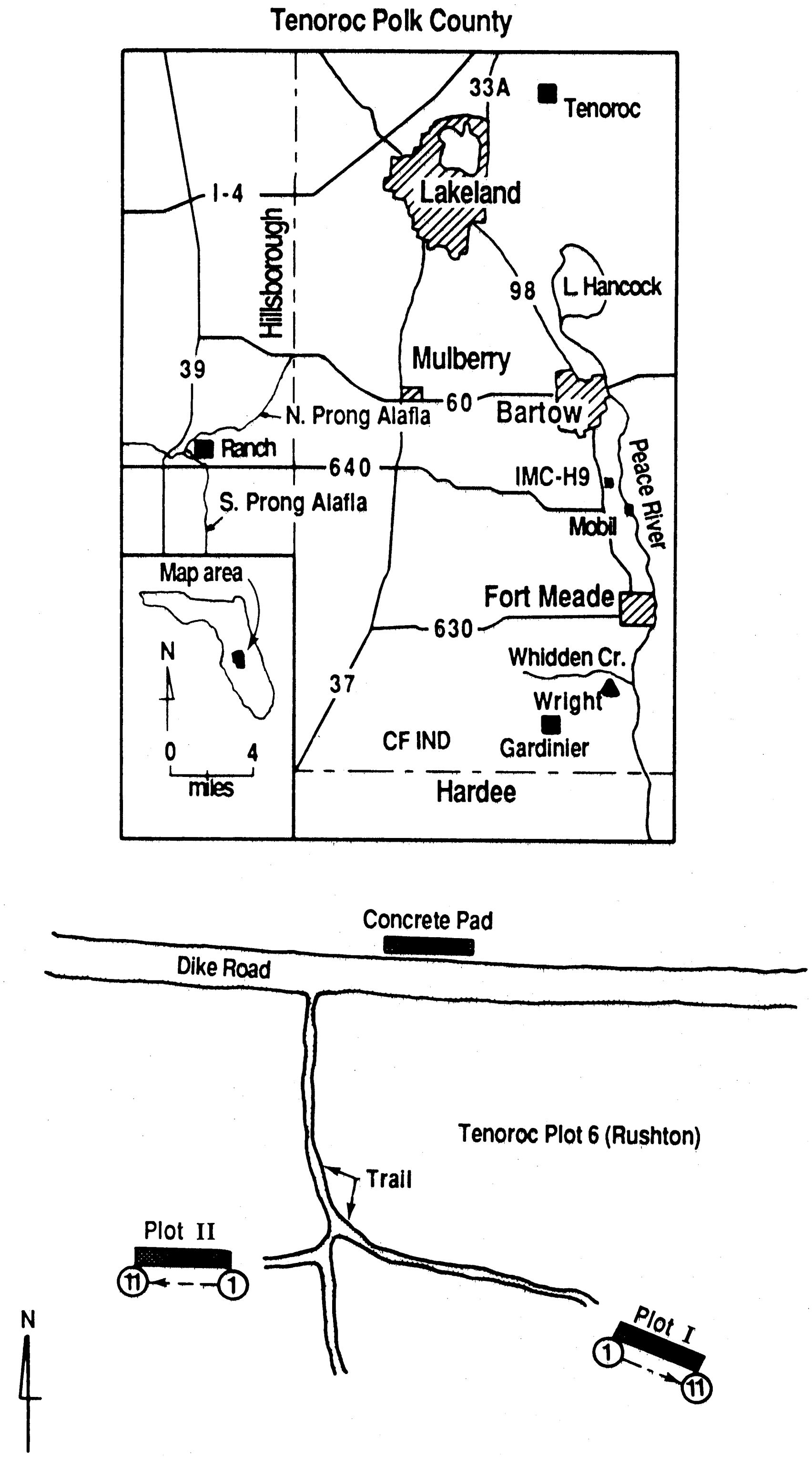
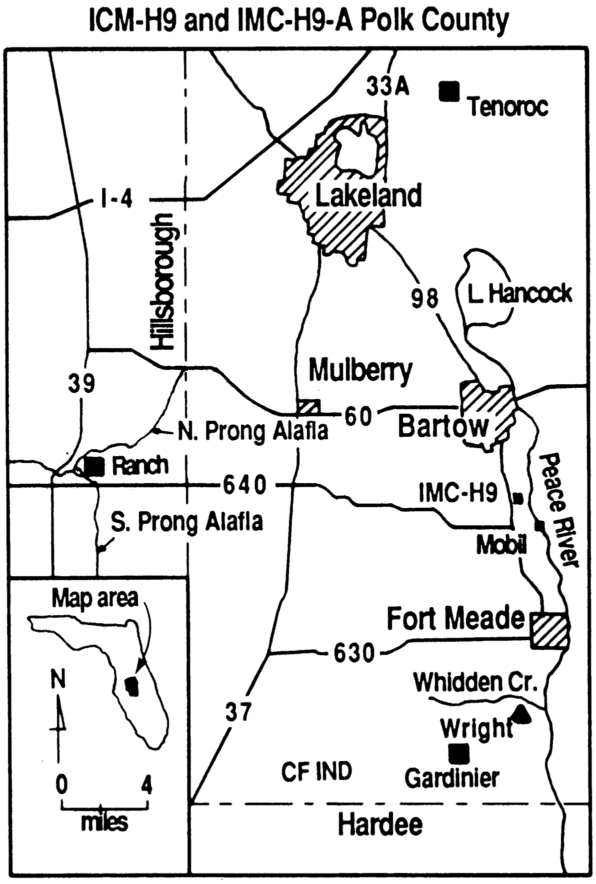

Tenoroc Polk County
33A
Tenoroc
I-4
Lakeland
Hillsborough
98
L. Hancock
Mulberry
39
60
Bartow
N. Prong Alafla
Ranch
640
IMC-H9
Peace River
S. Prong Alafla
Mobil
Map area
Fort Meade
630
N
Whidden Cr.
37
Wright
0
4
miles
CF IND
Gardinier
Hardee
Concrete Pad
Dike Road
Tenoroc Plot 6 (Rushton)
Trail
Plot II
11
1
Plot I
1
11
N

Figure 3.


ICM-H9 and IMC-H9-A Polk County
33A
Tenoroc
Lakeland
I-4
Hillsborough
98
L Hancock
Mulberry
39
60
Bartow
N. Prong Alafla
Ranch
640
IMC-H9
Peace River
S. Prong Alafla
Mobil
Map area
Fort Meade
N
630
Whidden Cr.
37
Wright
0
4
miles
CF IND
Gardinier
Hardee

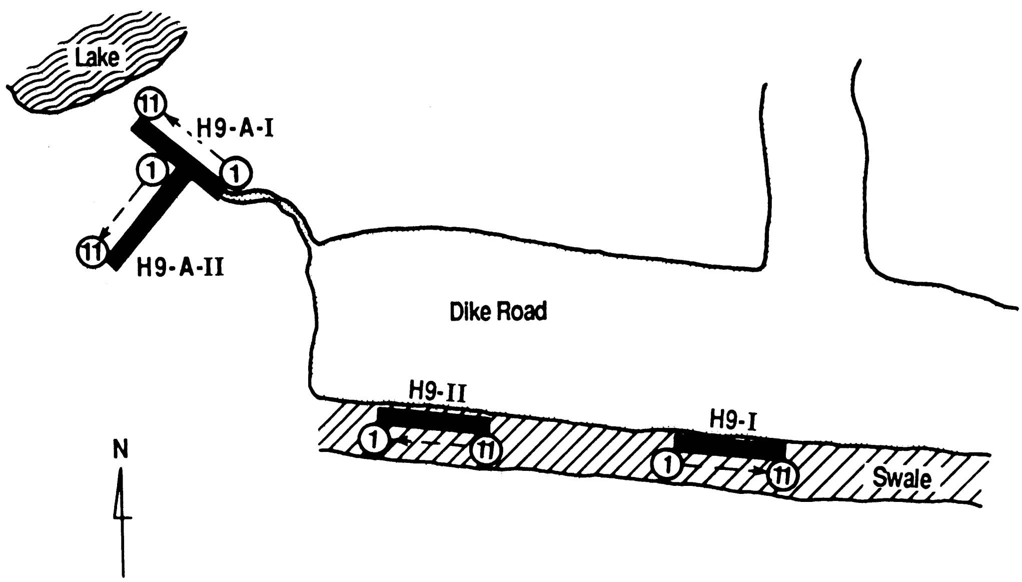

Lake
11
H9-A-I
1
1
11
H9-A-II
Dike Road
H9-II
H9-I
N
1
11
1
11
Swale

Figure 4.

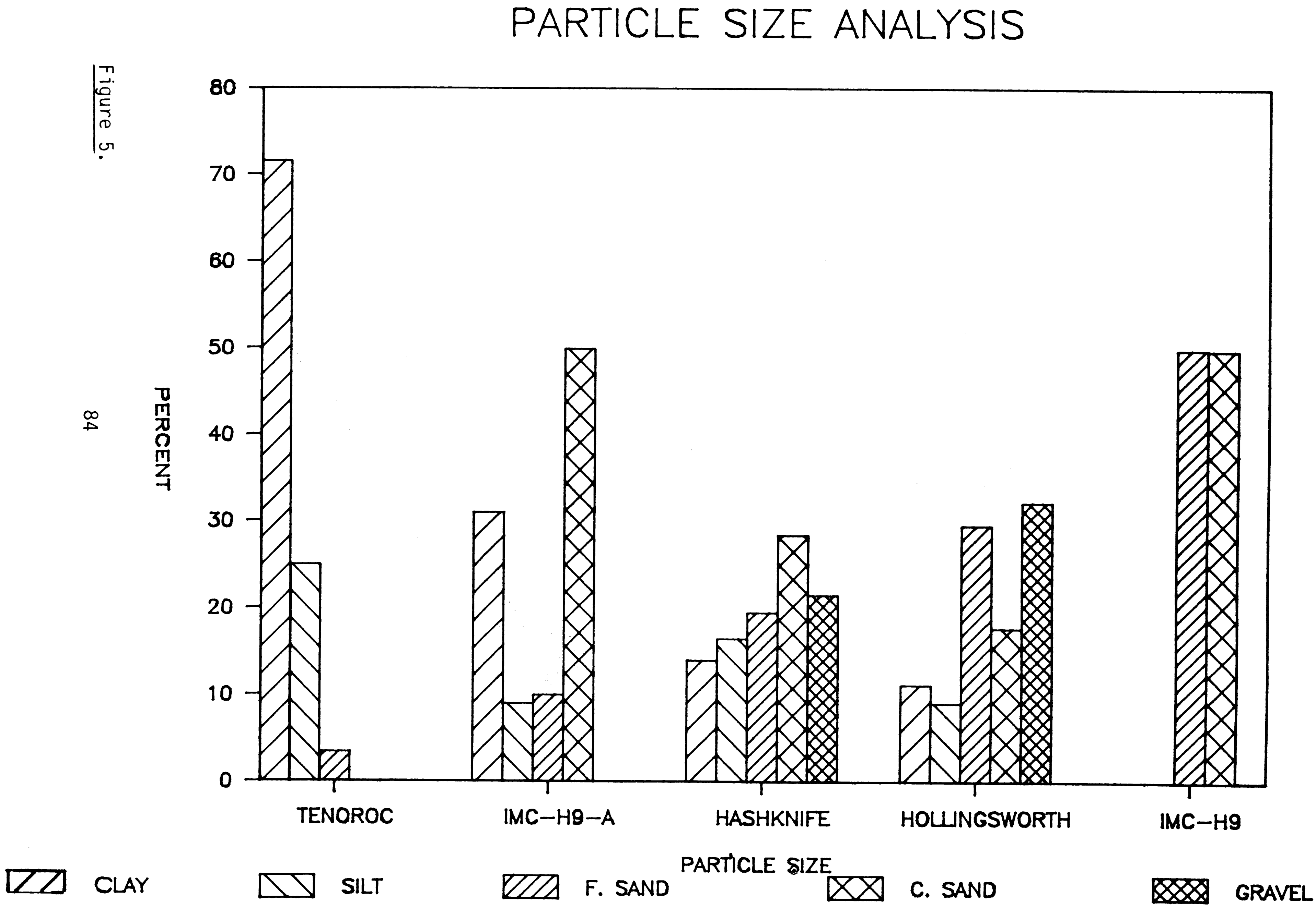

PARTICLE SIZE ANALYSIS
PERCENT
0
10
20
30
40
50
60
70
80
TENOROC
IMC-H9-A
HASHKNIFE
HOLLINGSWORTH
IMC-H9
PARTICLE SIZE
CLAY
SILT
F. SAND
C. SAND
GRAVEL

Figure 5.

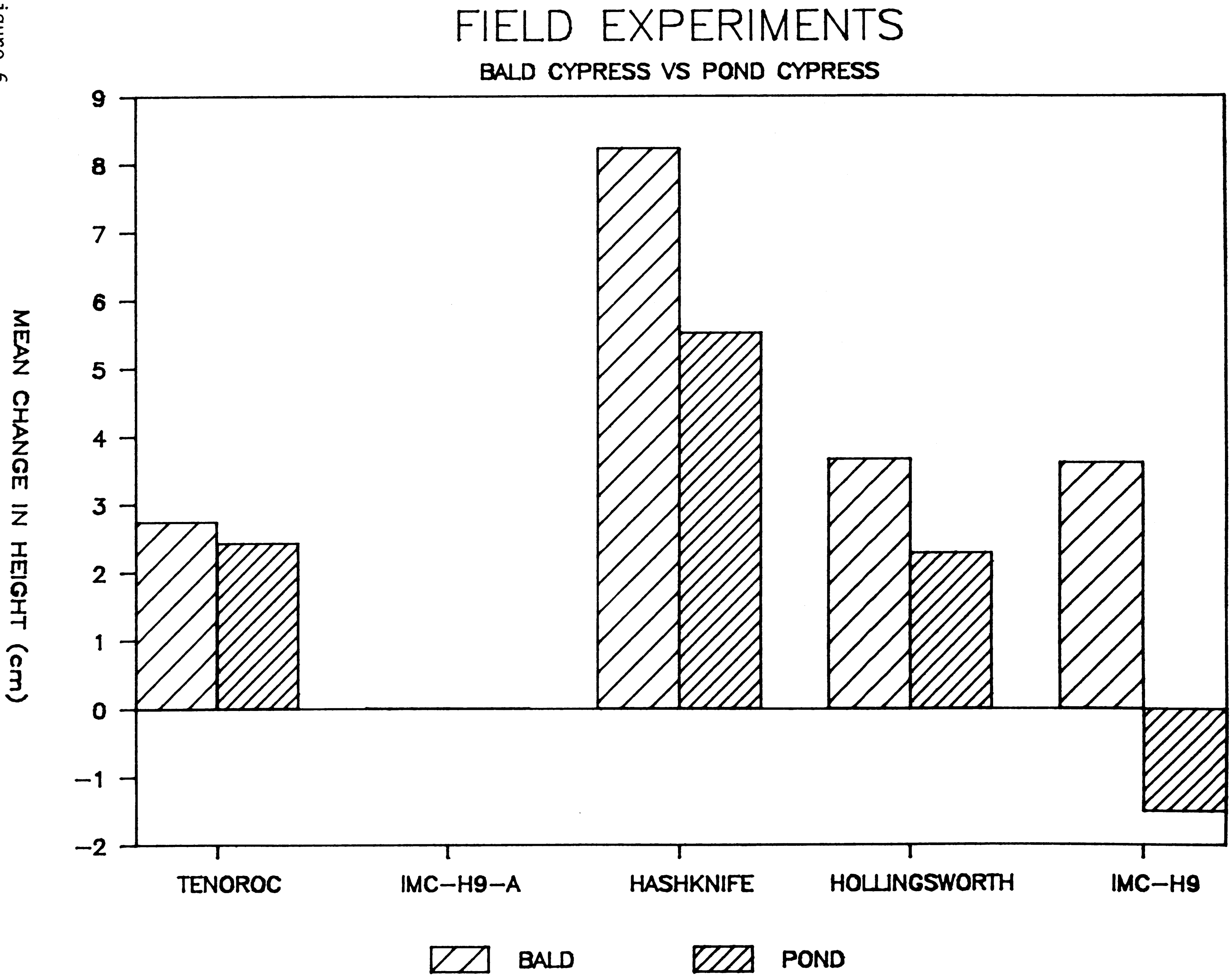

FIELD EXPERIMENTS
BALD CYPRESS VS POND CYPRESS
MEAN CHANGE IN HEIGHT (cm)
9
8
7
6
5
4
3
2
1
0
−1
−2
TENOROC
IMC−H9−A
HASHKNIFE
HOLLINGSWORTH
IMC−H9
BALD
POND

Figure 6.

adjacent and are a clay settling area abandoned in 1970 and capped with sand in 1979. The plots of IMC-H9 were planted in a wet swale, in soil that was essentially 100% sand. The IMC-H9-A plots were planted along the edge of a lake in flooded soil that was mixed sand and clay.

RESULTS

Field Plots

The 9 month height growth differences within each field plot were subjected to a paired-plot t test at a 0.05 level of significance. There was no significant difference in the growth performance of baldcypress when compared to pondcypress (Figure 6).

A comparison of sites did yield significant results. Both species performed much better at Hashknife than at any other plot. Tenoroc's poor growth results were, in large measure, influenced by rabbit grazing and actually approached Hashknife results when grazed plants are discounted.

A significant difference occurred in the survival of species. Baldcypress showed a 92% overall survival, whereas pondcypress managed only 76%. This variation was apparently due to the poor accommodation of pondcypress to inundation stress. Survival differences were particularly noticeable at IMC plots where water levels rose following planting. IMC-H9-A was essentially drowned (Figure 7).

	Greenhouse	Tenoroc	Hashknife	Hollingsworth	IMC-H9	IMC-H9-A
Bald	100%	91%	95%	82%	96%	18%
Pond	100%	82%	86%	78%	64%	0%

Figure 7. Survival.

Greenhouse Plots

Statistical analyses showed significant growth differences between species at Hashknife and IMC-H9 with baldcypress outperforming pondcypress (Figure 8). Major inter-site differences also occurred. Hashknife soil, once again, produced the greatest growth, almost doubling the height changes found elsewhere. Greenhouse survival for all plants was 100%.

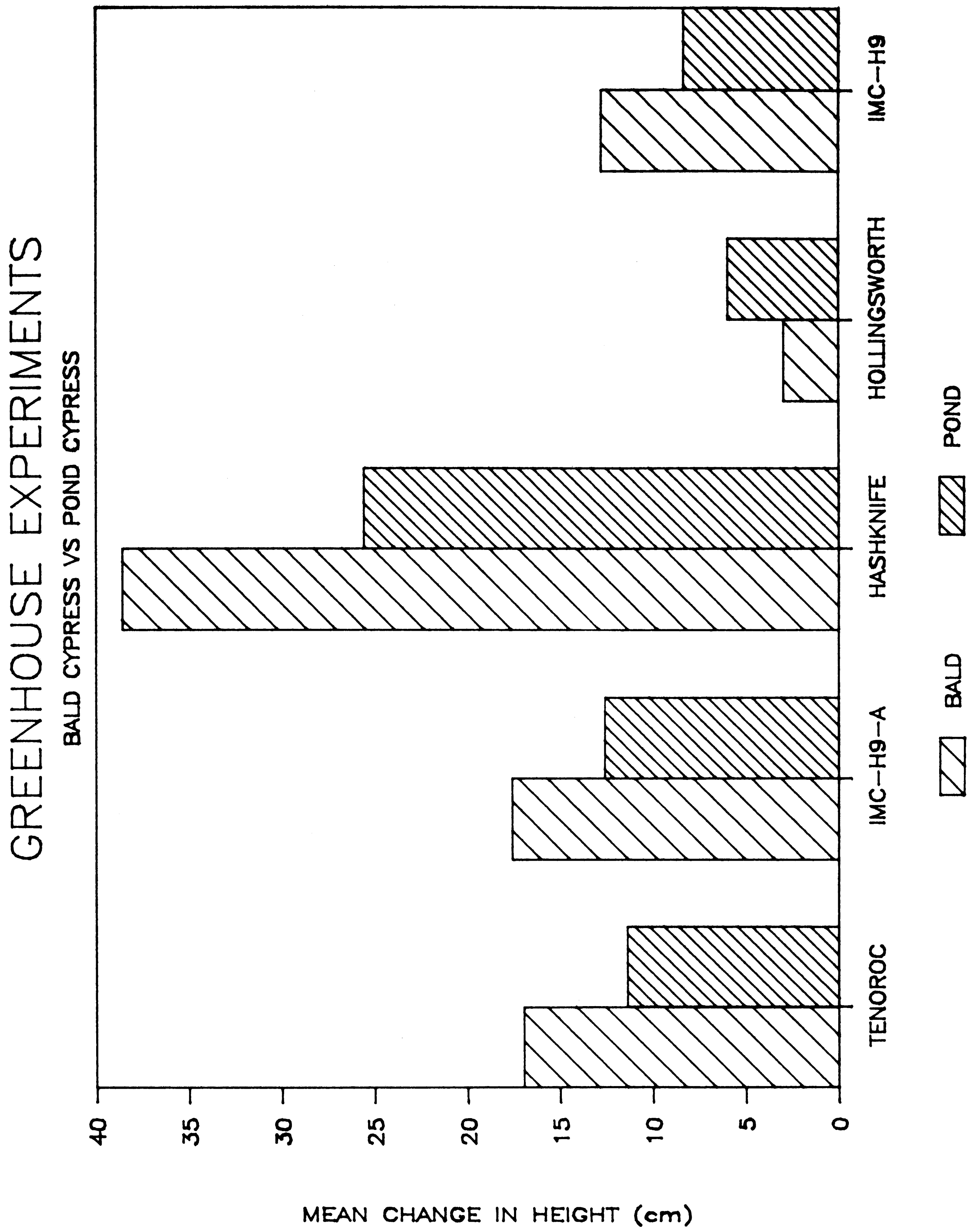

GREENHOUSE EXPERIMENTS
BALD CYPRESS VS POND CYPRESS
MEAN CHANGE IN HEIGHT (cm)
0
5
10
15
20
25
30
35
40
TENOROC
IMC–H9–A
HASHKNIFE
HOLLINGSWORTH
IMC–H9
BALD
POND

Figure 8.

DISCUSSION

Any results from this project should be viewed with caution as this reports only a 9 month grow out period for the seedlings.

Survival and change in height results suggest that both pond and baldcypress seedlings do best in soil that is a mixture of sand and clay. The maximum growth occurred at Hashknife, where the sand to clay ratio was approximately 3.4:1. This is in agreement with previous studies (Ravina & Magier 1984; Peterson 1945) showing that clay soils influenced by medium to high levels of sand and coarse fragments were much better for tree growth than either pure clay or pure sand. The mixture provides better water conductivity as well as better aeration and resistance to compaction. Hashknife soil contained large quantities of gravel and chert fragments and had the most even size distribution.

Another, and perhaps more important fact, leads to the idea that growth differences could also be attributed to nutrition. Field results were replicated in greenhouse studies, where such factors as temperature, moisture, and shading were controlled. A comparison of growth with nutrient levels showed nitrogen to be an apparent controlling agent.

A problem, inherent in the experimental design, emerged during the project and warrants discussion. The use of height growth as the major criterion for seedling success now appears to be inadequate. Baldcypress and pondcypress were observed to be using two different initial growth strategies. After 90 days of growth, pondcypress demonstrated a predilection for vertical growth while baldcypress expended its energy in increasing foliation around the apical bud and increased basal diameter. At 9 months of growth, the height differences diminished and baldcypress appeared to be the height leader.

Perhaps, with increased grow out time, differences due to growth strategies will become less significant and differences between the taxa due to site variables will become more evident.

Subsequent research on this project should include a greater emphasis on differences in nutrient uptake, survival, and total biomass, as well as height measures.

LITERATURE CITED

Brown, Clair A., and N. Montz. 1986. Cypress--The tree unique, the wood eternal. Clairtor's Publishing Division, Baton Rouge, Louisiana.

Gunderson, Lance H. 1984. Regeneration of cypress logged and burned strands of Corkscrew Swamp Sanctuary, Florida. In Katherine Ewel and H. T. Odum (Eds.), Cypress Swamps. University of Florida Press, Gainesville, Florida.

Klute, Arnold (Ed.). 1986. Methods of soil analysis part 1, physical and mineralogical methods. American Society of Agronomy, Inc. and Soil Science Society of America, Inc., Madison, Wisconsin.

Little, Thomas M., and F. Jackson Hills. 1978. Agriculture experimentation-design and analysis. John Wiley and Sons, New York, New York.

McClanahan, Timothy R. 1986. The effect of a seed source on primary succession in a forest ecosystem. Center for Wetlands, University of Florida, Gainesville, Florida.

Miller, Michael A. 1987. Groundwater control of wetlands restoration in reclamation of clay settling areas. Accelerating natural processes for wetland restoration after mining. Center for Wetlands, University of Florida, Gainesville, Florida.

Monk, Carl D., and Timothy W. Brown. 1965. Ecological considerations of cypress heads in North Central Florida. In The American Midland Naturalist 74(1):126-140.

Murphy, Brad J., Robert G. Stanley, and Charles A. Hollis, III. 1974. Seedling studies in cypress wetlands for water management, recycling and conservation. Annual Report on Research Project to National Science Foundation (GI 38721) and The Rockefeller Foundation (RF 783029).

Neufield, Howard S. 1983. Effects of light on growth, morphology, and photosynthesis in baldcypress (Taxodium distichum (L.) rich) and pondcypress (T. ascendens Brongn.) seedlings. Bulletin of the Torrey Botanical Club, Jan.-Mar. 1983, Vol. 110:1, pp. 43-54.

Odum, H. T. 1984. Summary: Cypress swamps and their regional role. In Katherine Ewel and H. T. Odum (Eds.), Cypress Swamps. University Presses, Gainesville, Florida.

Peterson, J. D. 1945. The role of clay minerals in the foundation of soil structure. Journal Paper No. J. 1309 of the Iowa Agricultural Experiment Station, Soils Subsection, Project No. 583.

Ravina, I., and J. Magier. 1984. Hydraulic conductivity and water retention of clay soils containing coarse fragments. In Soil Science Society American Journal 48:736-740.

Rushton, Betty T. 1983. Examples of natural wetland succession as a reclamation alternative. In D. J. Robertson (Ed.), Symposium on Reclamation and the Phosphate Industry (pp. 148-89). Florida Institute of Phosphate Research, Bartow, Florida.

Rushton, Betty T. 1987. Wetland reclamation by accelerating succession. Ph.D. Dissertation, 266 pp. University of Florida, Gainesville, Florida.

Wharton, C. H., H. T. Odum, K. Ewel, M. Duever, A. Lugo, R. Boyt, J. Bartholomew, E. DeBellevue, S. Brown, M. T. Brown, and L. Duever. 1977. Forested wetlands of Florida: Their management and use. Final Report to Division of State Planning, 348 pp. Center for Wetlands, University of Florida, Gainesville, Florida.

Zar, Jerrold H. 1974. Biostatistical analysis. Prentice-Hall, Inc., Edgewood Cliffs, New Jersey.

COASTAL VEGETATION PROGRAM OF THE LOUISIANA DEPARTMENT OF NATURAL RESOURCES

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ABSTRACT

Louisiana has 40% of the coastal marsh in the contiguous United States, and an annual wetland loss rate of about 100 km^2. To combat the loss of this important resource the Louisiana Legislature created the Coastal Environment Protection Trust Fund in 1981. The Coastal Vegetation Program was developed in 1986 as part of a comprehensive Coastal Protection Master Plan implemented by these funds. The goals of this program are to conduct applied research on vegetative wetland restoration techniques, to facilitate vegetative planting projects designed for reducing wetland loss, and to monitor these and other restoration projects that may affect coastal plant communities. The project purpose, design, locations, results of vegetative wetland restoration projects (when available), and findings from other wetland restoration monitoring will be discussed.

INTRODUCTION

Louisiana is losing 100 km^2 (25,000 acres) of coastal wetlands per year (Gagliano et al. 1981). The causes of this problem have been the subject of much study and discussion, and include sediment deprivation, canal dredging, subsidence, saltwater intrusion, development, and wave erosion (Craig et al. 1979; de Mond et al. 1986; Gagliano et al. 1981; Sasser et al. 1986; Scaife et al. 1983; Wells & Coleman 1987). The economic value per 0.4 ha (acre) of marsh has been approximated at $1,300-$4,000 based on the estimated value of commercial fisheries, commercial wildlife, recreation, reduced hurricane surges, and real estate (U.S. Army Corps of Engineers 1987). Louisiana's annual loss thus amounts to between $44 and $100 million.

In response to this problem, in its 1981 special session the Louisiana Legislature appropriated $35 million to the now-defunct Coastal Environment Protection Trust Fund (Act 41). Currently, the state's coastal restoration efforts are funded on a yearly appropriation basis. A 10-year "Coastal Protection Master Plan" was developed by the Coastal Protection Section (CPS) of the Louisiana Department of Natural Resources (DNR) and approved by the legislature in 1985. The Coastal Vegetation Program is an outgrowth of this plan and was initiated in August 1986. This paper briefly describes the wetland restoration efforts that the CVP has been involved in. The CVP's work focuses on 1) designing and implementing wetland restoration and erosion control projects that utilize vegetation as a main component, and 2) monitoring these and other wetland restoration projects sponsored by DNR that indirectly influence wetland vegetation.

APPLIED RESEARCH PROJECTS

Erosion Control in High Wave-Energy Environments

Shoreline vegetation is known to absorb wave energy, and thus to reduce erosion rates; however, in areas where stabilization is most needed, high wave-energy environments, it is difficult to establish plants--seeds will wash away and transplants tend to wash out. A test was conducted of the effectiveness of a relatively low-cost vegetative erosion control system, similar to that described by Allen et al. (1986), at four locations representing different wave-energy intensities. In order of decreasing intensity, these areas were:

1) the Gulf shore of Rockefeller Wildlife Management Area, unlimited fetch (Figure 1, #8).

2) Turtle Cove, Manchac Wildlife Management Area, SE-NW fetch across Lake Pontchartrain of approximately 42 km (Figure 1, #14).

3) Northwest shore of Lake Salvador, Salvador Wildlife Management Area, SE-NW fetch of approximately 11 km (Figure 1, #18).

4) North-west shore of Lake Cataouatche, Salvador Wildlife Management Area. SE-NW fetch of approximately 6 km (Figure 1, #16).

Two rows of 50 smooth cordgrass (*S. alterniflora*) plants were planted through fiber mats (*see* Allen et al. 1986). The mats were 14.6 m (48 ft) long, 1.4 m (5 ft) wide, and oriented parallel to the shore. The mats ("Paratex," 5-cm thick rubberized hair and fiber) were rolled out onto the site, then a welded wire fabric with a mesh size of 15.2 cm x 15.2 cm (6 in. x 6 in.) was rolled out on top of the mat. This was held in place with 1.8-m (6-ft) "staples" made of 0.95-cm (3/8-in.) concrete reinforcement rods laid out in three rows at 0.9-m (3-ft) intervals (51 rods) (Figure 2).

A plastic fence 0.6 m (2 ft) high and 15 m (50 ft) long was placed 1.5 m (5 ft) seaward of each mat to protect it from direct wave impact. The fence material was a high-strength plastic with apertures 10 cm x 1.5 cm and attached to 0.95-cm (3/8 in.), 3.7-m (12-ft) long concrete reinforcement rods with a 15 cm (6 in.) hook in the end to prevent the fence from slipping off the rods. The fence material was folded over into a 0.6-m (2-ft) width, woven through, and attached at the top of the rods with a ratchet-type plastic cable tie. Five replicates of these plots were installed in the intertidal zone at each of the four project sites. These plots were completed on 17 June 1987.

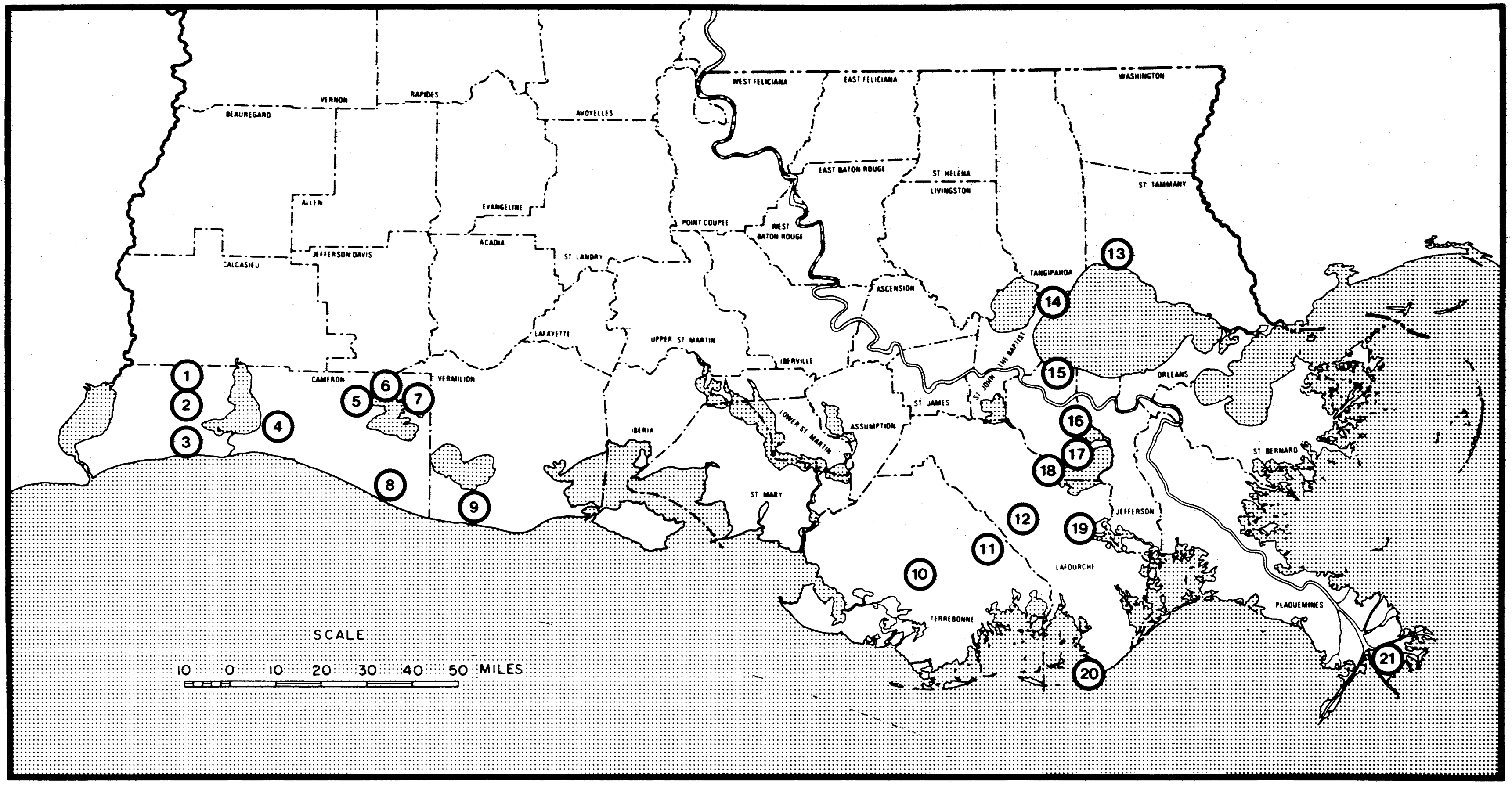

Figure 1. Map of coastal Louisiana showing location of projects described in text.

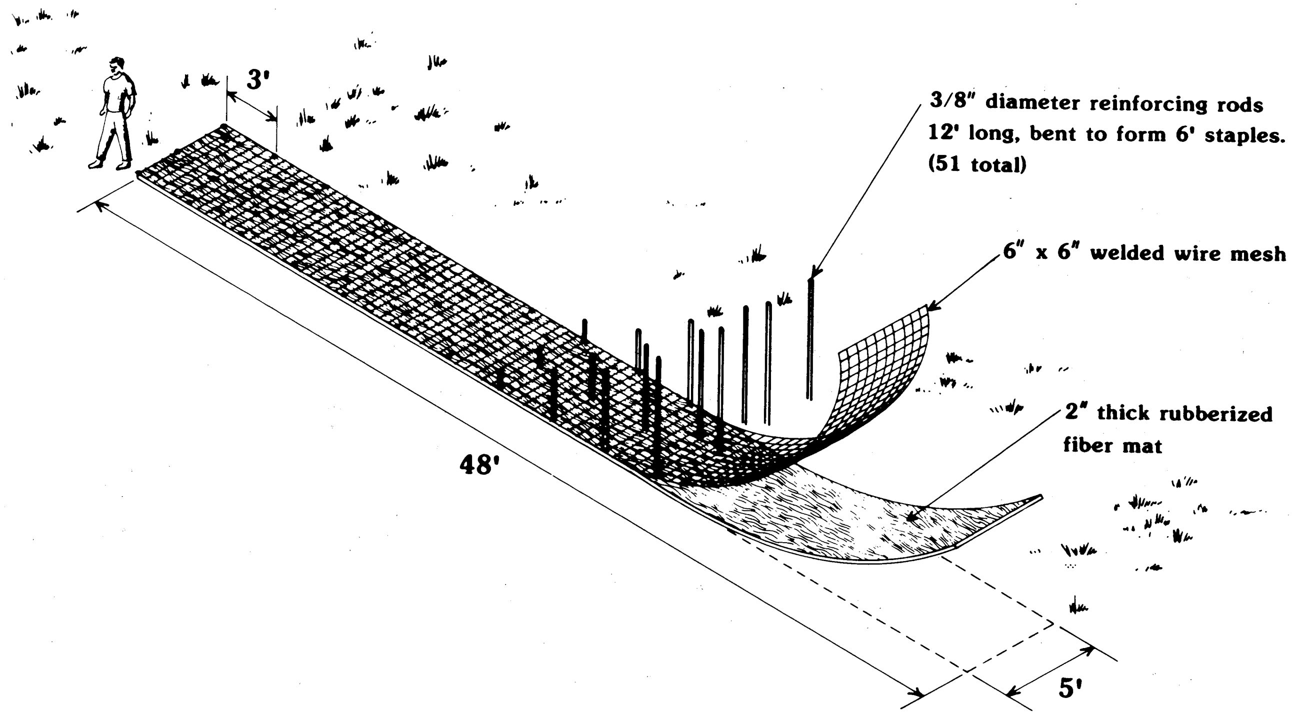

Figure 2. Fiber mat installation.

It was hypothesized that there would be an inverse relationship between survival and wave energy (as indicated by fetch). The results did not support this because most of the fences failed. The wave action caused the fence material to rub up and down against the supporting rods, and many of the fences were cut through in a few weeks. The severed fences washed back and forth over the plantings and caused a significant amount of damage; the fences were ultimately removed.

As a result of the above pilot study the fence was redesigned. Treated lumber posts spaced at 1.8-m (6-ft) intervals now are used as support for the fence material, which is attached to the posts with treated facing boards (Figure 3). This design has been used on several moderate wave energy projects along canals, and has performed satisfactorily so far. Additional testing of fence designs in high wave energy environments is planned.

The mat assemblage lasted for a year or longer except where there was scouring underneath the mat. It appears that in areas where this is a feasible approach, the present design may be over-built--the number of staples could probably be reduced by at least one-third without reducing mat durability. Wave energies, mat placement relative to mean water level, soil bearing capacities and the slope appeared to influence mat success. However, complications arising from the fence design render observations to date inconclusive. Further work on this concept is needed.

Hydroseeding

Research and development on wetland vegetation seed technology is urgently needed. Seeding may prove to be a very cost-effective way to introduce desirable plant species over vast areas of wetlands. *S. alterniflora* is an important species in this regard, but it is difficult to work with because it must be kept wet to insure satisfactory germination rates, and should be embedded in the soil in order to avoid washing out (Garbisch 1987; Woodhouse et al. 1974). Hydroseeding seems to be particularly well suited to this species because it fulfills both of these requirements.

The CVP completed an experimental hydroseeding project with 50,000 (pure live seed, Environmental Concern, Inc.) *S. alterniflora* seeds in a brackish marsh near Sabine National Wildlife Refuge (Figure 1, #3). In spite of a severe local drought, 15% of the seeds germinated and were successfully established 5 weeks after sowing.

IMPLEMENTATION PROJECTS

Several pilot planting projects have been sponsored. Two main strategies are being employed: soil stabilization (planting vegetation along an erosion-prone area to reduce wave energy and bind the sedi-

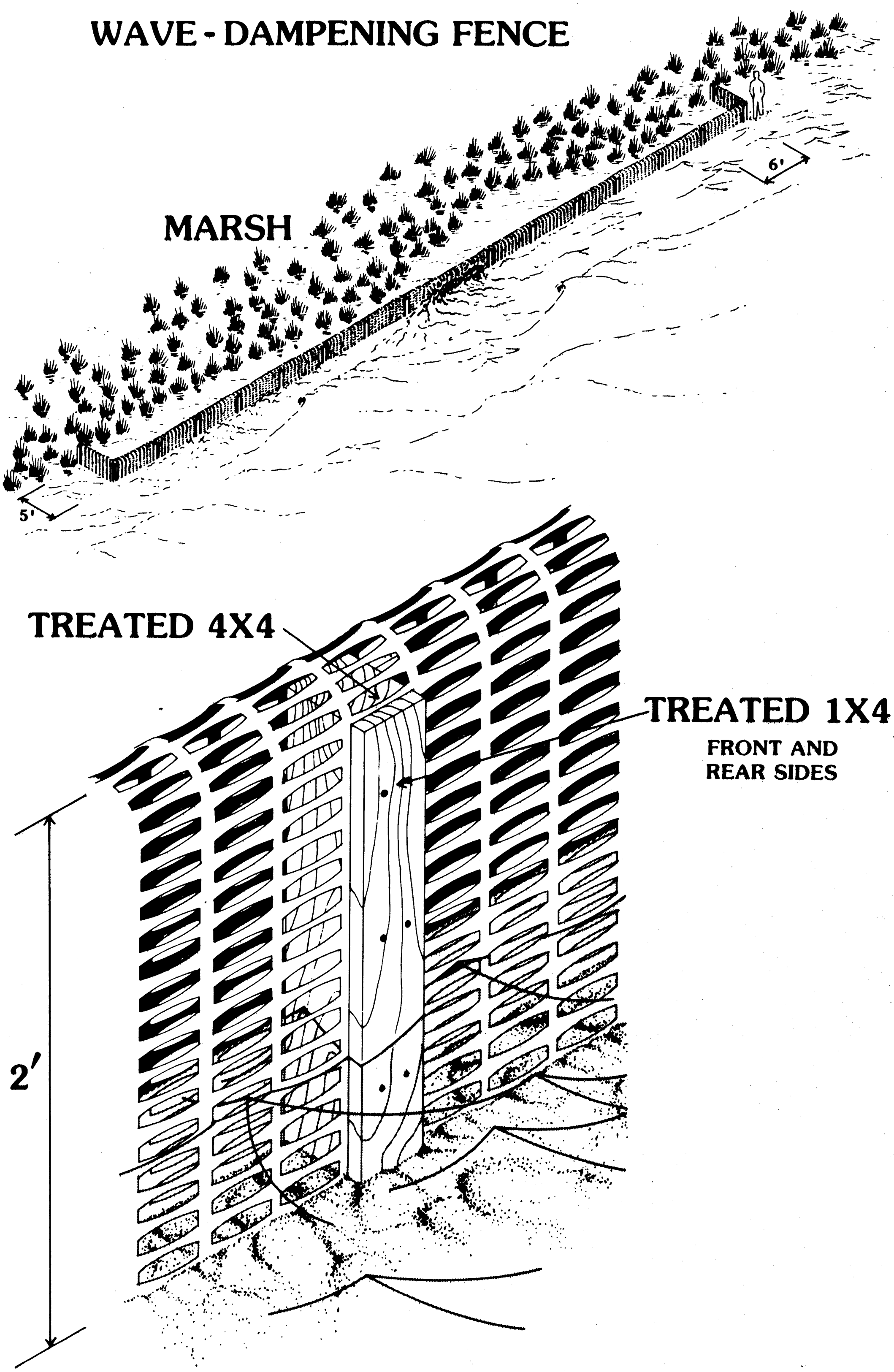

Figure 3. Wave-dampening fence.

ments together); and succession acceleration (the introduction of a salt-tolerant plant into brackish and intermediate areas that are in danger of becoming too saline for the existing vegetation).

Pilot Planting Projects Completed in 1987

Cutgrass planting: Grand Lake, Cameron Parish. For eight weeks two persons collected cutgrass (*Zizaniopsis miliaceae*) from local drainage ditches and planted it along the badly eroding northwest shore of Grand Lake (Figure 1, #5), within the Lacassine National Wildlife Refuge. The project was completed on 15 June 1987.

Only 2% of the transplants could be found on 28 July 1987. Two factors were responsible: nutria (*Myocastor coypus*) and water hyacinth (*Eichhornia crassipes*). The nutria preyed on the transplants by digging them up and eating the roots. This seemed to be more problematic for plants on and adjacent to the bank than for those planted a few feet out in the water. However, the plants in the water were more susceptible to damage from water hyacinth. If the wind blows huge rafts of water hyacinth against the shore, the plants may be scoured over in the process. That this had occurred on a large scale was evident during our monitoring trip of 28 July 1987.

Nutria are a pernicious problem for many transplanting schemes. Short of removing the nutria from the planting site, chemical or physical barriers appear to be the only possible deterrent. It is generally agreed that the chemical repellents now on the market are not effective against nutria.

Although physical barriers add substantially to the cost and time requirements per plant, they may be the only feasible way to successfully use vegetation to control erosion where nutria are abundant. Physical barriers, such as fences, could be designed to serve two additional important functions: they can eliminate damage to the plants from water-borne material, such as water hyacinth and debris; and they can also protect against washing-out by waves (Figure 4). Because it is semi-permanent, plastic fence material can become litter and a hazard to navigation. Two solutions to this problem are: 1) remove the fence material after stand establishment and use it for subsequent projects, or 2) use wooden snow fence or other materials that will eventually decompose.

Smooth cordgrass planting: Brown's Lake, Cameron Parish. For eight weeks two workers collected smooth cordgrass (*S. alterniflora*) from local salt marshes and planted it along an abandoned oil exploration access road north of Brown's lake at the Sabine National Wildlife Refuge (Figure 1, #2). This project was completed 22 July 1987.

When the planting sites were first examined on 27 July 1987, it was estimated that at least 90% of the smooth cordgrass transplants had survived. Many of those planted during the first few weeks of the

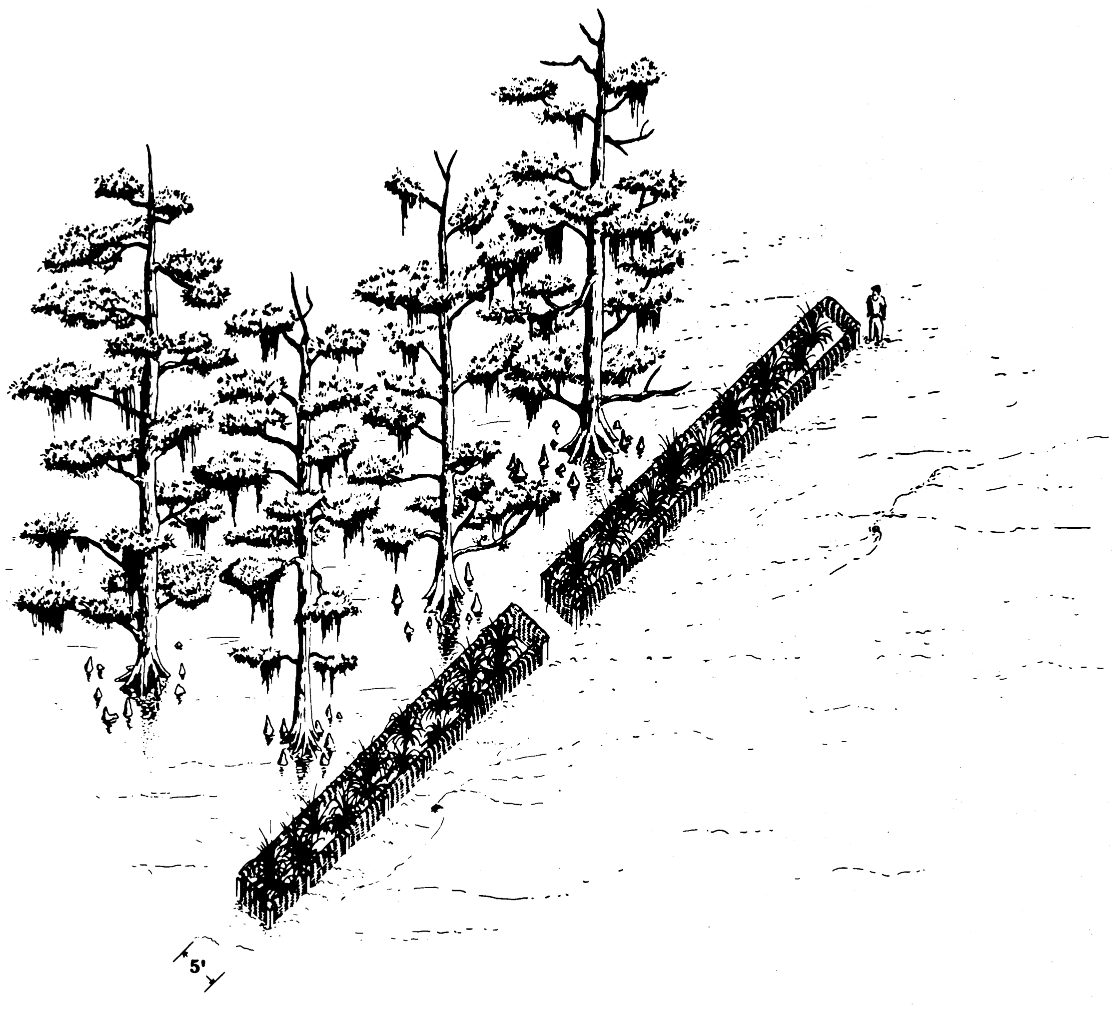

Figure 4. Combined wave-dampening fence and herbivore exclusion around an erosion-control planting.

project had prolific basal sprouts and other manifestations of vigorous growth.

Similar planting projects on the refuge have spread and coalesced rapidly, forming substantial stands of smooth cordgrass in two to three years. Based on this observation and the overall excellent condition of the transplants at the first monitoring, it appears likely that a substantial amount of wetlands will be stabilized through this project.

This project had three important favorable components: the substrate was high enough, the wave energy regime was relatively low, and the herbivore pressure was negligible. Because of its relatively low cost, this type of wetland restoration should be pursued wherever similar opportunities arise.

Pilot Planting Projects to be Completed in 1988

Private sector projects. The CVP designed and released for bid several relatively large vegetative wetland restoration projects that are currently being installed by commercial contractors.

1) 10,000 *Taxodium distichum* (baldcypress) seedlings will be planted at Lacassine National Wildlife Refuge with nutria exclosures around each tree (Figure 1, #6).

2) *S. alterniflora* will be planted along 9,060 m of eroding bayou bank and lake shore at the Sabine Wildlife Refuge (Figure 1, #4). 610 m of wave-dampening fence will protect the transplants in a moderate wave-energy section of this project.

3) A 960 m project will be installed along an eroding canal bank at Salvador Wildlife Management Area (Figure 1, #17). Several species (*S. alterniflora*, *Scirpus californicus*, *E. crassipes*, and *Typha* sp.) will be planted with and without wave-dampening fences so that both alternatives can be evaluated.

4) *S. alterniflora* will be planted along 2,220 m of eroding canal bank at Pointe aux Chenes Wildlife Management Area behind a series of wave-dampening fences (Figure 1, #12).

5) *S. alterniflora* will be planted along 14,640 m of bayou bank at the Clovelly marsh (Figure 1, #19) to prevent wetland deterioration through succession acceleration.

Interagency projects. In addition to the projects designed and released by the CVP for bid by private contractors, several interagency erosion control plantings were implemented and managed under a joint agreement by DNR, the Louisiana Soil and Water Conservation Commission (SWCC), and the U.S. Department of Agriculture Soil Conservation Service (SCS). Under this agreement CVP acted in an advisory role, while the design and implementation were carried out by SCS and SWCC.

The following projects were installed under this agreement:

1) A 1,370 m *S. alterniflora* planting near the Madisonville Lighthouse on Lake Pontchartrain (Figure 1, #13).

2) A 10,980 m *S. alterniflora* planting on the shoreline of Lake DeCade (Figure 1, #10).

3) A 3,050 m *S. alterniflora* planting along eroding "cutbanks" at Rollover Bayou (Figure 1, #9).

4) A 3,050 m *S. alterniflora* planting along the shoreline of Black Lake (Figure 1, #1).

5) A 1,373 m *Z. miliacea* planting along canal banks near Mallard Bay (Figure 1, #7).

6) *S. spartinae* dune stabilization trials using seed and transplants at Timbalier Island (Figure 1, #20).

MONITORING PROJECTS

Garden Island Bay Subdelta Crevasse Splay Development

The most significant monitoring project to date is that of three freshwater diversion projects at the Mississippi delta (Figure 1, #21). In this instance, the process of bay infilling is being followed from the dredging of crevasses through the development of emergent marsh communities.

It is imperative that crevasse splay formation processes be used effectively if the tremendous land loss problem in the Mississippi delta is to be reduced (Gagliano & van Beek 1970). Wetland loss here is more rapid than anywhere else in the entire deltaic plain region (Gagliano et al. 1981). The modern delta of the Mississippi River is a composite system of subdeltas that are all in a fairly advanced state of decay; it consists mainly of ring levees surrounding sunken interior bays. As a result, there are numerous opportunities to initiate crevasse splays through levee breaching.

Three such projects facilitated by CPS were completed in July 1986 at the Pass a Loutre Wildlife Management Area (Figure 5). Observations at the South Pass site indicate that this area is developing more rapidly than the other two. A well-defined system of distributaries and depositional centers was evident there by August 1987, whereas this was not the case at either Loomis Pass or Pass a Loutre (Figure 5). The South Pass site has an obvious advantage in hydrologic gradient because it empties directly into an arm of the Gulf, whereas both the Pass a Loutre and the Loomis Pass crevasses empty into interior bays. The hydrologic efficiency of the proposed parent-pass/crevasse-channel/

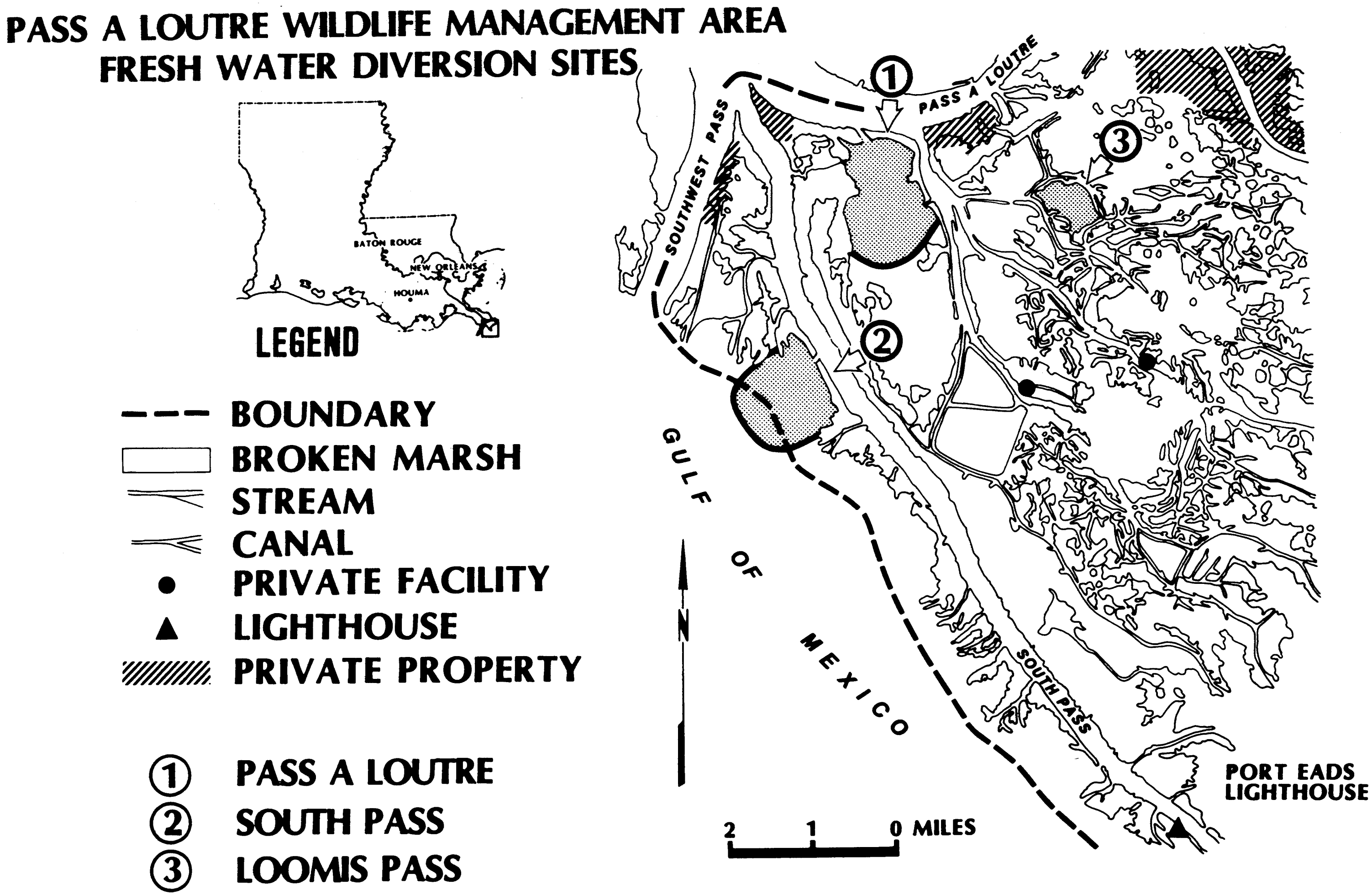

Figure 5. Location of the Pass a Loutre, South Pass, and Loomis Pass breaches and extent of new marsh expected to form.

receiving-bay system must be examined as a whole to predict the potential extent of splay development.

Subaerial accretion during major floods far exceeds that associated with minor floods (van Heerden 1980). There has not been a major flood in the area since the creation of these sediment diversion projects; therefore, sediment accretion has been minimal.

Montegut Marsh Restoration Project

This project (Figure 1, #11) was developed by the CPS in order to reduce marsh loss resulting from increasing levels of saltwater intrusion. As in many coastal marshes (Chabreck & Linscombe 1982; Sasser et al. 1987), subsidence and canal connections to the Gulf of Mexico have increased the influence of marine water in the area. As an apparent consequence, the vegetation has shifted from fresh-water marsh (O'Neil 1949) to a rapidly deteriorating brackish-intermediate marsh (Chabreck & Linscombe 1978).

The area is bounded on the west, north, and east sides by a natural levee system formed by the bifurcation of Bayou Terrebonne and Bayou St. Jean Charles. The thrust of the restoration project was to construct a levee and water control system across the southern portion of this area that would diminish the influence of marine water from the south. Before project construction, permanent transects were established inside and outside of the project area in order to collect data on vegetation cover, and standing crop by species. In addition, data was collected for the construction of a baseline vegetation map. The local marsh vegetation consists almost exclusively of *S. patens*, with very small amounts of *Distichlis spicata* and a few other species including *S. alterniflora*. It is expected that the marsh inside the project area will stabilize while that immediately outside will continue to deteriorate at pre-project rates, with some increase in the incidence of *S. alterniflora* possible.

La Branche Shoreline Repair Project

The La Branche wetlands (Figure 1, #15) on the southwest shore of Lake Pontchartrain are typical of many of the remaining marshes surrounding the lake. The marsh inland of the low lake berm has deteriorated, leaving numerous interior ponds often separated from the lake by little more than the lake berm. The berm in this area had been breached, and it was feared that the observed erosion of the interior wetlands would accelerate if the shoreline was not closed to reduce the influence of the lake on this interior wetland. A construction project was completed in July 1987 that reduced the impact of the lake on these wetlands by restoring the lake shoreline with new material. The CVP established four permanent transects in October 1987 to determine the relative cover of the vegetation present at that time. In addition, ground truthing was conducted for baseline vegetative type mapping of the entire area.

The marsh was dominated by *S. patens*, and several secondary species such as *Polygonum* sp., *Vigna luteola*, *Iva frutescens*, and *Ipomoea sagittata*. Because the purpose of this project was primarily to stabilize the area, rapid change in vegetational composition is not expected.

CONCLUSIONS

Because vegetative measures have proven to be an effective component of wetland erosion control, the CVP is an important part of Louisiana's effort to combat its coastal wetland loss. The program's primary goal is to maximize the effectiveness of vegetative techniques by using an integrated system of applied research, design, implementation, and monitoring. The CVP has been involved in 21 projects throughout the state so far.

Vegetative techniques can be effective in wetland restoration if the design is appropriate to site conditions. As the science of matching these design requirements to site conditions becomes more reliable, the cost effectiveness of vegetative restoration techniques will increase. Given the current rapid loss of Louisiana's marshlands and their importance to the state and the nation, the impetus for these projects is expected to increase.

ACKNOWLEDGEMENTS

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LITERATURE CITED

Allen, H. H., S. E. Shirley, and J. W. Webb. 1986. Vegetative stabilization of dredged material in moderate to high wave-energy environments for created wetlands. In F. Webb (Ed.), Proceedings of 13th Annual Conference on Wetlands Creation and Restoration. Hillsborough Community College, Tampa, Florida, May 15-16, pp. 19-35.

Chabreck, R. H., and G. Linscombe. 1978. Vegetative type map of the Louisiana coastal marshes. Department of Wildlife and Fisheries, New Orleans, Louisiana. Scale 1:62,500.

Chabreck, R. H., and G. Linscombe. 1982. Changes in vegetative types in Louisiana coastal marshes over a 10-year period. Louisiana Academy of Sciences 45:98-102.

Craig, N. J., R. E. Turner, and J. W. Day, Jr. 1979. Land loss in coastal Louisiana. In J. W. Day, D. D. Culley, Jr., R. E. Turner, and A. J. Humphrey, Jr. (Eds.), Proceedings of the third coastal marsh and estuary management symposium. Louisiana State University, Baton Rouge, March 6-7, 1978, pp. 227-254.

De Mond, J., D. R. Clark, and B. E. Spicer. 1986. A review of wetland restoration, enhancement, and creation practices in the Louisiana Coastal Zone. In F. Webb (Ed.), Proceedings of the 13th Annual Conference of Wetlands Restoration and Creation. Hillsborough Community College, Tampa, Florida, pp. 64-74.

Gagliano, S. M., K. J. Meyer-Arendt, and K. M. Wicker. 1981. Land loss in the Mississippi River deltaic plain. Transactions-Gulf Coast Association of Geological Societies 31:295-300.

Gagliano, S. M., and J. L. van Beek. 1970. Hydrologic and Geologic Studies of Coastal Louisiana. Rpt. number 1. Geologic and geomorphic studies of deltaic processes, Mississippi Delta system. Center for Wetland Resources, Louisiana State University, Baton Rouge, 140 pp.

Garbisch, E. W. 1987. Personal communication on December 15. Environmental Concern, President, St. Michaels, Maryland.

Louisiana Sea Grant. 1987. The imported plague. E. Coleman (Ed.), Aquanotes 16(4):1-5. Center for Wetland Resources, Louisiana State University, Baton Rouge.

O'Neil, T. 1949. Map of the southern part of Louisiana showing the vegetation types of the Louisiana marshes. Louisiana Department of Wildlife and Fisheries, New Orleans. Scale 1:62,500.

Sasser, C. E., M. D. Dozier, J. G. Gosselink, and J. M. Hill. 1986. Spatial and temporal changes in Louisiana's Barataria Basin marshes, 1945-19080. Environmental Management 10:671-680.

Scaife, W., R. E. Turner, and R. Costanza. 1983. Coastal Louisiana recent land loss and canal impacts. Environmental Management 7:433-442.

U.S. Army Corps of Engineers. 1987. Louisiana Coastal Area, Marsh Value Analysis, New Orleans District, Louisiana, 20pp.

van Heerden, I. L. 1980. Sedimentary Responses During Flood and Non-flood Conditions, New Atchafalaya Delta, Louisiana. M.S. thesis. Louisiana State University, 76 pp.

Wells, J. T., and J. M. Coleman. 1987. Wetland loss and the subdelta life cycle. Estuarine, Coastal and Shelf Science 25:111-125.

Woodhouse, W. W., Jr., E. D. Seneca, and S. W. Broome. 1974. Propagation of *Spartina alterniflora* for substrate stabilization of salt marsh development. Technical Memorandum No. 46. U.S. Army Corps of Engineers, Coastal Engineering Research Center, Ft. Belvoir, Virginia, 155 pp.

RECLAMATION OF SMALL STREAMS AND THEIR WATERSHEDS AT MOBIL'S CENTRAL FLORIDA PHOSPHATE MINES

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ABSTRACT

Mobil currently has 12 dredge and fill permits for the mining and reclamation of small stream channels at the company's central Florida phosphate mining operations. Reclamation activities have been completed on seven projects and are in progress on an eighth. This is the largest number of stream reclamation projects known to be under construction in the southeastern United States. Completed projects range in size and age from the initial 0.4 ha (1 ac) pilot scale project completed in 1980 to a 29.5 ha (73 ac) floodplain and associated wetland system completed in 1987.

A monitoring program for the revegetation efforts began in 1985. Four to six belt transects (elongated quadrats) were established across the streams and monitored once per year for the following parameters: tree height, crown cover, and condition of seedling. A control wetland was selected and sampled using the same methods. Two to three line intercept transects were also established at each stream to record the percent cover of herbaceous species. This data was used to calculate tree survival, tree density, percent crown cover, tree height, diversity indices, and Morisita's similarity indices. The oldest stream reclamation project, Sink Branch, has met all of the Department of Environmental Regulation's success criteria except for percent cover by primrose willow.

INTRODUCTION

Mobil Mining and Mineral Company, a subsidiary of Mobil Oil Corporation, owns three existing mines in the central Florida phosphate field. The Fort Meade Mine is located along the Peace River in Polk County. The Nichols Mine is located along the North Prong Alafia River also in Polk County. The Big Four Mine is located along the South Prong Alafia River in Hillsborough County. In addition to the three existing mines, Mobil also owns the South Fort Meade tract, a large undeveloped phosphate ore body along the Peace River in Polk County.

All four of Mobil's Mining tracts are located along either the Peace River or one of the two prongs of the Alafia River. At all four mining tracts, there are numerous small tributary streams leading to these larger streams. Significant phosphate reserves occur under these tributaries.

Typically, the small tributary streams have intermittent flow patterns and narrow low water channels, usually no more than a few feet wide. They generally have a wetland floodplain of various width along the low water channel. They are typically very heavily wooded and most people find them to be very aesthetically attractive. They represent valuable wildlife habitat and, perhaps most important of all, they contribute flow of good quality water to their receiving streams. Overall, Mobil recognizes these streams to be the most environmentally sensitive areas which the company is mining today.

Mining of tributary streams in Florida is strictly regulated by federal, state and in some cases, local authorities. In fact, before a mining company can mine even a minor tributary stream, it must, at a minimum, have the following permits and regulatory approvals:

- a dredge and fill permit from the Department of Environmental Regulation (DER)
- a dredge and fill permit from the Corps of Engineers (COE)
- a works of the district permit from the water management district
- a detailed reclamation plan on file with the Florida Bureau of Mine Reclamation

All these permitting steps require the preparation of detailed reclamation plans for the stream which is to be mined. In addition, the mining company must have demonstrated the ability to reclaim similar streams mined under previous permits.

Mobil has been involved with stream reclamation since 1979. The company has a total of eight currently active stream reclamation projects. The projects range in size and age from the initial 0.4 ha (1 ac) pilot scale project completed in 1980 to a 29.5 ha (73 ac) floodplain and associated wetland system completed in 1987.

During the eight-plus years in which Mobil has been involved with stream reclamation, a detailed methodology for designing projects has been developed. The methodology deals with both the physical design and revegetation design phases of the projects.

Physical Design of Streams and their Floodplains

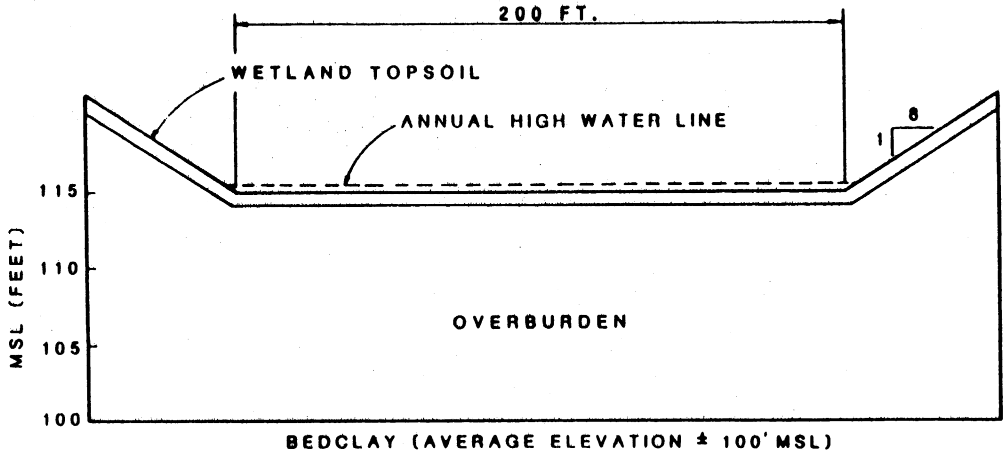
For the physical design of stream floodplains, Mobil relies on methods adapted from standard Soil Conservation Service (SCS) methodology for open channel flow. The first step in utilizing the SCS design methodology is to define the watershed for the reclaimed stream. Mobil designs its stream floodplains based on the ultimate post-reclamation conditions even though different drainage conditions usually prevail when mining is in progress. Active mining and waste disposal areas must drain to the mine water recirculation system which can discharge only through permitted and monitored points. Such areas do not contribute runoff to reclaimed streams until reclamation has been completed in the areas. However, since these areas will eventually contribute runoff to the reclaimed streams, the design of the stream floodplains must consider the peak flow contribution from them.

Once the size of the watershed and the general soil types in the watershed have been defined, the peak runoff can be estimated for various rainfall events. Then based on the predicted peak flows, the gradient of the floodplain, and a roughness factor for the reclaimed channel, a floodplain cross-section is designed to meet the design criteria. Mobil designs reclaimed floodplains to meet two criteria. First, the floodplain must be broad enough to carry the peak flow from a 25-year, 24-hour storm at nonerosive velocity. Second, the floodplain must be flat enough to be inundated by the flow generated by an annual, 24-hour storm.

Figure 1 depicts the design cross-section for the reclaimed floodplain of Rocky Branch. The design parameters and the calculated maximum flows, velocities, and water depths in the reclaimed floodplain are also provided in this figure. According to the calculations, the trapezoidal-shaped floodplain with a 61 m (200 ft) wide flat bottom is capable of carrying the peak flow from a 25-year, 24-hour storm at a velocity of 0.65 meters per second (2.14 feet per second). The threshold velocity for erosion is generally considered to be approximately 0.76-0.91 meters per second (2.5-3.0 feet per second); therefore, the floodplain meets the first design criterion. The flow from an annual, 24-hour storm produces a water depth of 0.2 m (0.66 ft), thereby inundating the entire 61 m (200 ft) wide floodplain and meeting the second design criterion.

In practice, the difficult part of floodplain construction is getting the floodplain cross-section flat and the gradient even down the length of the floodplain. Since wetland creation is the goal, equipment must operate very close to the water table when final grade is approached. It is very difficult to operate heavy equipment under these conditions. In Mobil's previous stream reclamation projects, the surface of the reclaimed floodplains is generally somewhat irregular as opposed to perfectly flat. This unevenness often creates small water pockets that provide microhabitats for aquatic organisms, particularly during the dry season. These microhabitats increase the diversity in a wetland system and are generally considered beneficial.

In most cases, Mobil has made no attempt to construct a low water channel within reclaimed floodplains. The first substantial rain usually generates enough flow to start the formation of a meandering low water channel. Ideally, the low water channel should meander throughout the floodplain. The key to achieving this is getting the floodplain as flat as possible.



DESIGN BASIS: (1) 24 HR., 1-YR. STORM = 3.75 INCHES OF RAINFALL
(2) 24 HR., 25-YR. STORM = 8.5 INCHES OF RAINFALL

DESIGN PARAMETERS:

DRAINAGE BASIN = 2000 ACRES
GRADIENT OF FLOODPLAIN = 0.0024
CHANNEL COEFFICIENT OF ROUGHNESS = 0.05
HYDROLOGIC GROUP B SOILS

CALCULATED MAXIMUM FLOWS, VELOCITIES AND DEPTHS:

	24 HR., 1-YR. STORM	24 HR., 25-YR. STORM
MAXIMUM FLOW	150 cfs	800 cfs
VELOCITY OF MAXIMUM FLOW	1.11 ft./sec.	2.14 ft./sec.
DEPTH OF MAXIMUM FLOW	0.66 ft.	1.75 ft.

Figure 1. Typical cross-section of reclaimed floodplain of Rocky Branch.

In most of Mobil's stream reclamation projects, a thin layer of wetland topsoil has been added to reclaimed floodplains as a finishing touch to the earthmoving phase. Whether this is essential or not is somewhat controversial. The DER certainly believes that it is and generally makes wetland topsoil replacement a condition of permit approval. In projects where it has been possible to transfer wetland topsoil directly to a reclamation site from another wetland that is to be mined, considerable volunteer propagation of desirable herbaceous wetland species has been observed. On the other hand, wetland topsoil is sometimes very difficult to acquire and, in cases where long-term stockpiling or destruction of unmined wetlands is required, Mobil regards wetland topsoiling as a very questionable practice.

Revegetation of Streams and their Floodplains

Mobil has used the Peace River and its tributaries as reference areas for designing the revegetation program for reclaimed floodplains. Figure 2 depicts a typical reforestation plan for a reclaimed floodplain. True wetland species such as bald cypress (*Taxodium distichum*), black gum (*Nyssa sylvatica* var. *biflora*), water hickory (*Fraxinus caroliniana*) are concentrated along the water channel. Near the margins of the floodplain, transition species such as laurel oak (*Quercus laurifolia*), water oak (*Quercus nigra*, sweetgum (*Liquidambar styraciflua*), Florida elm (*Ulmus americana*), and red maple (*Acer rubrum*) dominate the plantings. In addition, all projects have a greenbelt of transition and upland species on both sides of the reclaimed floodplain. Species such as slash pine (*Pinus elliottii*), live oak (*Quercus virginiana*), and pignut hickory (*Carya glabra*) are planted in this greenbelt along with laurel oak and water oak.

Containerized seedlings are used as the primary planting stock. These seedlings are essentially the same size as bare-root seedlings but have an intact soil and root mass. Whenever soil conditions permit, a standard tractor-drawn pine seedling transplanter is used to plant seedlings. Compared to hand planting, this is a much quicker and more cost efficient planting method. Hand planting is done in areas that are inaccessible to machinery.

Mobil uses two additional cultural practices to enhance the survival and growth of transplanted tree seedlings--subsoiling under planting rows prior to planting and irrigation of reforestation areas through their first growing season. During reclamation, the use of heavy machinery tends to compact some reclaimed soils and make it difficult for roots to penetrate them. Experience has shown that using a single row subsoiler under the rows prior to planting helps to loosen compacted soils and improve seedling survival. Irrigation of reforestation areas using portable volume guns has also become a standard practice for Mobil stream reclamation projects. Irrigation water is pumped from nearby mine pits. Seedlings are extremely susceptible to drought stress in their establishment year. Irrigation during this critical period has proven very beneficial in improving seedling

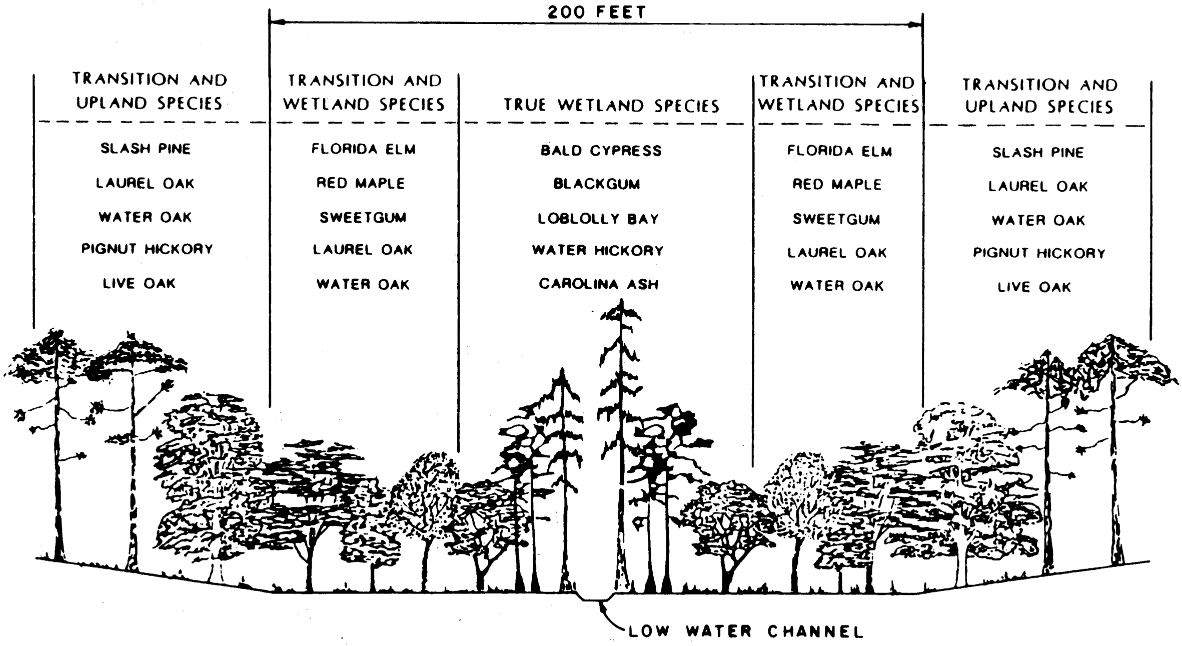


Figure 2. Typical revegetation plan for reclaimed floodplain.

survival. After the first growing season, reforestation areas depend on rainfall to maintain tree growth and survival.

STUDY SITE

Listed below are the eight stream reclamation projects and the control area that are described in the monitoring program.

Project Name	Designed Wetland Area Hectares (Acres)
• Myers Branch (Lower Portion)	0.8 ha (2 ac)
• Guy Branch	3.6 ha (9 ac)
• George Allen Creek	1.6 ha (4 ac)
• McCullough Creek	9.3 ha (23 ac)
• Sink Branch	0.4 ha (1 ac)
• Bird Branch, South	14.6 ha (36 ac)
• Bird Branch, North	15.0 ha (37 ac)
• Lake Branch Tributary	5.3 ha (13 ac)
• Rocky Branch (Control Area)	-- --

The locations of these projects are shown in Figures 3 and 4.

MATERIALS AND METHODS

Mobil began monitoring four of the projects in 1985. The monitoring program was established for the following reasons:

1. To enable Mobil to evaluate the success or failure of current reclamation practices so that future work can be improved.

2. To develop a data base that provides proof of successful reclamation in order to facilitate future permitting.

3. To satisfy Department of Environmental Regulation (DER) monitoring requirements.

DER's Monitoring Requirements

Mobil's twelve dredge and fill permits date from 1980 through 1985. During this time interval, monitoring required by the DER has increased considerably. Some of the early permits either had no monitoring requirement or had only very general requirements. The more recent permits have very explicit and detailed monitoring requirements. These requirements and success criteria for forested wetland restoration are summarized below.

a. An average of at least 988 trees per hectare (400 trees per acre) are growing above the herbaceous stratum;

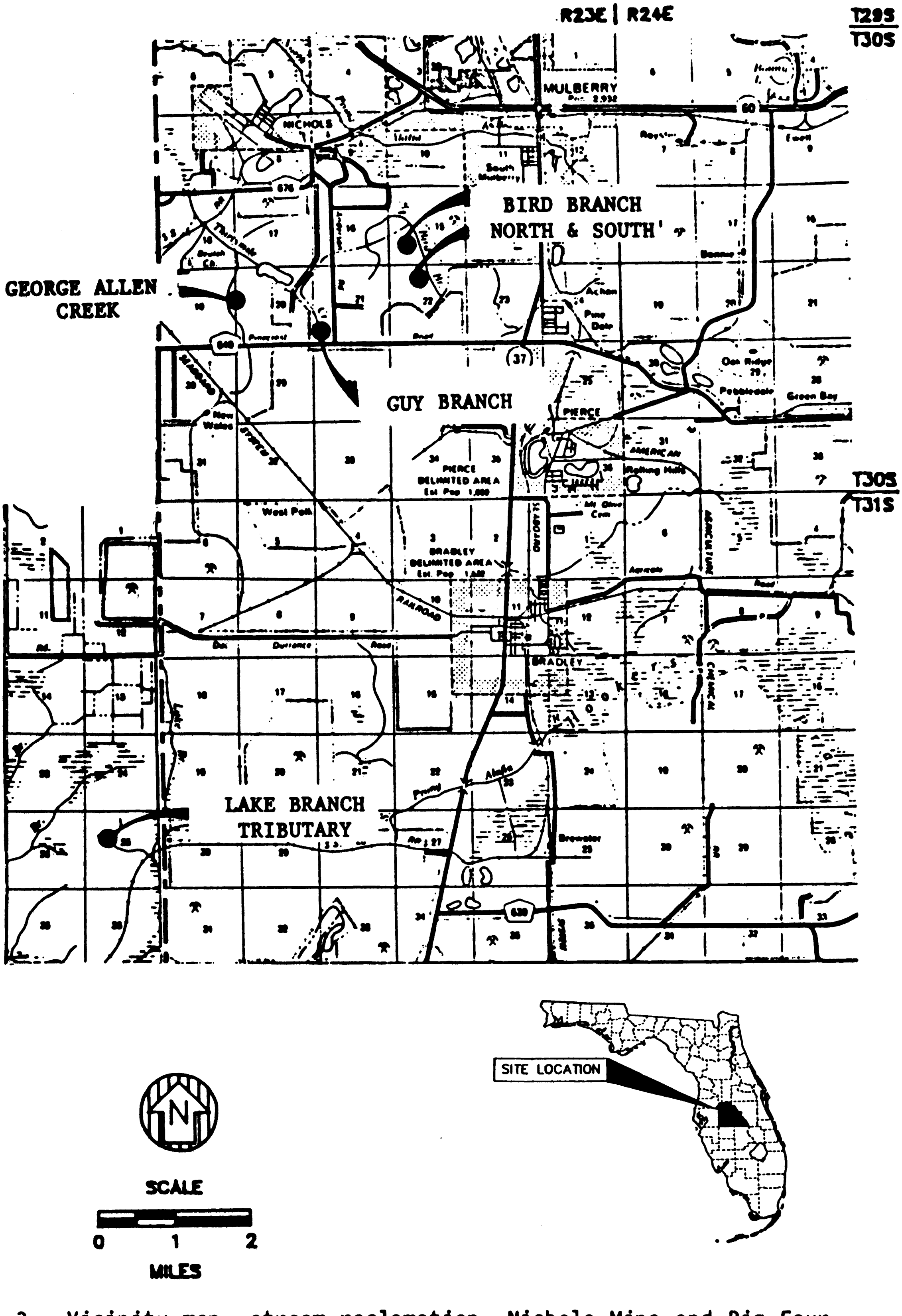
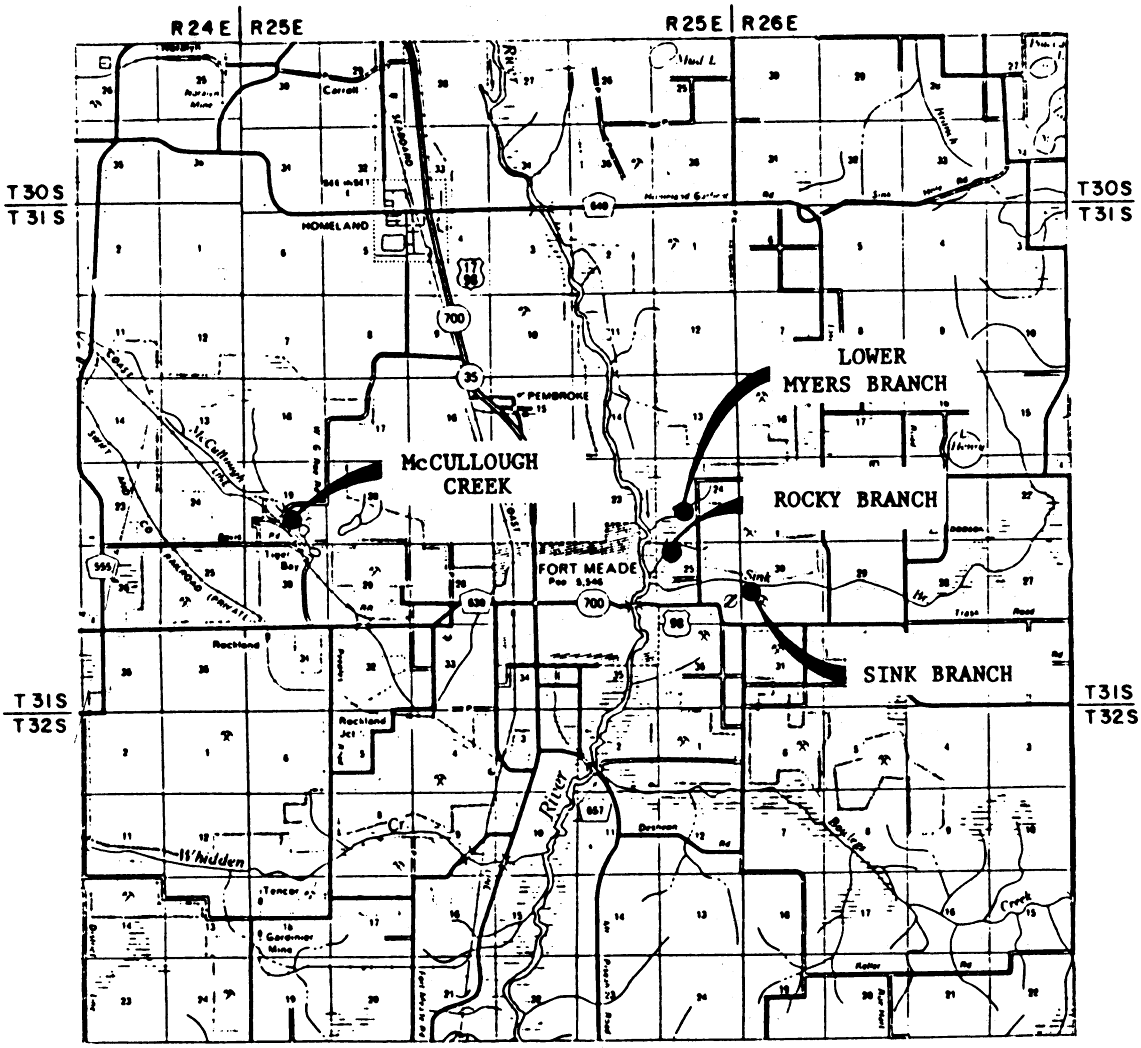
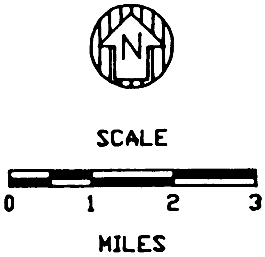


Figure 3. Vicinity map, stream reclamation, Nichols Mine and Big Four Mine.





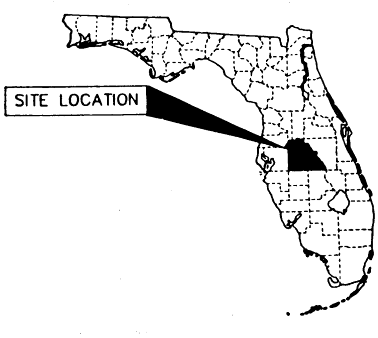


Figure 4. Vicinity map, stream reclamation, Fort Meade Mine.

b. The tree cover exceeds 33 percent of the vegetational cover and in no area of an acre or more in size is the tree cover less than 20 percent. Cover measurement will be limited to 1) those trees exceeding the herbaceous stratum in height, and 2) those indigenous species that contribute to the overstory of the mature riverine forest of the North Prong of the Alafia River (or the Peace River for Fort Meade reclamation projects).

c. Understory vegetation is reproducing naturally. Cattail, primrose willow and exotics are limited to 10 percent or less of the total cover. If these species exceed 10 percent of the total cover, their density must be declining over several years, which would be considered a positive indication that they are under control; and

d. The restored wetland has a similarity of 0.6 (as determined using Morisita's index) and 75 percent of the diversity of a mutually agreed upon control wetland (based on literature values, if possible).

In order to meet Mobil's goal of proving successful reclamation to facilitate future permitting, it is apparent that current monitoring requirements will have to be applied to the older projects even though such detailed monitoring may not be specifically required in the older permits. Mobil's monitoring program has, therefore, been designed to apply current monitoring requirements to all active dredge and fill reclamation projects.

Vegetation Sampling Methods

The vegetation sampling design was established by the Center for Wetlands during the 1984-1985 monitoring effort (Best et al. 1986). At Mobil's request, this design was repeated for the 1986 and 1987 study period. The sampling technique utilizes belt transects (elongated quadrats) for monitoring trees and a line-intercept transect for monitoring herbaceous plants. Sampling in 1986 and 1987 was conducted once each year between June 1 and July 15.

Tree sampling. Belt transects for trees were sampled as follows: Transects were established perpendicular to the slope and run across the stream. A permanent rod marks the starting point for the center line of the transect. A long measuring tape was placed along the center line of the transect across the permanent markers established at 30 meter intervals. The tape marks the center of the long axis of the belt transect. Seedlings within a 4.5 meter band on either side of the line were counted as part of the sample population. The following data was recorded for all tree seedlings encountered in the belt transect: species, height, crown diameter, condition of seedling, and water depth. Open water or marsh areas along the transects that were not planted with trees were excluded in calculations for tree density and crown cover.

The same sampling techniques described above were also used for the tree seedlings (less than 2.5 cm dbh) and saplings (2.5 cm to 10 cm) at Rocky Branch (Control Area). Only tree diameters were recorded for the overstory trees (greater than 10 cm dbh) at Rocky Branch.

Tree diversity was calculated using the Shannon diversity index (H'):

$$H' = (N \log N - \Sigma n_i \log n_i)/N$$

where N is total number of individuals, and n_1 is the number of individuals in species i.

Morisita's similarity indices were calculated in 1987 when the control wetland (Rocky Branch) was monitored. The Morisita index of community similarity (I_M) may range from 0 (no similarity) to approximately 1.0 (identical) (Brower & Zar, 1984). The similarity index is calculated as follows:

$$I_M = \frac{2 \Sigma x_i y_i}{(l_1 + l_2) N_1 N_2}$$

where N_1 and N_2 is the total number of individuals in communities 1 and 2, x_i and y_i is the number of individuals in species i in communities 1 and 2, and l_1 and l_2 are calculated as follows:

$$I_1 = \frac{\Sigma x_i (x_i - 1)}{N_1 (N_1 - 1)}$$

$$l_2 = \frac{\Sigma y_i (y_i - 1)}{N_2 (N_2 - 1)}$$

Herbaceous sampling. Herbaceous sampling transects were monitored using the line-intercept method. The method consists of taking measurements along a tape transect line placed across the study area. Each herbaceous species touching, overlying, or underlying the line was recorded along with the line distance that the species covers along the line. The individual intervals were totaled to yield percent cover for each species. Open water areas or upland areas along the transect were excluded in calculation of percent cover.

RESULTS

Tree Data

Tree survival. Tree seedling survival in 1987 ranged from 15 percent to 69 percent (Table 1). These results are based on the average planting density at each site and include any replanting efforts after the initial year of revegetation.

Tree density. Density of live trees in 1987 ranged from a low of

Table 1. Tree data for designed wetland area, 1987.

STREAM	AGE OF PROJECT (YEARS)	AVERAGE PLANTING DENSITY (TREES/AC)	LIVE TREES PER ACRE[1]	PERCENT SURVIVAL	AVERAGE HEIGHT (METERS)	AVERAGE CROWN DIAMETER (METERS)	PERCENT CROWN COVERAGE	SHANNON DIVER-SITY INDEX	SIMILAR-ITY INDEX[2]
Lower Myers Branch	4	1,335	195	15%	1.6	0.6	1.5%	0.75	0.5
Guy Branch	4	888	319	36%	1.1	0.4	1.2%	0.69	0.7
George Allen Cr.	3	1,005	321	32%	1.3	0.3	1.1%	0.63	0.3
McCullough Cr.	3	827	170	21%	1.4	0.5	1.0%	0.74	0.5
Sink Branch	7	904	619	69%	5.0	2.6	41.7%	0.85	0.7
Bird Branch, South	2	1.061	541	51%	0.6	0.2	0.6%	0.89	0.6
Bird Branch, North	1	1,387	485	35%	0.3	0.1	0.1%	0.97	0.5
Lake Br. Tributary	1	506	133	26%	0.4	0.2	0.1%	0.76	0.5
Rocky Br. (Control Area)									
- Overstory (4" dbh)	N/A	N/A	178	N/A	N/A	8.5"(dbh)	N/A	0.98	1.0
- Understory (1-4' dbh)	N/A	N/A	136	N/A	4.2	2.4	17.5	0.89	0.7
- Ground layer seedlings	N/A	N/A	7,832	N/A	0.5	0.3	11.8	0.80	0.6

[1]Tree data does not include volunteer Carolina willow (*Salix caroliniana*).

[2]The similarity index compares species composition of planted seedlings at the reclaimed sites to the overstory trees at Rocky Branch (Control Area).

SOURCE: Gurr & Associates, Inc., 1987.

329 trees per hectare (133 per acre) at Lake Branch Tributary to a high of 1530 trees per hectare (619 per acre) at Sink Branch (Table 1).

Five of the reclaimed wetland sites and the control area would not meet DER's current density requirement of 988 trees per hectare (400 per acre). For comparative purposes, the tree density at Rocky Branch (control area) was 776 trees per hectare (315 per acre) (combined overstory and understory).

Tree crown cover. Except for Sink Branch, all of the stream reclamation projects are relatively young and have very low canopy cover. Sink Branch is seven years old and exceeded DER's crown cover requirement of 33 percent in the sixth year. The percent crown cover for the other sites has doubled each year but is still less than 4 percent (Table 1).

Tree height. Average tree height for the projects (not including Sink Branch) is 0.9 meters (3.0 feet). Tree heights at Sink Branch averaged 4.5 meters (14.8 feet) (Table 1). The growth rate averaged 0.4 meters (1.2 feet) for the four projects that were monitored in 1985 through 1987.

Tree diversity. Tree diversities were relatively high, ranging from 0.63 to 0.97 (Table 1). The tree diversity at the control area (Rocky Branch) was 0.98 for the overstory and 0.89 for the understory (0.93 combined average).

The DER success criterion specifies that the diversity of the restored wetland shall be 75 percent of the control wetland but does not indicate what vegetative stratum is to be used. If the diversities for the overstory and understory for Rocky Branch are used for comparative purposes, the diversity for the reclaimed streams would need to be 0.69. All streams except George Allen Creek (0.63) meet this diversity requirement (Table 1).

Similarity indices. The DER success criterion specifies that the Morisita's similarity index for the restored and control wetland shall be 0.6 or greater. The average similarity index for all streams was 0.54, using the overstory trees from Rocky Branch as the control (Table 1). The similarity indices were lower when the understory or ground layer strata at Rocky Branch were used for comparison.

Some of the notable differences in species composition between the reclaimed wetlands and Rocky Branch (Control Area) are: the presence of slash pine at all reclaimed streams and the absence at Rocky Branch; an abundance of ironwood at Rocky Branch; and, a much greater abundance of swamp tupelo and water oak at the reclaimed stream sites. If Rocky Branch will continue to be used as the control area on future projects, the similarity indices could be increased by adjusting the species composition of the planted stock at the reclaimed sites.

Table 2. Summary of herbaceous vegetation[1] for all streams.

STREAM	AGE OF PROJECT (YEARS)	YEAR OF STUDY	PERCENT COVERAGE[2] ALL SPECIES	CATTAIL	PRIMROSE WILLOW	RELATIVE COVERAGE BY CATTAIL & PRIMROSE WILLOW[3]
Lower Myers Branch	2	1985	115.8	0.0	18.2	15.7
	3	1986	142.5	2.0	16.3	12.8
	4	1987	131.2	1.7	17.4	14.6
Guy Branch	2	1985	107.0	16.5	3.9	19.1
	3	1986	131.0	16.7	11.4	21.4
	4	1987	105.0	15.9	19.5	33.7
George Allen Cr.	1	1985	92.9	32.0	2.9	37.6
	2	1986	102.7	16.8	7.3	23.5
	3	1987	94.8	17.8	12.8	32.3
McCullough Cr.	1	1985	100.8	1.1	40.6	41.4
	2	1986	116.2	1.9	44.8	40.2
	3	1987	111.6	1.4	57.1	52.4
Sink Branch	6	1986	88.4	0.0	53.3	60.3
	7	1987	99.5	0.0	72.8	73.2
Bird Branch, South	1	1986	64.8	4.7	2.4	11.0
	2	1987	88.6	6.8	3.4	11.5
Bird Branch, North	1	1987	47.3	2.1	8.8	23.0
Lake Branch Tributary	1	1987	73.2	0.0	9.1	12.4

[1]Herbaceous vegetation consists of woody vegetation less than 2.5 cm dbh and all other herbaceous vegetation.

[2]Percent cover can exceed 100% due to overlapping coverage of different species.

[3]Relative coverage = (% cover of cattail + % cover of primrose willow) ÷ % cover by all species.

SOURCE: 1985 Data - Best et al.
1986 and 1987 Data - Gurr & Associates, Inc.

Herbaceous Cover

The percent cover of all herbaceous vegetation in 1987 ranged from 47.3 percent at Sink Branch to 131.2 percent at Myers Branch (Table 2). Percent cover can exceed 100 percent due to overlapping coverage of different species.

The relative coverage by cattail (*Typha* spp.) and primrose willow (shrub like varieties of *Ludwigia* spp.) varied widely and exceeded the 10 percent minimum required by DER. Coverage by cattail and primrose has increased slightly at the sites that were monitored for two successive years but clear trends in coverage by these species are lacking to date. The high percent cover of primrose willow at Sink Branch was concentrated along the 5 to 6 meter (15 to 20 feet) wide stream channel.

DISCUSSION

None of the eight stream reclamation sites meet all of DER's revegetation success criteria as specified in Mobil's most recent dredge and fill permits. However, this is not surprising considering the relatively young age of the projects.

The Sink Branch project is seven years old and has met DER's requirements for tree density, percent crown cover, diversity and similarity to a control wetland. The only success criterion that Sink Branch has not yet achieved was the percent cover of cattail and primrose willow. As the tree canopy continues to develop, it is reasonable to expect a decline in the coverage by these invader species.

Based on the vegetation growth at Sink Branch, it appears that DER's success criteria for Mobil's current dredge and fill permits can be met in approximately five to six years, except for the 10 percent cover limitation on cattail and primrose willow. Revegetation efforts should include a wide variety of species (ten species) to meet diversity requirements and sufficient planting densities to offset seedling mortality. The achievement of declining coverage by cattail and primrose willow may have to await the development of a more well developed tree canopy.

LITERATURE CITED

Best, G. R., R. A. Garren, M. H. Rector, and P. M. Wallace. 1986. A predictive assessment of long term trends towards forest development on Mobil's reclaimed phosphate mined lands. Final report to Mobil Chemical Co., Nichols, Florida. Prepared by Center for Wetlands, University of Florida - Gainesville.

Brower, J. E., and J. H. Zar. 1984. Field and laboratory methods for general ecology. Wm. C. Brown Publishers, Dubuque, Iowa, 226 pp.

Soil Conservation Service. 1975. Engineering field manual for conservation practices. Washington, D.C.

BANK STABILIZATION AND SHORELINE WILDLIFE HABITAT IMPROVEMENT IN A LARGE NORTH LOUISIANA RESERVOIR (TOLEDO BEND)

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ABSTRACT

Four species of vascular plants were evaluated under field conditions to determine their effectiveness for erosion prevention and shoreline stabilization. The four plants were: (1) cutgrass (*Zizaniopsis miliacea*) also known as southern wildrice and water millet, (2) maidencane (*Panicum hemitomon*), (3) spikerush (*Eleocharis macrostachya*), and (4) soft rush (*Juncus effusus*). A total of 1920 plants (480 of each species) were planted in two 12 x 15 meters (m) test plots. The plants were spaced at 25 cm intervals on the shoreline at elevations ranging from 169.5 ft. mean sea level (msl) to 171.5 ft. msl. Each planting was fertilized with 4.12 grams of Osmocote 17-7-12 controlled-release fertilizer at the time of planting (January-February, 1986). A barrier was constructed to protect the plantings from the waves. Plants were evaluated on their ability to survive and prosper (includes vigor, spread, etc.) under inundation periods varying from 65 to 121 days.

At all periods of inundation, cutgrass thrived and evidenced superior performance in relation to the other species tested. Data analysis indicates that this plant is the most effective in qualities of vigor, survival, spread, and apparent wave protection value. This was particularly evident for longer periods of inundation.

Soft rush and maidencane appear to have acceptable survival and coalescence characteristics during shorter periods of inundation (less than 90-100 days). Additional study is needed to determine the erosion protection effectiveness of these species.

Spikerush had excellent survival and spread at most periods of inundation. While the plant may have excellent promise as an erosion control species, additional time is needed to determine if it can be grown as effectively on eroded shorelines as it grows in its native habitat.

INTRODUCTION

The state of Louisiana ranks fourth in the United States in surface water acres with about 1.9 million acres of inland waters. Maintaining and improving water quality of its inland waters is necessary in order to continue to produce wildlife, fur, and fisheries. In addition, a multi-million dollar industry of recreation and tourism is directly dependent upon water that is environmentally and aesthetically acceptable. The demand for high quality freshwater for home, industry, and agriculture has never been higher, and with increasing growth of urban areas, population, and agricultural irrigation will continue to rise (Long et al. 1974). The last several years has revealed a new menace, that of shoreline erosion, which threatens to severely diminish the employment of freshwater impoundments for the above mentioned purposes. Although the problem has just become apparent, it is evident that erosion of shorelines will continue to accelerate. This erosion process has been proceeding since the construction of these impoundments but is now reaching the critical stage, with some shoreline already lost. The problem is one that is typically not considered in the planning of freshwater impoundments (Frantz 1951). It is directly related to the inability of upland hilltop soils, totally unsuited in structure, resilience, and type of vegetational communities supported, to withstand the effects of water level fluctuations, saturation, and wave action for extended periods of time. It has become evident that specialized research is needed to develop technologies to assist land and water control agencies and users in applying the best management practices for prevention or reduction of soil erosion from shorelines, thereby substantially improving water quality and wildlife habitat and lessening reservoir siltation. Currently, little is known concerning the capabilities and limitations of control of shoreline erosion of moderate-to-large freshwater impoundments by vegetative means (Draft 1983-84; Holmes 1985). The major emphasis relative to this has centered on seacoasts and beach areas (Born & Stephenson 1973; Cutshall 1985; Sharp & Vaden 1970; Sharp et al. 1980.) which provides little information of value to freshwater shoreline management.

The present report is the result of the first year of field work of a five year project undertaken to address the above problem. The objective of the five year study is to determine the most ecologically sound, aesthetically pleasing, and economically feasible method of shoreline erosion control of impounded lakes by the use of indigenous species of plants. The aim of the first year of study was to identify those species of plants that offer the most promise to achieve this end. Initially, attention was focused on determining which plants offered the greatest success in establishing themselves and simultaneously controlling erosion under the varying environmental conditions. Primarily, the environmental conditions impacting the shoreline of major concern are the long periods of drought alternating with extended periods of inundation and waves. Promising plants will be the object of continued studies.

DESCRIPTION OF THE STUDY AREA

The experimental sites were located in Sabine Parish, Louisiana, along the shoreline of Toledo Bend Reservoir (Figure 1), which covers 186,000 acres and is the fourth largest impounded lake in the United States (URS/Forrest & Colton 1979). It was created by the damming (late 1966) of the Sabine River, and now forms much of the border of Texas and Louisiana. The reservoir is 72 miles long north to south and varies from 1 to 5.5 miles wide. It has 1265 miles of shoreline on the Louisiana side which encompasses nearly every type of shoreline conditions typically found in freshwater impoundments within the state.

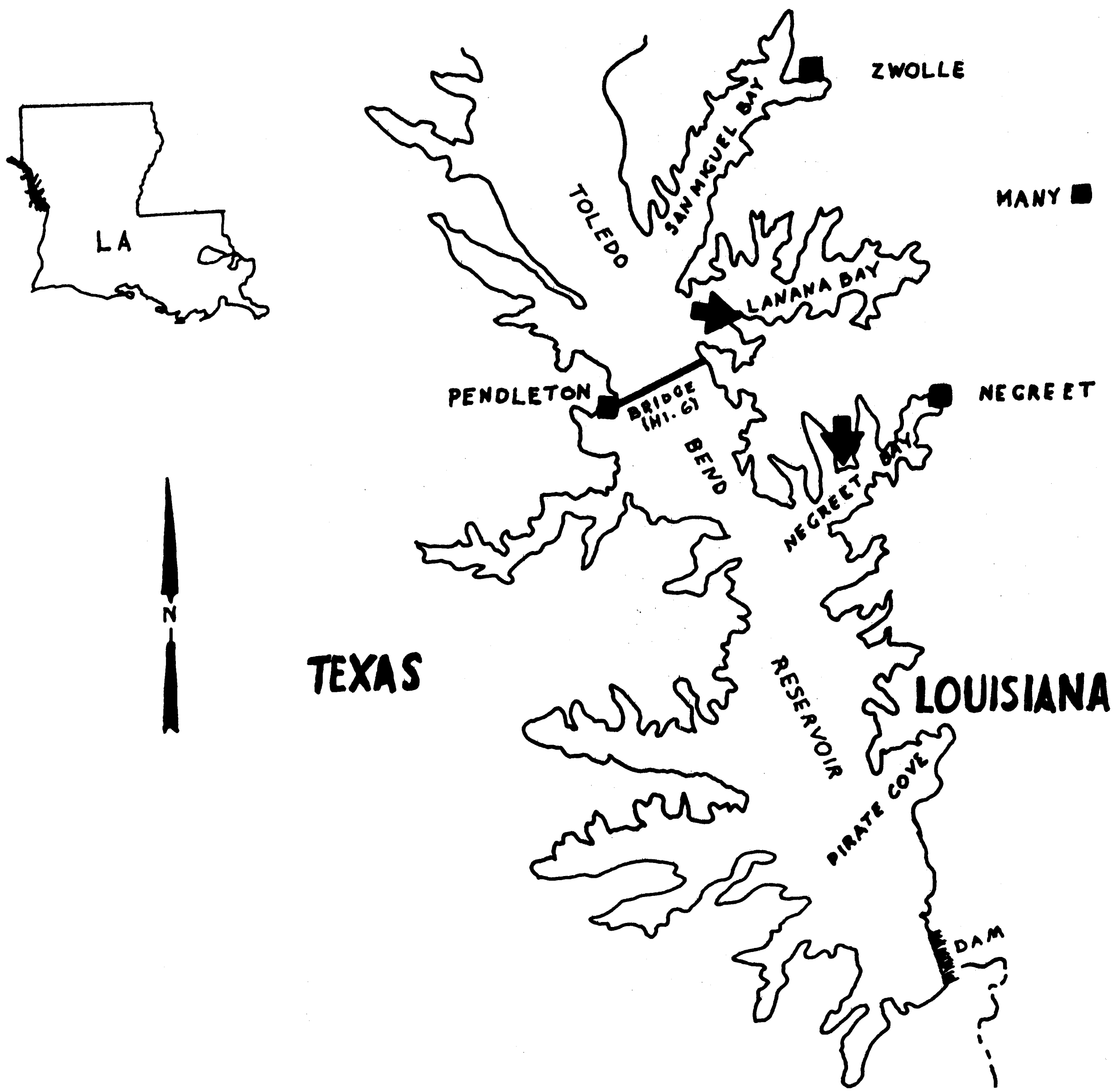


Figure 1. Location of the study site.

Additionally, large segments of the shoreline are nearly devoid of vegetation and are presently in various stages of erosion. The reservoir has a yearly fluctuation of water level that usually varies from a low pool stage of 168.0 ft. msl from September to January, to a high pool stage of 172.0 ft. msl in May and June. In exceptional years the pool stage may vary from a high of 173.0 or more to a low of 167.0 ft. msl (Figure 2). Prevailing winds are generally from the west, with winds of the greatest velocity occurring during the late winter through spring months. The shoreline is subjected to moderate to severe wave action, the principal erosion force, during these times. This erosion process appears to be accelerating because of the loss of standing timber, which, inundated at the time the reservoir was filled, served to moderate the wave action.

The area is also subjected to high humidity as rainfall averages 132cm per year. The maximum rainfall is attained during the month of May, while the minimum occurs during the autumn months. The average annual temperature is 19° C., summer average is 27° C., and winter average is 10.5° C. The average frost free period is 230 days and generally lasts from late March to Early November (Anderson 1960).

The soils of the study area are on moderately steep wooded hillsides. The topsoil is a brown to light yellow fine sandy loam that is medium to strongly acidic. These overlay a strong brown to light brownish gray clay that is very acidic (Soil Conservation Service 1973). On the shorelines of the reservoir, the topsoil is nearly to totally eroded away, exposing the clay subsoil.

STUDY METHODS

Plant Components

The general lack of specific information concerning plants suitable for shoreline erosion control of freshwater reservoirs required screening of indigenous vegetation for use in the plantings. The following criteria were used in selecting species for testing (Gray & Leiser 1982).

1. Ability of the plant to withstand extended periods of drought followed by long periods of inundation and waves.

2. Perennial, allowing propagation by cuttings, rhizomes, etc.

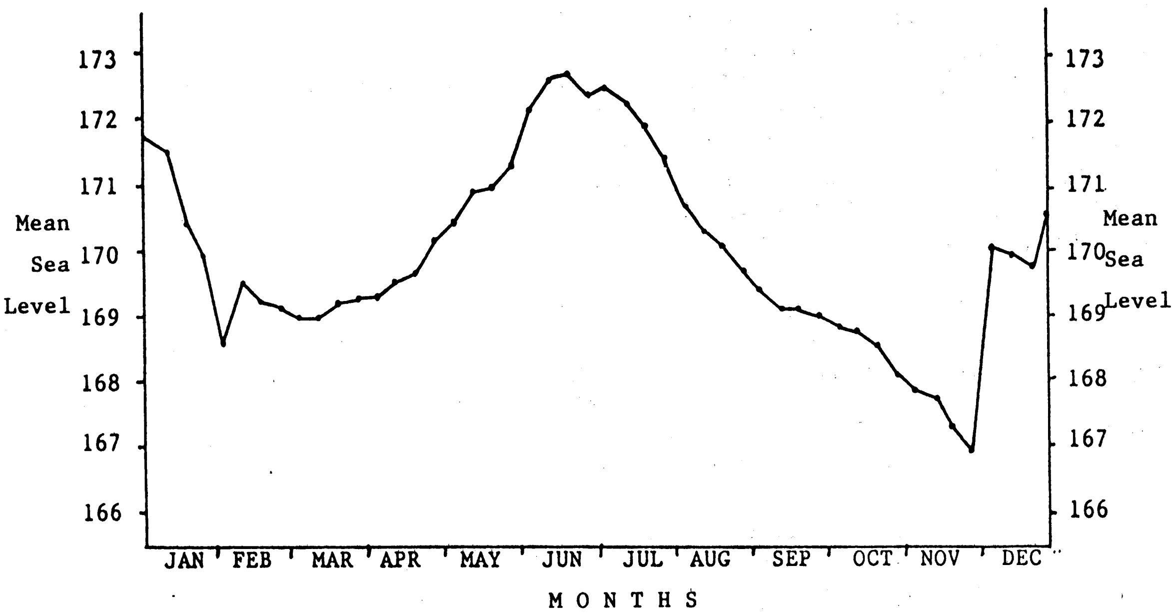
3. Growth potential, clumped growth form with soil binding ability.

4. Availability in the local area.

5. Value to wildlife.

6. Lack of undesirable (problem) characteristics.

Toledo Bend Reservoir Water Levels
(feet above mean sea level (MSL)) on
Days 1, 8, 15 & 22 for Each Month, 1986



Toledo Bend Reservoir Water Levels
(feet above mean sea level (MSL)) on
Days 1, 8, 15 & 22 for Each Month, 1987 (Inc.)

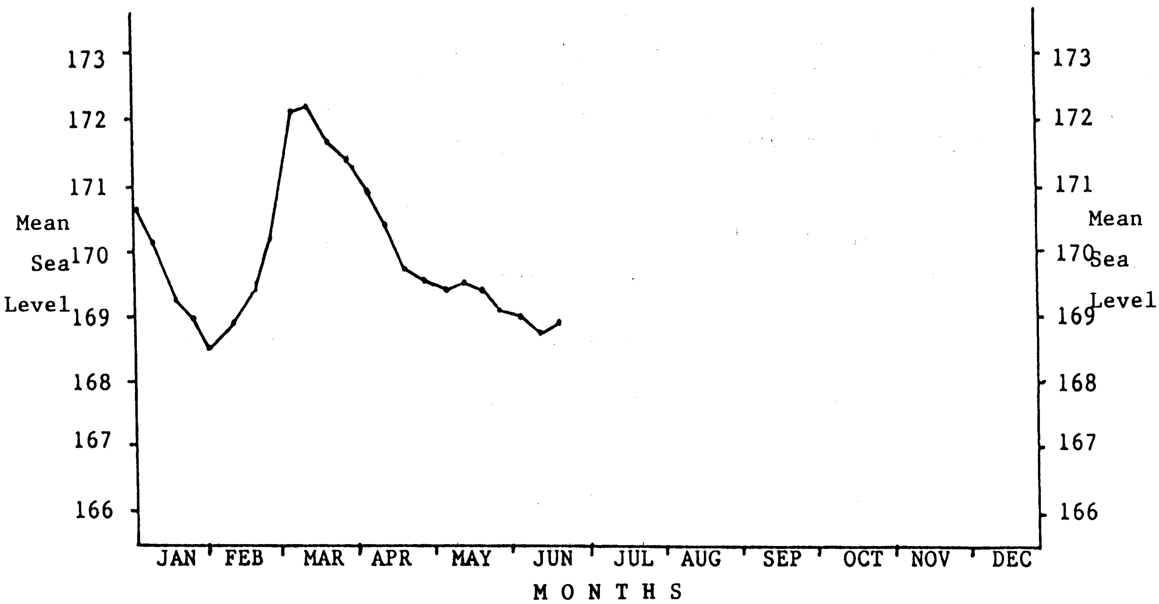


Figure 2.

Four species of vascular plants were selected for the first year's plantings. These included:

1. Cutgrass (Zizaniopsis miliacea (Michx.) Doell. & Asch. This plant is also called southern wildrice and water millet.

2. Maidencane (Panicum hemitomon Schult.).

3. Soft rush (Juncus effusus L.).

4. Spikerush (Eleocharis macrostachya Britt.).

Planting materials were collected in December 1985 and January 1986. Soft rush and spikerush were collected by using a bulb planter which removed a three inch diameter by five inch long plug. Cutgrass and maidencane were collected by digging the rhizomes which were later cut to planting size. All plants were placed in containers, covered to prevent drying, then stored at 3.3° C. until planting time.

Test Site Establishment

In October and November, 1985, two moderately eroding shorelines were selected for establishment of test plots. Each plot was 12 x 5 m, the greater dimension being parallel to the water. Plot elevations (in feet msl) varied from 169.49 to 171.51 for the North site and 169.85 to 171.20 for the South site. Each plot was divided into three equal subunits of 4 m x 5 m. The subunits were further subdivided into strips 1 m wide and 5 m long (replicates) with the greater dimension being perpendicular to the water. Each of the 4 strips of each subunit was planted with one of the plant species being tested. Locations of replicates in a subunit were determined randomly. Individual plantings were spaced at 25 cm, thus each square meter was planted with 16 plants, and each 1 x 5 m strip with 80 plants. This planting arrangement was repeated in the other two subunits, with the location of each replicate also being determined randomly. Each 12 x 5 m plot thus contained 960 individual plantings equally divided between the four test species. Each plant was fertilized with 4.12 grams (equivalent to 100 lbs. of Nitrogen per acre) of Osmocote controlled-release fertilizer at the time of planting. The elevation of each plant was determined so that the number of days of inundation could be computed by comparing its elevation with reservoir stage levels (obtained from the reservoir engineer's office). A plant was considered inundated if the reservoir pool stage (in feet msl) equaled or exceeded the elevation of the plant. The South site was established 31 January through 1 February, 1986. The North site was planted 9-10 February, 1986. Each plot was surrounded, on the lake side, by snow fencing to protect the plantings from waves (Webb & Dodd 1983) and from being covered by washed-in aquatic vegetation. Plots were watered weekly as required during periods of non-inundation.

Preinundation survivability for both plots was evaluated on 4

April 1986. Known non-survivors of soft rush, maidencane, and spikerush were replanted at that time assuring 100% survival prior to inundation. Very retarded springtime growth of cutgrass, apparently the normal situation, prevented similar determination of survivability and replanting in April. Heavy rains in the reservoir watershed followed, causing the reservoir pool stage to increase continuously until early June, when it attained a pool stage of over 173 feet (Rumsey 1984-1987). This prevented accurate determination of preinundation survivability and replanting of non-survivors. Inundation times of the plantings varied from 65 to 121 days. Post inundation survival and vigor were evaluated in September and October. The spreading growth habit and numerous new stems produced by spikerush prohibited evaluation of post inundation survival and vigor. Instead, five 25 x 25 cm (0.16 m^2) stratified-random quadrats were taken in each replication and used to compute density. Vigor ratings were based on the number of stems present, number of leaves, and height of the plant. Vigor was used to compare the condition and success of plants subjected to the various inundation periods. The different growth forms for the species being tested mandated that each have its own vigor criteria (cutgrass vigor = no. stems x no. leaves x plant height; soft rush vigor = no. stems x plant height; maidencane vigor = no. rooted plants x plant height; spikerush vigor = no. stems per quadrat). This prevented interspecific comparison of vigor. Regression coefficients were calculated for each species on each replicate (days of inundation vs. vigor for each propagule).

RESULTS

Preinundation Survival

Preinundation survival was evaluated two months after test plot establishment. All species tested exhibited excellent survival (Table 1), irregardless of position on the shoreline.

Table 1. Inundation survival in percent. (Pre-April 1986; Post-Sept. 1986)

SPECIES	NORTH PLOT Pre	NORTH PLOT Post	SOUTH PLOT Pre	SOUTH PLOT Post	MEANS Pre	MEANS Post
SPIKERUSH	99.2	*	99.2	*	99.2	*
SOFT RUSH	95.2	39.6	95.8	47.1	95.5	43.4
MAIDENCANE	93.3	58.8	87.1	46.3	90.2	52.6
CUTGRASS	87.1	70.9	85.0	54.2	86.0	62.6

*Information not available due to plant coalescence.

Spikerush demonstrated the highest percentage survival at nearly 100%, with the failure of the very few survivors attributable to incorrect planting. Most of the plants developed flower clusters, but indications of spreading were not evident. Soft rush had a survival rate averaging slightly over 95% on both plots. Numerous plants developed flower clusters, but lateral spread appeared inhibited. Maidencane had an acceptable survival rate of 93.3% and 87.1%. Although easily detectable as survivors, the species showed a very slow growth rate during this period. Production of flowers was not expected since the plant rarely flowers even under optimum conditions and if flowering does occur, it is in the autumn. Preinundation survival for cutgrass was difficult to evaluate. The plant apparently is slow to initiate growth in the spring, often taking months to show positive growth signs. At this stage, numerous plantings were brown and dried and showed signs of decay. Removal of several planters showed positive growth was occurring below ground. The survival figures are adjusted to include as preinundation survivors those plants suspected of not surviving but later found to have survived the period of inundation. The actual preinundation survival should be slightly higher than the figures presented, but it could not be accurately determined if the plants were dead prior to inundation or if they failed to survive the inundation period. Both plots has survival percentages of 85% or more, which is exceptionally good. Growth consisted mainly of culm elongation and production of leaves. Flowering did not occur because of the small size of the plants.

Post Inundation Survival

Table 1 shows there was a definite trend for soft rush and maidencane to have dramatically decreased survivability as inundation time increased. This was particularly obvious in the South plot. Cutgrass on the South plot showed this same general trend, but its survivability on the North plot seemed to be little affected by the length of inundation. Due to the rapidly spreading nature of spikerush in favorable areas, determination of survival in individual plantings was impossible.

Table 2 lists by plots and replications the mean and standard error for days of inundation and for vigor in the surviving plants. Due to previously mentioned problems with spikerush, data collection was done by a previously explained quadrat sampling procedure. Mean inundation times were longer on the North plot and vigor values for cutgrass and maidencane were obviously higher here. However, more vigor was seen in the soft rush and spikerush survivors in the South plot.

Mean inundation, mean vigor, and regression coefficients were calculated for each replicate, using the plant vigor (0 if it died) and length of inundation for each replicate's planting (Table 3). As in Table 2, obvious differences occur in replicate vigor values for all four species. Positive regression coefficients indicate a positive

Table 2. Sample size, days of inundation, and vigor of surviving plants.

SPECIES, & AREA	SAMPLE SIZE	DAYS OF INUNDATION MEAN	DAYS OF INUNDATION S.E.	VIGOR MEAN	VIGOR S.E.
CUT GRASS					
*NP,N	64 plantings	98.9	1.40	42.0	5.18
NP,M	59	92.8	1.93	52.3	8.78
NP,S	47	94.3	2.53	55.0	8.23
SP,N	57	88.2	0.83	26.9	2.48
SP,M	38	88.2	1.49	24.1	3.94
SP,S	35	91.3	1.70	39.1	6.29
MAIDENCANE					
NP,N	47 plantings	96.4	1.36	14.7	1.11
NP,M	54	91.0	1.88	12.6	0.88
NP,S	40	85.8	2.40	7.7	0.68
SP,N	41	85.4	1.24	6.6	0.72
SP,M	35	84.4	1.02	5.7	0.65
SP,S	35	84.3	1.11	5.7	0.65
SOFT RUSH					
NP,N	29 plantings	86.2	1.39	11.5	1.66
NP,M	42	86.8	1.92	19.4	2.14
NP,S	24	84.6	2.83	12.1	1.97
SP,N	48	84.4	0.74	19.5	1.66
SP,M	33	83.9	0.61	18.7	1.79
SP,S	32	81.5	1.11	12.0	1.26
SPIKERUSH					
NP	14 quadrats	92.0	4.09	27.0	5.47
SP	14	88.4	2.58	39.0	7.86

*NP = North Plot
SP = South Plot
N = North Replicate
M = Middle Replicate
S = South Replicate

Table 3. Replicate's mean inundation days, mean vigor, and regression coefficients.

SPECIES, & AREA	SAMPLE SIZE	MEAN INUNDATION	MEAN VIGOR	REGRESSION COEFFICIENT
CUTGRASS				
*NP,N	80 plantings	98.70 days	33.58	1.50
NP,M	80	92.95	38.57	1.46
NP,S	80	93.70	32.32	1.10
SP,N	80	88.35	19.17	0.97
SP,M	80	89.40	11.44	0.15
SP,S	80	91.40	17.11	0.96
	N = 480	$\bar{X}$ = 92.42		$\bar{X}$ = +1.02
MAIDENCANE				
NP,N	80 plantings	101.00 days	8.65	-0.41
NP,M	80	94.15	8.48	-0.09
NP,S	80	93.55	3.84	-0.16
SP,N	80	87.35	3.39	-0.07
SP,M	80	88.00	2.50	-0.12
SP,S	80	90.60	2.49	-0.11
	N = 480	$\bar{X}$ = 92.44		$\bar{X}$ = -0.16
SOFT RUSH				
NP,N	80 plantings	95.55 days	4.17	-0.33
NP,M	80	92.45	10.17	-0.49
NP,S	80	94.15	3.63	-0.24
SP,N	80	88.75	11.55	-1.09
SP,M	80	91.25	7.72	-0.59
SP,S	80	87.95	4.72	-0.34
	N = 480	$\bar{X}$ = 91.68		$\bar{X}$ = -0.51
SPIKERUSH				
NP	15 quadrats	93.33 days	25.20	-0.46
SP	15	89.47	36.40	-1.35
	N = 30	$\bar{X}$ = 91.40		$\bar{X}$ = -0.91

*NP = North Plot
SP = South Plot
N = North Replicate
M = Middle Replicate
S = South Replicate

relationship between length of inundation and vigor with an inverse relationship noted by negative regression values. Only cutgrass showed a positive effect with increased inundation and spikerush showed the most negative response to inundation.

DISCUSSION

Preinundation Survival

The results of the study showed that the four species tested can be successfully transplanted to and established on the eroded shoreline of Toledo Bend reservoir. All species tested exhibited very good to excellent survival (Table 1). Watering during dry periods appears to have negated differences which could be caused by differential moisture retention at the various plot levels or positions relative to the reservoir. The size and growth form of the species apparently has an affect on the survival percentages. Spikerush, with densely tufted culms averaging 25 cm in height (Correll & Correll 1972) displayed the highest survival rate. Soft rush, with an overall survival of 95.5%, and maidencane, with a survival of 90.2%, both have culms about one meter in height, but the former has a densely clumped growth form and the latter a creeping growth form. Cutgrass, the largest of the plants with a culm height of 3 m and a densely clustered growth, showed the lowest overall survival percentage of 86%. The apparent trend is that smaller sized and denser culmed species exhibit a greater survival because more of the plant (or plants) were transplanted into the same sized planting hole. These relatively larger plantings (as compared with the size of the plant) also had more rhizomes, roots, buds, and culms thus increasing transplanting success. This is especially noticeable when a plug of the plant is used as a planter, as was done with spikerush and soft rush. The plantings of maidencane consisted of hand-cut portions of the rhizomes which relatively reduced the amount of planting material per planting hole. The planting material of cutgrass consisted of a portion of a coarse and thick (about 1 cm) rhizome with at least one bud present. No culms or roots were noticeable, thus lowering transplanting success. This may also, in part, be responsible for the slow initial growth rate exhibited by the species. Consideration should be given to another method of propagation or possibly a later planting date (March).

Evidence of lateral spread was not apparent during the preinundation period. It may have been impeded by the tight clayey nature of the soils, which although watered regularly, were not as pliable as they would have been during inundation. Additional observations are needed on this subject.

Post Inundation Survival and Vigor

The decreased survivability of maidencane and soft rush (see Table 1) with longer inundation times was probably due to several factors, but certainly plant height played an important role. Cutgrass survivability was only slightly diminished by prolonged inundation and never in actively growing plants was the entire plant totally inundated. Total submersion of the other three species undoubtedly created oxygen deficiency problems and greatly slowed photosynthesis due to decreased light intensities and limited availability of gaseous carbon dioxide. The very spongy nature of the cutgrass leaf sheaths and its long, thin leaves probably aid in the uptake of oxygen and carbon dioxide from the air and its transport to the submerged plant base. The sturdy nature of well established cutgrass plants also helps to decrease planter loss due to wave action.

The longer mean inundation days on cutgrass, maidencane, and soft rush survivors (Table 2) typically results in greater vigor, but as previously noted, did reduce survivability. This would appear to indicate that soil saturation with water is desirable for maximum growth stimulation of these species. Spikerush would appear to be more vigorous under less saturated or inundated conditions.

The regression coefficients (Table 3) indicate that cutgrass responds positively to increased lengths of inundation, but does so weakly. Spikerush, with the smallest coefficient, is most adversely affected by inundation on the South plot. Maidencane's regression coefficient approaches zero suggesting that barring wave damage, would do equally well at all levels. Soft rush's coefficient fell between these last two species and thus probably planting at upper levels would be best.

The differences noted in the vigor versus inundation regression coefficients for the four species replicates are felt to be due to several factors, but two are of primary importance. First, the exposure of the plants to wave action. Although a wave barrier fence was built across the front of the plot, winds blowing at 60° or more angle to either side of a line from the plot center to the fence center could cause considerable damage especially during the exceedingly high pool stage of the reservoir in June. Damage would be caused by direct wave damage and by washing in of _Hydrilla_ and the scouring action of its wave tossed strands. Secondly, soil types varied from one end of the plot to the other and no doubt greatly affected plant growth due to nutrient availability, leaching of fertilizer, porosity, stability to roots, and other similar factors. The low vigor values seen in the cutgrass and maidencane replicates of the South plot are felt to be caused by poor soil characteristics; whereas more desirable soil characteristics resulted in the high correlations seen in all three species on the middle replicates of the North plot.

CONCLUSIONS

The four species of plants tested can be successfully transplanted to and survive the environmental conditions of the eroded shoreline of Toledo Bend Reservoir. It should be realized that the time period of investigation was unusual in water heights and wind severity compared with past records of the area. It is expected that more moderate environmental constraints will allow higher survival rates in the plantings and more vigorous growth in most species. Relationships between growth success and soil saturation conditions indicate that a regular watering scheme to keep the soil saturated during non-inundation periods would increase planting success. Results of the study showed that cutgrass, because of its large size and sturdy growth characteristics, demonstrated the greatest potential for wave protection and erosion control at all inundation times tested. Soft rush, spikerush, and maidencane, while probably suitable for erosion control in areas subject to short periods of inundation, need further study.

In addition to protection from waves during the establishment phase, the use of a wave-stilling device appears essential to avert damage to the plantings that can be caused by the scouring and wrenching action of masses of aquatic vegetation that are periodically washed ashore.

LITERATURE CITED

Anderson, H. V. 1960. Geology of Sabine Parish. Department of Conservation, Louisiana Geological Survey, Baton Rouge, 164 pp.

Born, S. M., and D. A. Stephenson. 1973. Water management for shoreline erosion control on the Chippewa flowage. Journ. Soil & Water Conserv. 27:57-61.

Correll, D. S., and H. B. Correll. 1972. Aquatic and wetland plants of southwestern United States. U.S. Government Printing Office, Washington, D.C., 1777 pp.

Craft, B. 1983-84. Test plot for shoreline erosion control on Toledo Bend Reservoir. Unpub. Report, SCS, Alexandria, Louisiana, 6 pp.

Cutshall, J. R. 1985. Vegetative establishment of smooth cordgrass (Spartina alterniflora) for shoreline erosion control. Unpub. Report, SCS, Alexandria, Louisiana, 6 pp.

Frantz, C. F. 1951. Field notes and comments: Non-farm erosion problem often neglected by engineers. Journ. Soil & Water Conserv. 6:218.

Gray, D. H., and A. T. Leiser. 1982. Biotechnical slope protection and erosion control. Van Nostrand Reinhold, New York, 271 pp.

Holmes, W. C. 1985. Report on the state of the preliminary investigation of the Toledo Bend Reservoir Shoreline Erosion Control Project. Report submitted to Sabine Soil Conserv. Dist., 10 pp.

Long, W., et al. 1974. Development plan for Toledo Bend. Report submitted to the Sabine River Authority, Planning and Research Associates, Natchitoches, Louisiana, 38 pp.

Rumsey, B. 1984-87. Plottings of daily lake levels of Toledo Bend Reservoir. La. St. Dept. Trans. & Development, Toledo Bend Dam.

Sharp, W. C., and J. Vaden. 1970. Ten year report of sloping techniques used to stabilize tidal river banks. Shore & Beach 38:31-35.

Sharp, W. C., C. R. Belcher, and J. Oyler. 1980. Vegetation for tidal shoreline stabilization in the mid-Atlantic states. USDA-Soil Conserv. Service pamphlet, 19 pp.

Soil Conservation Service. 1973. Soil survey of the Toledo Bend area. SCS, U.S. Dept. Agriculture, Alexandria, Louisiana.

URS/Forrest and Colton. 1979. State of Louisiana Water Quality Management Plan. Basin 11, Sabine River. Final draft report prepared for Louisiana Department of Wildlife and Fisheries, Baton Rouge.

Webb, J. W., and J. D. Dodd. 1983. Wave-protected versus unprotected transplantings on a Texas bay shoreline. Journ. Soil & Water Conserv. 38:363-366.

Zar, J. H. 1984. Biostatistical analysis. Prentice-Hall, Inc., Englewood Cliffs, New Jersey, 718 pp.

ROLE OF LOCAL GOVERNMENTS REGARDING WETLANDS PROTECTION IN MICHIGAN

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ABSTRACT

In Michigan there is a need for a partnership between the Department of Natural Resources (DNR) and the local governmental units. The principal wetland protection vehicle in Michigan is Public Act 203 (Wetland Protection Act) which was promulgated in 1979. P.A. 203 provides the Michigan DNR with permit authority over wetlands larger than five acres in size and those contiguous to lakes and streams. However, the permit process is quite controversial, and wetland violations appear to exceed permitted activities. As of 1 January 1988, only 14 local governmental units had adopted some form of community wetland protection. Unless the local units of governments map, rezone, and protect the MDNR-regulated wetlands, wetland violations and the wetland controversy is expected to continue.

INTRODUCTION

This article addresses the role of local governments in Michigan as regards wetland protection. As of 1 January 1988 only 14 local townships and/or municipalities had wetland protection ordinances or other policy measures. Most of these local governmental measures were enacted after the State of Michigan promulgated Public Act 203 (Wetlands Protection Act) of 1979 which took effect in October 1980. However, there are 83 counties and over 1,500 local governmental units in Michigan. Because the Michigan Department of Natural Resources (MDNR) expends more time on wetland violations than on permit applications, the need for a state-local government partnership in wetland protection is most pressing.

Public Act 203 is a stringent, statewide wetlands protection measure. According to this statute, the MDNR has jurisdiction over all wetlands contiguous to lakes and streams, as well as all wetlands over five acres in size (except in counties under 100,000 persons), wetlands within 500 feet of a lake or stream and within 1000 feet of the Great Lakes, and those with an open-water area greater than one acre. Prior to granting a permit approving even small acreage of regulated wetland displacement, the permit applicant must demonstrate that: 1) the proposed development is wetland dependent, and 2) there is no feasible and prudent alternative in the uplands on site or on other nearby commercially-available parcels.

In 1987, the State of Michigan processed approximately 7,000 wetland permit applications (Hall 1988). Although there are no firm data, it is estimated that two to three times that number of land projects were undertaken which were not accompanied by permits. Wetland violations are continuing to occur on both existing developments and on lands being readied for future development. Today, most developers and engineering firms are aware of the P.A. 203 requirements, but their compliance is not voluntary. Exacerbating this wetland violation problem is the lack of support for wetland protection on the local level. In brief, most communities have not inventoried their wetland resources, nor are their zoning district maps and comprehensive land use plans consistent with the State's P.A. 203.

LOCAL GOVERNMENTS WITH WETLAND ORDINANCES

There were 14 local townships and municipalities with wetland ordinances as of 1 January 1988 (Table 1). Many of these wetland ordinances are coupled with floodplain and lakeshore protection. Most include an official wetlands map (usually based on hydric soils), a standard permit procedure, provisions for a private consultant to ensure accurate wetlands mapping, and a wetlands board or committee to review permit applications. Although several ordinances have weak wetland definitions, many are based on a list of wetland soils and often specify a minimum size of two acres. Only one ordinance provides for a buffer, e.g., West Bloomfield Township of Oakland County. In general, a local wetlands permit is more comprehensive and involves more site plan considerations than a state permit.

Table 1. Local communities in Michigan with a wetlands ordinance.

Local Government	County
Village of Saugatuck	Allegan
Hayes Township	Charlevoix
Burt Township	Cheboygan
City of Novi	Oakland
Independence Township	Oakland
West Bloomfield Township	Oakland
Bloomfield Township	Oakland
City of Lake Angelus	Oakland
City of Wixom	Oakland
Oakland Township	Oakland
Milford Township	Oakland
Manistee Township	Manistee
Charlestown Township	Kalamazoo

Eight of the 14 communities with wetland ordinances are located in Oakland County, the wealthiest and one of the fastest growing counties in the State. Figure 1 demonstrates the spatial relationship between

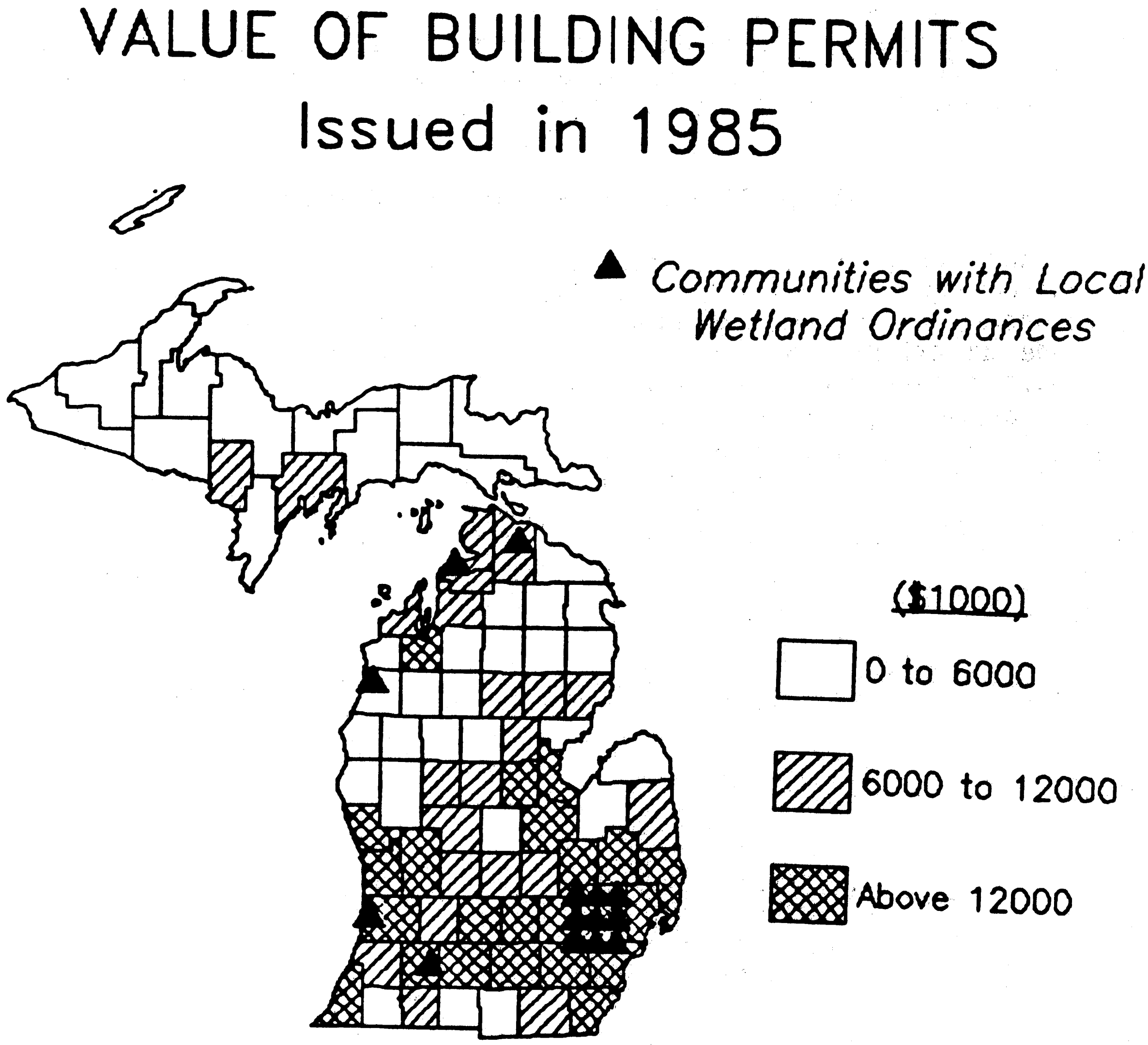

Figure 1. Relationships between counties with large number of building permits and wetland protection measures.

economic growth (i.e., number of building permits) and local communities with wetlands ordinances. In general, communities with local wetland protection measures can be characterized by the following:

- Rapid increase in growth, especially residential and commercial development.

- Near or at sewer capacity.

- Recent experience with a surge in multi-family development, particularly apartments or condominiums.

- Subject to traffic congestion, particularly during rush hour periods.

- Tend to have large lot requirements for residentially-zoned land, e.g., 2.3 acre minimum.

- Predominance of middle and upper income residents.

In contrast, there are two identifiable regions in Michigan which appear to be particularly opposed to local wetland protection. These two regions include the older, somewhat economically-depressed communities of greater Metropolitan Detroit including the down-river area, and northern Michigan where lumbering and mining are important. In southeastern Michigan there are communities with high levels of unemployment, poverty and demand for services, but very limited tax base and economic development potential. Consequently, the existing industrial parks and major transportation nodes are regarded as development centers in spite of the fact these areas may contain wetlands. Macomb County, located north of the City of Detroit, may be involved in more wetland violations than any other county in Michigan. Many county agencies resent having to secure an MDNR permit for drain, highway and utility improvements. In comparison, communities in northern Michigan, particularly those in the Upper Peninsula, generally feel that since historical wetland losses were largely confined to southeastern Michigan, these northern counties should be exempt from P.A. 203.

WETLAND FUNCTIONS AND VALUES

The need for wetland protection is difficult to convey to local governmental officials and developers unless there is an appreciation of wetland functions and values. For example, to replace the waterfowl breeding, fish spawning, and nutrient removal functions of natural wetlands it costs between $7,000 and $28,000 per acre (Jaworski 1980). Moreover, costs of replacing filled regulated wetlands with artificially-created wetlands appear to range between $10,000 and $30,000 per acre. However, many of the wetland values are public values, and as such are not vendable in the marketplace. That is why wetlands are considered part of the "public trust" and are protected by governmental regulations. Moreover, not all wetlands have similar functions and values, nor are the various wetland types equally distributed across the landscape. Forested wetlands, e.g., Alder swamps, are abundant in Michigan, and until 1984 the MDNR was permitting a replacement of only 50% of such lower quality wetlands.

Currently, the State of Michigan is assuming the posture of permitting "no net loss of regulated wetlands." Thus, in addition to justifying the loss of regulated wetlands, the applicant must replace all filled or otherwise developed wetland on at least an acre-for-acre basis. This policy was implemented in 1984 when the U.S. Environmental Protection Agency (U.S. EPA) delegated administration of wetlands dredge and fill permits under Section 404 of the 1977 Clean Water Act to the State of Michigan. Should Michigan fail to satisfy federal reviewers in administering the Section 404 dredge and fill permit program, the U.S. EPA could resume that permit authority.

PROMOTING WETLAND PROTECTION AT THE LOCAL LEVEL

As in the case of groundwater protection, it is at the local level where land use decisions are most often made as well as where zoning and site plan review authority rests. Therefore, to minimize future wetland violations, it is important that the local units of governments "buy in" to wetland protection. Based on our experience working as the wetland consultant for Oakland Township of Oakland County, a number of suggestions are listed below which may promote the state-local government partnership in wetland protection. It is important to realize that under P.A. 203 the MDNR will not grant a wetlands permit if a local wetlands permit has been denied unless there is a larger concern regarding public health and safety.

- Continue to educate the people regarding the function and value of freshwater wetlands. Emphasize the historical loss of wetlands and importance of hydrologic and ecosystem support values, not just wildlife values or the high cost of wetland restoration if cited for a violation.

- Tie in wetland protection with floodplain and shoreline protection as well as with stormwater management where appropriate. Floodplain management has received community support through the FEMA (Federal Emergency Management Act) Program, and most citizens can identify with shoreline and lake protection.

- Make available the National Wetlands Inventory Maps which cover approximately 70% of Michigan, as well as the statewide Natural Resource Inventory Maps which includes 21 counties thus far. Demonstrate that local wetlands maps can be provided to users on a cost-recovery basis, like aerial photographs and topographic maps. On site field investigation of wetlands is required for actual site plan development.

• Permit replacement of regulated wetlands only when justified by a "no feasible and prudent alternative" analysis. Continue the exemption for most lumbering, mining, and agricultural activities which do not involve mass grading and drainage alterations. Unless a wetlands consultant is employed, allow the MDNR to monitor projects with wetland mitigation.

• Encourage the inclusion of cluster options and density discount features in local zoning ordinances so as to facilitate wetland preservation on parcels with 50% or more unbuildable areas including regulated wetlands. Lowering the unbuildable percentage to 30% may be justified in some communities.

• An alternative to a local wetlands ordinance would be a site analysis requirement as part of a site plan ordinance. Developers should be encouraged to perform a site analysis, including a wetlands assessment, prior to submitting a preliminary site plan.

• Encourage communities to reassess real properties that contain wetlands. Reduced property taxes may discourage more intensive development as well as lower the selling price of land. Ad valorem taxation of wetlands at the agricultural rate has been practiced in Canada. However, down-zoning of previously zoned properties may be illegal, and thus difficult to accomplish.

• Provide information on various conservation and open space agreements can be incorporated into local ordinances.

• In economically-depressed counties and townships, attempt to develop community-wide wetland replacement (mitigation) agreements with the MDNR as regards existing industrial parks and large-scale development projects. For example, Wayne County, which includes the City of Detroit, is considering a county-wide plan which provides for the mitigation of wetlands in selected areas in exchange for the replacement/restoration of wetlands in targeted areas.

FUTURE WETLAND PROTECTION IN MICHIGAN

Although unpublicized, there is much controversy over wetland protection, and an estimated 40 court cases against the MDNR are in progress. However, given the Section 404 delegation and the current health of Michigan's economy, changes in P.A. 203 do not appear to be forthcoming. Rather, the Rules of Implementation for P.A. 203, which were finally approved in June 1988, give the MDNR additional jurisdiction over wetlands and has firmed up wetland replacement. At present, the State of Michigan is striving to more evenly administer P.A. 203, especially in areas outside of southeastern Michigan. In general, wetland education is intensifying, with recent focus on realtors and contractors.

Wetland protection, particularly wetland preservation, entails higher costs of new development. By increasing the quantity of unbuildable areas and by incurring the costs of wetland replacement and monitoring, additional costs of development are passed on to the consumer. Such higher costs of new development should render existing developments and older communities more attractive to both land developers and to individual buyers. As a result, more emphasis may have to be placed on restoration or enhancement within existing developments, including previously-impacted wetlands.

In order to convince local governmental officials to enact wetland protection measures and to report wetland violations to the MDNR, there must be strong positive financial incentives. Having the developer establish an escrow account to defray wetland protection costs on a project-by-project basis is an important first step in reducing local costs. However, to be truly effective, it must be demonstrated that wetland protection, and the prevention of expensive wetland restoration, pays real dividends to both the developer and the home community. At present, only the wealthier and rapidly-growing communities in Michigan appear to be perceiving that payback.

The lack of official wetland maps exacerbates wetland protection in Michigan. Because the National Wetlands Inventory Maps were based on high-altitude aerial photography, these maps serve as a general guide only. In Wisconsin, the Department of Natural Resources maps the wetlands and allows the counties a specified amount of time to accept and/or amend the official wetland maps. However, even if official wetland maps were available in Michigan, it still would be exceedingly difficult for the State to administer wetland protection without the partnership of the local governments. Because lumbering and agriculture are exempt from many of the provisions of PA 203, many landowners and developers are clearing and cultivating wetland areas on parcels which were previously in agriculture.

LITERATURE CITED

Hall, Dennis. Personal communication, 23 June 1988. Chief, Wetlands Unit, Land & Water Management Division, Michigan Department of Natural Resources, Lansing, Michigan.

Jaworski, Eugene. 1980. Value of freshwater wetlands in Michigan. Paper at Function and Value of Freshwater Wetlands in the Midwest Conference, June 1980, at Minneapolis, Minnesota. Sponsored by American Water Resources Society, 7 pp.

IMPACTS OF CLEAR CUTTING ON NORTHEASTERN PALUSTRINE FORESTED WETLANDS

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ABSTRACT

Three major palustrine forested broad-leaved deciduous (PF01) wetlands were sampled during the summer of 1987 to assess the vegetational changes associated with the construction and maintenance of transmission line rights-of-way in New Jersey. Forested wetland types sampled were *Acer rubrum*/*Nyssa sylvatica*, *A. rubrum*/*Liquidambar styraciflua*, and *A. Rubrum*/*Fraxinus pennsylvanica* in three different physiographic regions across the state. Detailed comparisons of compositional and structural variables were made between unmanaged forest and disturbed forest within the transmission corridor. Data were compared with previous research in Florida and Massachusetts describing the impacts of right-of-way construction on forested wetlands.

INTRODUCTION

Palustrine forested wetlands are the most abundant and widespread wetland type in New Jersey, comprising 67% of the state's wetland acreage (Tiner 1985). But, because they lack the dramatic expanse of the salt marsh or the distinctive vegetation of riverine wetlands, they are the most easily overlooked. Forested wetlands are mainly found in the floodplains of rivers and perennial streams, although they may form in upland depressions and along the borders of coastal marshes. Such wetland communities are very complex and extremely diverse, varying widely in community composition in response to local edaphic and hydrologic conditions (Tiner 1985).

Wetland protection in New Jersey is strictly enforced through state and federal statutes. Adverse impacts of human activities in these environmentally sensitive communities has grown as the wetland resource has been progressively degraded and lost. As human population pressure increases, it is inevitable that remaining wetland systems will be impacted by human activity. In particular, the siting of transmission line rights-of-way (ROW) cannot practically avoid all wetland systems.

Much interest has been generated about the environmental impacts of the construction and maintenance of transmission line rights-of-way on wetlands. Hypothesized adverse impacts on wooded wetlands include (Belyea 1981):

1) siltation of streams

2) interruption of drainage patterns

3) wide-spread damage from wind-throw

4) raising or lowering of water tables

5) invasion by non-indigenous species.

However, the paucity of conclusive scientific data makes the development of comprehensive environmental assessments for the siting of transmission corridors difficult if not impossible. The generation of such data would enable power-generating utilities to develop and expedite future rights-of-way construction and maintenance programs that will effectively address the concerns of regulatory agencies, conservation groups, and the general public.

STUDY SITE DESCRIPTIONS

Three wetland complexes were selected for study (Figure 1). All sites were palustrine forested broad-leaved deciduous (PFO1) wetlands (classification follows Cowardin et al. 1979) located beneath existing Jersey Central Power & Light Company (JCP&L) transmission lines. All wetland forests studied were associated with the headwaters of perennial streams. The first study site (Turkey Swamp) was located on the Outer Coastal Plain in southeastern Monmouth County along a tributary of the North Branch of the Metedeconk River, while the second site (Old Bridge) was located in southeastern Middlesex County in the northern part of the Inner Coastal Plain of the state. The transmission corridor at Old Bridge had been constructed across the northern part of Burnt Fly Bog, an extensive wooded wetland complex. The gently rolling topography of the floodplains at these sites supported a forest association typical of the *Acer rubrum* (red maple)/*Nyssa sylvatica* (black gum) seasonally flooded hardwood swamp type described by Tiner (1985) from central New Jersey. *A. Rubrum* dominated the wetland at both study sites with *Liquidambar styraciflua* (sweet gum), *N. sylvatica*, and *Magnolia virginiana* (sweetbay) present in the canopy in response to differences in site-specific edaphic factors and land-use history. Very dense shrub layers are typical of these wettest of New Jersey's northern forested wetlands, the soil being saturated for most of the year due to extensive flooding early in the growing season (Tiner 1985). A mixed *Pinus rigida* (pitch pine)/oak--primarily *Quercus alba* (white oak), *Q. rubra* (northern red oak), and *Q. prinus* (chestnut

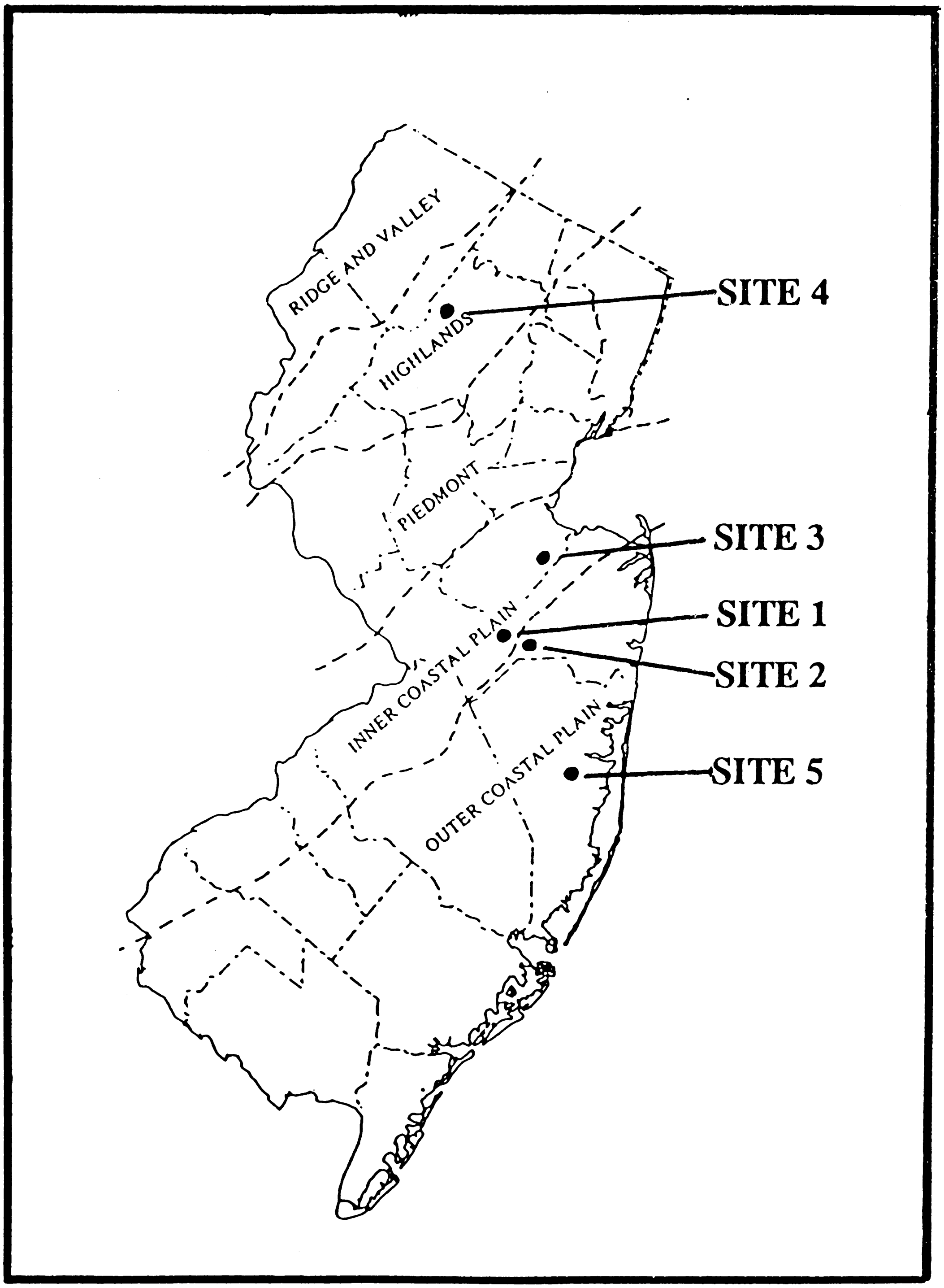

Figure 1. Location of the sampling site in New Jersey.

oak) dominated the adjacent uplands. The northernmost study site (Flanders) was located in the very steeply sloping topography of the Highlands geographic province. The transmission corridor crossed Turkey Brook, a shallow perennial stream cutting a narrowly confined floodplain. The ROW passed parallel to the stream channel and through a mixed seasonally flooded, wetland forest of *A. rubrum* and *Fraxinus pennsylvanica* (green ash). The shrub layer was fairly open. Surrounding upland forest was dominated by *A. saccharum* (sugar maple), *Q. alba*, and *Betula lenta* (black birch).

METHODS

Five (5) 10 m x 10 m quadrats were established randomly within the ROW zone at each study site. Five similar quadrats were established in the unmanaged forest upstream and downstream of the ROW, at least 30 m away from the edge of the ROW to reduce the edge effect of clearing on the adjacent wooded swamp. All quadrats were separated by at least 10 m from the next nearest quadrat. Within each quadrat all trees >3.0 m in height were recorded to species, diameter at breast height and crown cover using a vertical projection of the tree's canopy to a nylon tape stretched perpendicular to the widest diameter of the crown (Harms et al. 1981). Two 5 m x 5 m plots were established within each quadrat by randomly selecting 2 quadrat corners. Within each subquadrat all woody vegetation between 1.0 and 3.0 m in height was recorded by species, number of stems, number of clumps, height and canopy cover. At each corner of the 10 m x 10 m quadrat, one 1 m x 1 m plot was established. All plants <1.0 m in height were recorded by species, number of stems, mean stem height and percent plot coverage in the subquadrat (Daubenmire 1959).

Several indices of community composition were calculated from vegetation data collected in sampling quadrats. Species diversity implies both the number of species and the number of individuals of each species found in a given ecological community (Smith 1980). Community diversity in both the herbaceous and shrub communities in all quadrats was estimated using the index developed by Shannon and Wiener (1963):

$$H' = - [(n/N) \log(n/N)]$$

where H' = community diversity
n = total stems of a given species
N = combined total stems of all species
log = natural logarithm

Diversity is greatest if each individual belongs to a different species and least if all individuals belong to one species.

The diversity index takes into account both species richness (the number of species present in the community) and the evenness with which individuals are distributed among species. These components were explicitly separated to better understand the effects of ROW management on the wetland community. Evenness was measured using an index proposed by Shannon and Wiener:

$$e = H'/\log S$$

where e = community evenness
H' = community diversity
S = total number of species present

The evenness index drops as more individuals are concentrated in fewer species. Species richness was measured using Margalef's (1958) index:

$$d = S-1/\log N$$

where d = community richness
S = total number of species present
N = total number of individuals

The value of the index is zero when only one species is present in the community and increases with the addition of other species. Nickerson and Thibodeau (1984) suggested that the index was sensitive to the clumped growth form of many wetland shrubs, in that N increases faster when clumping occurs.

Separate analyses of variance (ANOVA) were calculated for each measured variable and each calculated index. Site means were compared using Duncan's new multiple range test (Zar 1984). Comparisons between the ROW and forest community variables measured or calculated at each site were conducted using one-tailed t-tests. All analyses were conducted using SAS computer packages (SAS 1985).

RESULTS

Analysis of Variance

After community structural and compositional indices were calculated for the herbaceous and shrub communities in all quadrats (wooded and right-of-way) for all five sites, a two-way analysis of variance (ANOVA) for balanced designs was employed to determine whether the indices varied significantly between study sites. In the case of every variable, with the exceptions of total shrub cover and herbaceous stem density, ANOVA demonstrated a significant ($p<.01$) site effect. These results were not surprising in that the study sites were selected from three different physiographic provinces across the state and were found to differ appreciably in species composition. Significant differences in community indices were also detected between managed (ROW) and unmanaged quadrats and there were also significant interaction effects between management and site, indicating that the communities within the powerline ROWS were significantly different ($p<.01$) in both composition (diversity, evenness and richness) and structure (total cover, density) from adjacent unmanaged communities.

Unmanaged Quadrats

Unmanaged forest areas tended to exhibit highly diverse (mean H' = 1.43) canopy layers. Canopy closure was generally moderate to high, comprising some 56.9% total cover at Turkey Swamp, but complete canopy closure at the Old Bridge and Flanders study sites. Consequently, shrub layers were fairly open (mean total shrub cover on unmanaged quadrats = 46.9%). Shrub layers at Turkey Swamp and Old Bridge were very diverse, comprising some 13 and 14 recorded species per quadrat, respectively. At Flanders, the shrub layers of unmanaged quadrats were dominated by *Lindera benzoin* (spicebush) (mean relative cover = 55.1%), with no other species achieving more than 5% relative cover on any quadrat. Herbaceous layers in the unmanaged wetland areas tended to be composed of large numbers of species, but were typically dominated by woody species which were also well represented in the shrub and/or canopy layers. At Old Bridge, an average of 68.4% of all species recorded in the herbaceous layers of unmanaged quadrats were also found in the shrub layer. The dominant herbaceous layer species at Turkey Swamp was *Clethra alnifolia* (sweet pepperbush) (comprising a mean relative cover of 37.2% in sampled quadrats) which also accounted for an average of 36.0% of the total shrub cover on these quadrats.

Row Quadrats

No canopy species were recorded in the ROW quadrats at the Turkey Swamp study site. Mean shrub height (mean = 1.38 ± 0.05 m) was significantly shorter than that in the unmanaged plots (Table 1). Mean shrub layer cover was much less than that in the wooded quadrats. The shrub layer was dominated (mean relative cover = 93.2%) by *A. rubrum*, which was primarily represented by stump sprouts. In all, only 7 species were recorded in the shrub layer of managed quadrats. *L. styraciflua* stump sprouts accounted for an average relative cover of 32.5%, while *C. alnifolia* occurred at a density not significantly different from that in wooded plots. In contrast to the unmanaged quadrats, *Gaylussacia* spp. (huckleberry) figured prominently in managed quadrats, *Vaccinium corymbosum* (highbush blueberry) was not recorded in any ROW quadrat and shrub species diversity was much reduced (mean H' = 0.65). Herbaceous cover in ROW quadrats was significantly greater and more diverse than in the wooded zones. Some 40 species were recorded (mean = 13.4 species/quadrat). Dominant species were representative of marshy conditions (e.g., *Carex bullata* (button sedge), *Thelypteris thelypteroides* (marsh fern), *Sphagnum* spp.). Only 5 species (12.5%) were also found in the shrub or canopy layers. In general, the ROW community was a dense cover of *C. bullata*, in association with a wide array of freshwater marsh species. Scattered pools and inundated areas supported *Hypericum virginicum* (marsh St. Johnswort), *Phalaris arundinacea* (reed canarygrass), and *Eriophorum virginicum* (cottongrass). The shrub layer consisted of widely-scattered stump-sprouting *A. rubrum* and

L. styraciflua.

Table 1. Results of t-tests comparing shrub and herbaceous layer community composition and structure between unmanaged and ROW quadrats at Site 1: Turkey Swamp (values are mean ± SE).

	ROW (N=10)	Unmanaged (N=10)
SHRUB LAYER		
Total Species	2.70 ± 0.42	5.50 ± 0.56
Shrub Cover (%)	7.82 ± 1.53	41.57 ± 5.35**
Total Stems	31.10 ± 6.27	77.70 ± 7.38**
Stem Height (m)	1.38 ± 0.05	1.81 ± 0.06**
Density (stems/sq.m)	0.62 ± 0.13	1.55 ± 0.15**
Diversity	0.65 ± 0.16	1.25 ± 0.11**
Evenness	0.58 ± 0.12	0.75 ± 0.04
Richness	0.54 ± 0.14	1.06 ± 0.14*
HERBACEOUS LAYER		
Total Species	13.40 ± 0.94	6.70 ± 0.52**
Herb Cover (%)	111.30 ± 4.69	55.90 ± 7.70**
Total Stems	151.10 ± 25.57	50.90 ± 7.46**
Density (stems/sq.m)	37.78 ± 6.39	12.73 ± 1.87**
Diversity	1.62 ± 0.13	1.14 ± 0.10*
Evenness	0.62 ± 0.04	0.60 ± 0.04
Richness	2.57 ± 0.23	1.55 ± 0.19

*p < 0.05
**p < 0.01

Canopy layer species were absent from the ROW quadrats at the Old Bridge study site. Mean shrub height in managed plots was significantly lower than that in wooded plots. Total shrub cover was also significantly lower in managed plots, as was total number of recorded species (Table 2). In general, shrub layer structure and community composition were not significantly different between wooded and ROW quadrats. The shrub layer in managed quadrats tended to be dominated by V. corymbosum (mean relative cover = 42.9%), C. alnifolia (14.7%), and A. rubrum stump sprouts (10.8%). In contrast to the unmanaged quadrats, relative cover values for V. corymbosum and A. rubrum were higher in the ROW quadrats and relative cover of C. alnifolia was reduced. Herbaceous cover in managed plots was significantly greater than in wooded plots. Nineteen species were recorded (mean = 10.2 species/quadrat), of which only five were also represented in the shrub layer. Osmunda cinnamomea (cinnamon fern), Sphagnum spp., and C. bullata were the dominant herbaceous species. The ROW community tended to be a mixed association of C. bullata and O. cinnamomea, featuring H.

virginicum, *H. prolificum* (shrubby St. Johnswort), and *V. corymbosum*, interrupted by hummocks supporting *V. corymbosum* and *C. alnifolia*. Flooded areas supported *E. virginicum*, *Eleocharis palustris* (creeping spikerush), and *Juncus canadensis* (Canada rush), whereas drier areas were monocultures of *O. cinnamomea*.

Table 2. Results of t-tests comparing shrub and herbaceous layer community composition and structure between unmanaged and ROW quadrats at Site 2: Old Bridge (values are mean ± SE).

	ROW (N=5)	Unmanaged (N=10)
SHRUB LAYER		
Total Species	5.00 ± 0.55	6.40 ± 0.31*
Shrub Cover (%)	0.17 ± 0.03	0.47 ± 0.08**
Total Stems	60.20 ± 16.89	77.40 ± 12.84
Stem Height (m)	1.50 ± 0.04	1.80 ± 0.06**
Density (stems/sq.m)	1.20 ± 0.34	1.55 ± 0.26
Diversity	1.19 ± 0.12	1.34 ± 0.07
Evenness	0.75 ± 0.05	0.72 ± 0.03
Richness	1.01 ± 0.13	1.30 ± 0.08
HERBACEOUS LAYER		
Total Species	10.20 ± 0.73	6.10 ± 0.50**
Herb Cover (%)	133.60 ± 17.02	82.48 ± 9.14*
Total Stems	121.40 ± 7.54	89.60 ± 7.08*
Density (stems/sq.m)	30.35 ± 1.89	22.40 ± 1.77*
Diversity	1.20 ± 0.14	1.16 ± 0.08
Evenness	0.52 ± 0.05	0.65 ± 0.03*
Richness	1.92 ± 0.15	1.15 ± 0.12**

*$p < 0.05$
**$p < 0.01$

No canopy species were recorded in the managed quadrats at the Flanders study site. Average shrub height in the ROW was significantly less than that in the wooded quadrats ($p<.01$). Mean total shrub cover was significantly less in the managed quadrats, although mean number of stems and stem density were not (Table 3). The shrub layer was dominated by three species: *A. rubrum*, *Cornus foemina* (gray dogwood), and *Rosa palustris* (swamp rose). The dominant shrubs tended to be unclumped, single stems with very little foliage. Thirteen shrub species were recorded in managed quadrats, five of which were also found in the wooded zones. In contrast to the unmanaged quadrats, mean relative cover values for *V. corymbosum* and *Ilex verticillata* (winterberry) were markedly reduced and *A. rubrum* replaced *L. benzoin* as the most abundant shrubby species. In general, woody species more evocative of shrub swamp conditions [C. foemina, C. amomum (silky dogwood), and Sambucus spp.] were most abundant in the managed quadrats, while drier site shrubs [Viburnum dentatum (southern arrowwood), L. benzoin, and Lyonia ligustrina (maleberry)] were found in the wooded plots. Herbaceous cover in the ROW quadrats (mean total cover = 100.0%) was significantly greater than that recorded in the wooded plots ($p<.01$). None of the other structural variables, however, were different from those in the unmanaged quadrats. Species diversity was lower and species evenness higher in ROW quadrats, reflecting the fact that the herbaceous community in those plots tended to be dominated by Carex stricta (tussock sedge) (mean relative cover = 42.5%) and that, of 31 other herbaceous species recorded, only two (6.5%) comprised more than 3% of the total herbaceous cover. These two species, T. thelypteroides and Onoclea sensibilis (sensitive fern), were also represented in wooded plots, although the former was almost twice as abundant in the unmanaged plots. Only ten herbaceous species recorded in ROW quadrats (31.3%), generally freshwater marsh species such as Polygonum sagittatum (arrow-leaved tearthumb) and Typha latifolia (broad-leaved cattail), were unique to the managed zone. Overall, the ROW community tended to be a C. stricta/C. amomum shrub swamp established to either side of the stream which paralleled the transmission lines.

Table 3. Results of t-tests comparing shrub and herbaceous layer community composition and structure between unmanaged and ROW quadrats at Site 3: Flanders (values are mean SE).

	ROW (N=5)		Unmanaged (N=10)	
SHRUB LAYER				
Total Species	4.60	1.29	4.00	0.49
Shrub Cover (%)	15.40	6.78	52.28	7.27**
Total Stems	55.20	23.49	33.50	5.02
Stem Height (m)	1.61	0.04	1.90	0.06**
Density (stems/sq.m)	1.10	0.47	0.67	0.10
Diversity	0.97	0.09	1.07	0.14
Evenness	0.77	0.09	0.78	0.06
Richness	0.89	0.22	0.88	0.13
HERBACEOUS LAYER				
Total Species	15.60	1.02	12.20	1.34
Herb Cover (%)	100.00	13.76	67.03	10.56**
Total Stems	74.20	8.27	51.30	10.68
Density (stems/sq.m)	18.55	2.07	12.83	2.67
Diversity	1.12	0.09	1.54	0.17*
Evenness	0.77	0.02	0.62	0.05*
Richness	3.40	0.19	3.03	0.33

*$p < 0.05$
**$p < 0.01$

DISCUSSION

Although the wetland habitats sampled at the three study sites were initially very different in species composition, there were some trends in the data recorded in the ROW wetland communities at all study sites. The shrub layer in the ROW at all sites tended to be less diverse and less well-developed, comprising fewer species and sparser overall quadrat coverage, relative to the adjacent unmanaged communities. The shrub layer in the unmanaged quadrats was generally highly diverse, featuring a large number of species, but with few species achieving real dominance, as is typical of hardwood swamp communities (Tiner 1985; Robichaud & Buell 1973). Right-of-way shrub communities were characterized by many species, poorly represented--although many species not in the wooded plots were established in the ROW--and one or a few species making up most of the shrub cover.

Dominant woody species in the ROW were generally species such as *A. rubrum* and *L. styraciflua*, prominently featured in undisturbed wetland canopies, and growing as stump sprouts and suckers from the cut stumps of ROW trees remaining since the previous corridor maintenance. *C. alnifolia* was dominant in the shrub layer at most study sites, accounting for much of the total shrub cover under a wide range of habitat conditions. Other species, notably *V. corymbosum* and *I. verticillata* tended to be reduced in abundance on managed quadrats. Thibodeau and Nickerson (1986) found *V. corymbosum* to be slow to re-establish after ROW construction in a Massachusetts hardwood swamp. Typically, in this study woody vegetation in ROW wetland communities was confined to hummocks and higher micro-elevations along the edges of the managed zone and access roads. In particular, *C. alnifolia* and many ericaceous shrubs formed dense mixed tickets with *Rhus copallina* (winged sumac) and *Rubus* spp. at the base of transmission tower footings. At Turkey Swamp (Site 1), in particular, the filling around the bases of towers interrupted the prevailing vegetation patterns and small islands of generally more mesic vegetation were established within what was otherwise a flooded shrub swamp community.

The herbaceous communities in the managed areas demonstrated marked divergence from the adjacent wooded plots. The ROW communities consisted of significantly higher total numbers of constituent species, were more highly developed with significantly greater total cover values, and were significantly more diverse than unmanaged communities. The unmanaged herbaceous communities tended to be dominated, sometimes exclusively, by the same species which dominated the shrub layer. That is, the unmanaged herbaceous communities appeared to represent the recapitulation of the corresponding shrub canopy. In contrast, the majority of species in the ROW herbaceous layer were unique to the ROW, were not woody species, and were generally present in low densities. Those herbaceous species which dominated managed quadrats, notably *C. stricta* and *O. cinnamomea*, formed dense mats of vegetation which prevented the establishment of other species. Shallow root systems of the shrubs and trees, as well as closed, multi-tiered canopies in adjacent undisturbed forests effectively discouraged the establishment of well-developed herbaceous layers. The effectiveness of this competition can be seen at Flanders (Site 3): where canopy cover was reduced there was a lush growth of O. cinnamomea and T. thelypteroides.

The severity of any adverse impacts generally depends on an array of site-specific factors including wetland type, size of the ROW, and the construction and maintenance techniques employed (Cutlip 1986). Wooded wetland systems tend to be subject to much greater initial disturbance because maturing vegetation of tree height poses serious direct threats to the transmission lines. Cutlip (1986) showed that transmission line ROW construction in a Florida bottomland forest resulted in appreciable changes in species composition within the ROW due to the fact that much of the potentially hazardous woody vegetation was completely removed. Conversely, similar construction in a herbaceous/shrub wetland resulted in very little initial change in species composition between ROW plots and controls because the dominant vegetation necessitated minimal clearing and maintenance.

There is some evidence to suggest that different wetland types under different edaphic and environmental constraints respond differently to ROW management practices. In general, the most dramatic changes in the wetland community occurred at Turkey Swamp. Changes at Old Bridge appeared to be somewhat less severe. The shrub layer in the unmanaged forest tended to be poorly developed as a result of a relatively closed canopy, so that reduction in shrub cover in the adjacent managed wetland zone resulted in no statistically detectable differences between managed and unmanaged quadrats. Very high topographic relief and the restricted nature of the stream channel at Flanders probably accounted for the relatively low levels of impact recorded there. Undetermined hydrologic and elevational differences resulted in the establishment of a wide array of shrub species, none comprising large cover values. Unlike the other study sites, the ROW corridor at Flanders paralleled the stream channel, did not interrupt the prevailing hydrology of the site, and caused none of the impounding of water which allowed the establishment of a diverse array of wetland emergents at the other sites.

Apparently, heavy equipment operation in the ROW during construction had resulted in the formation of large ponded areas. Construction of access roads across the creek channels, especially evident at Turkey Swamp, also had altered stream flow such that water > 12 in. in some places was held on much of the ROW community. Study sites were visited only a limited number of times, but sampling was conducted late in the growing season, suggesting that water stands on these sites throughout the year. Construction of an access road across a powerline ROW in a Florida bottomland forest interrupted the natural hydrologic regime (Cutlip 1986). As a result, water was impounded upstream from the road which delayed vegetation recovery by preventing seed germination. When swales were provided to restore through flow, increased growth rates rapidly equilibrated with downstream recovery. Similarly, a gravel access road constructed in eastern Massachusetts blocked water flow across a shrub swamp (primarily V. corymbosum) (Thibodeau & Nickerson

1985). Within one year the vegetation in the drained area below the road shifted substantially toward a denser and more diverse assemblage of species, while species numbers declined in the flooded area as once dominant species were extirpated. Conversely, borrow pits left from the construction of an access road across a Florida shrub swamp continued to hold water during dry seasons, providing needed habitat for fishes, amphibians, and wading birds (Cutlip 1986).

Flooded conditions were common to all ROW study sites and the species which established in these conditions, including Scirpus cyperinus (woolgrass), Aster nemoralis (purple-stemmed aster), and Dulichium arundinaceum (three-way sedge) were widespread across sites. Flooding, in addition to fostering the dense growth of herbaceous species, seems to prevent the establishment of woody species except on hummocks. Fast-growing species such as Rubus spp. and C. alnifolia have the advantage and other species which do succeed in establishing are slow to spread. Also, suckering species such as A. rubrum which can tolerate shallow flooding (Harms et al. 1980) gain a head start and ultimately dominate.

In the Florida study already discussed (Cutlip 1986), where the effects of the access road were removed, wooded swamp vegetation cleared during ROW construction was returning toward dominance by native wetland species by the second growing season after construction. The construction of 345 kV transmission lines through a mixed-species wooded wetland in eastern Massachusetts required the removal of all above-ground vegetation (Thibodeau & Nickerson 1986). By the end of the first growing season after clearing, community composition in managed plots was similar to undisturbed plots. Plots located in an adjacent ROW cleared in 1936 differed from control plots in having no tree species and having a more diverse shrub layer with V. corymbosum replacing C. alnifolia, R. viscosum and I. verticillata as the dominant shrub species. The authors concluded that the forested wetland was able to substantially recover from a single catastrophic perturbation within two growing seasons.

In general, ROW construction and maintenance activities at three New Jersey study sites appear to have created highly diverse, very complex assemblages of wetland species. Managed corridors apparently have not returned to original species composition and appear to have succeeded to structurally dissimilar wetland types. Previous studies have reported complete or nearly complete recovery after ROW construction of disturbed wetland vegetation in shrub swamps (Nickerson & Thibodeau 1984) and hardwood swamps (Belyea 1981; Cutlip 1986; Thibodeau & Nickerson 1986) within one or as little as five growing seasons after disturbance. In all cases, original species composition was restored naturally and invasion of disturbed sites by species not found in the adjacent undisturbed wetlands was minimal. In contrast, successional processes have not returned managed areas to the original species composition or structure at the three wetland study sites sampled in this study. With only one season's data, it is unclear how dependent these changes in vegetation are on continual ROW maintenance activities. The future persistance and stability of these assemblages is not yet known. Future sampling, in the absence of maintenance activity, is needed to judge the persistence of the created wetland systems and their susceptibility to invasion by undesirable canopy species.

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LITERATURE CITED

Belyea, D. W. 1981. Environmental monitoring of the construction of an EHV transmission line through a wooded swamp. In R. E. Tillman (Ed.), Proc. Second Symp. on Env. Concerns in Rights-of-Way Mgmt, University of Michigan, Ann Arbor, pp. 32-1 to 32-10.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U.S. Fish and Wildlife Service, FWS/OBS-79-31, Office of Biological Services, Washington, D.C.

Cutlip, C. G. 1986. The ecological impact of three power line corridors located in wetland systems of Florida. Bio-Scan, Inc., Lehigh, Florida, 84 pp.

Daubenmire, R. 1959. A canopy-coverage method of vegetation analysis. Northw. Sci. 33:43-64.

Harms, W. R., H. T. Schreuder, D. H. Hook, C. L. Brown, and F. W. Shropshire. 1980. The effects of flooding on the swamp forest in Lake Ocklawaha, Florida. Ecology 61:1412-1421.

Hays, R. L., C. Summers, and W. Seitz. 1981. Estimating wildlife habitat variables. U.S. Fish and Wildl. Serv., FWS/OBS-81/47, 111 pp.

Margalef, R. 1958. Information theory in ecology. Gen. Systems Bull. 31:36-71.

Nickerson, N. H., and F. R. Thibodeau. 1984a. The effect of power utility rights-of-way on wetlands in eastern Massachusetts. The New England Power Company, Westboro, Massachusetts, 40 pp. + appendices.

Robichaud, B., and M. F. Buell. 1973. Vegetation of New Jersey. A study of landscape diversity. Rutgers Univ. Press, New Brunswick, New Jersey, 340 pp.

Shannon, C. E., and W. Wiener. 1963. The mathematical theory of communication. University of Illinois Press, Urbana, Illinois.

Smith, R. L. 1980. Ecology and field biology. Harper & Row Publishers, Inc., New York, New York, 835 pp.

SAS Institute, Inc. 1985. SAS user's guide: Statistics. Version 5 Edition. SAS Institute, Inc., Cary, North Carolina, 956 pp.

Thibodeau, F. R., and N. H. Nickerson. 1985. Changes in a wetland plant association induced by impoundment and draining. Biol. Conserv. 33:269-279.

__________. 1986. Impact of power utility rights-of-way on wooded wetland. Env. Manage. 10:809-814.

Tiner, R. W., Jr. 1985. Wetlands of New Jersey. U.S. Fish and Wildlife Service, National Wetlands Inventory, Newton Corner, Massachusetts, 117 pp.

Zar, J. H. 1974. Biostatistical analysis. 2nd Ed. Prentice-Hall, Englewood Cliffs, New Jersey, 620 pp.

MANAGEMENT OF GEORGIA'S MARSHLANDS UNDER THE COASTAL MARSHLANDS PROTECTION ACT OF 1970

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ABSTRACT

The Coastal Marshlands Protection Act was passed by the Georgia General Assembly in 1970 to create an administrative review process for those activities that would convert the coastal marshes to nonmarshland uses. Since its passage, the Act has not been substantively amended. A study of the coastal marshland protection efforts under this Act was funded by the General Assembly to determine the effectiveness of the marshlands protection program and to ascertain if amendments to the Act were called for. Results indicate that the Act has been effective in protecting the coastal marshes of Georgia; less than 450 acres of marsh have been converted under the Act since its passage. Although amending the Act was not felt to be necessary, a variety of recommended changes were identified including: (1) the incorporation of "vital areas" language in the law; (2) refining the definitions; (3) updating procedural language; and (4) specifically addressing those activities with potentially major impacts on the marshes such as marinas, roads, and impoundments. Weaknesses of the law include the exemption of major marsh converting activities and the lack of a buffer zone around the marshes.

INTRODUCTION

Extensive documentation exists for marsh ecosystem values related to biological productivity, assimilating pollutants, minimizing storm impacts, decreasing erosion damage, and aesthetics.[1] Recognition of these values led to the adoption of protective measures for the Georgia marshlands with the passage of the Coastal Marshlands Protection Act of 1970.

Historic Perspective

In July 1968, Kerr-McGee Corporation submitted a bid to lease 25,000 acres of Georgia's offshore land for phosphate mining. The corporation also planned to use dredge from the mining operation to fill and develop 20 square miles of "high grounds" on Little Tybee and Cabbage Islands. Considerable public opposition ensued, resulting in the eventual expiration of the Kerr-McGee application. It was evident, however, that if the coastal marshes were to be protected, additional legislation would be necessary. During the 1969 session of the Georgia

General Assembly, legislation was introduced to protect the coastal marshlands. The bill, in its final form, called for the creation of the Coastal Marshlands Protection Agency. The seven member Agency was an autonomous division of the State Game and Fish Commission with the duty to administer the Act. The Agency was responsible for protecting the coastal marshes by regulating marsh-altering development by means of a permitting system. Applications for a marshland permit were filed with the State Game and Fish Commission. The Commission then reported on an application and sent it to the Agency for a determination. Before granting a permit, the Agency had to determine that the alteration was not contrary to the public interest.

Implementing the requirements of the Coastal Marshlands Protection Act took time. In the first two years after the passage of the law, concern focused on the effectiveness of the Agency. Marsh-altering activities were still occurring; State Game and Fish rangers reported violations, but the Agency had not taken any court action.[2] As indicated below, over half of the marshland acreage converted under permit since the passage of the Coastal Marshlands Protection Act occurred during the first two years of the program.

In 1972, the Executive Reorganization Act reassigned the duties of a variety of departments, agencies, boards, committees, and commissions, including the Coastal Marshlands Protection Agency's authority to conduct hearings and to prosecute court actions, to the Department of Natural Resources.[3] In 1978, the Coastal Resources Division was established within DNR to carry out the Department's coastal activities. With the consolidation of activities, the hiring of a professional staff, the reduction in Committee composition from seven to three members, and the provision of enforcement power to DNR, the Act's implementation became more effective. The Committee is currently composed of the Commissioner of the Department of Natural Resources and two persons selected by the Board of Natural Resources.[4] The Marsh and Beach staff is headquartered with other DNR personnel in Brunswick.

The Coastal Marshlands Protection Act

The Coastal Marshlands Protection Act of 1970 prohibits any alteration of salt or brackish marsh without a permit.[5] The Act encompasses all salt or brackish water marsh lying within the estuarine area including all lands that are tidally influenced and within the tide-elevation range of 5.6 feet above mean tide level.[6] To determine if a marsh exists, the Act has two additional tests. The first is the vegetation test which identifies three species of plants as indicators of coastal marshes: *Spartina alterniflora*, *Juncus gerardi*, or *Iva frutescens* var. *oraria*. The second is the soil test which bases marsh determination on the existence of salt marsh peat.[7] Any wetland located within the estuarine area meeting either of these tests is subject to the Act's permit requirements.

There are six exemptions to the Act's permit requirement:

1. activities of the Department of Transportation incident to construction, repairing, and maintaining a public road system in Georgia;

2. agencies of the United States charged by law with the responsibility of keeping the rivers and harbors of this state open for navigation, and agencies of this state charged by existing law with the responsibility of keeping the rivers and harbors of this state open for navigation including areas for utilization for spoilage designated by such agencies;

3. activities of public utility companies regulated by the Public Service Commission incident to constructing, erecting, repairing, and maintaining utility lines for the transmission of gas, electricity, or telephone messages;

4. activities of companies in construction, erecting, repairing, and maintaining railroad lines and bridges;

5. activities of political subdivisions incident to constructing, repairing, and maintaining pipelines for the transport of water and sewage; or

6. the building of private docks on pilings, the walkways of which are above the marsh grass not obstructing tidal flow, by the owners of residences located on high land adjoining such docks.

Any other alterations of the marshlands requires a permit.[8]

The Act creates the Coastal Marshlands Protection Committee. The Committee issues all orders including those granting, denying, or revoking a permit. Any person who is aggrieved or adversely affected by the Committee's decision may petition for an administrative hearing. Only after the administrative law judge makes a final decision is there a right of judicial review.[9]

The Committee, in making its determination on whether to grant a permit or not, must consider the public interest. The Act specifically outlines the areas of public concern to be considered:[10]

1. whether or not any unreasonably harmful obstruction to or alteration of the natural flow of navigational water within the affected area will arise as a result of the proposal;

2. whether or not unreasonably harmful or increased erosion, shoaling of channels, or stagnant areas of water will be created to such extent as to be contrary to the public interest; and

3. whether or not the granting of a permit and the completion of the applicant's proposal will unreasonably interfere with the conservation of fish, shrimp, oysters, crabs, clams, or any other marine life, or any wildlife, or any other natural resources, including but not

limited to water and oxygen supply, to such an extent as to be contrary to the public interest.

The Committee may condition the approval of any permit application in order to protect the public interest.[11] If it is determined a use is not contrary to the public interest, then a permit must be granted. The Committee's primary duty is the balancing of the public interest against private interests.

The Committee and DNR have enforcement authority under the Act. The Committee is authorized to issue cease and desist orders or any other appropriate order to any person altering the marsh without a permit. This authority does not extend to those activities exempted from the Act's permit requirements. Also, when the Committee determines that a violation is occurring, it can petition the courts for an injunction. If an administrative hearing determines that a violation exists, then the Committee can impose a civil penalty of up to $1,000 for each violation. Additionally, the Committee can order a separate fine of $500 a day for each day a violation continues. The Committee, as well as any party to the proceedings, may appeal any administrative law judge's adverse decision to superior court. The Act permits the Committee to use any one of these actions or any combination of these in enforcing the law.[12]

The Department of Natural Resources is authorized to hold hearings to determine if a violation has occurred. If DNR finds a violation, it can seek judicial prosecution.[13] The Department of Natural Resources is also responsible for administering the Act. To ensure compliance with the Act and permits issued under it, DNR has a legal duty to make reasonable inspections of the marshlands.[14]

STUDY SITE

The coastal marshland area of Georgia encompasses fresh, brackish, and saltwater wetlands and open water lying between the barrier islands and the mainland. Johnson and his associates[15] estimated the coastal marshland system to include 393,000 acres, of which 286,000 acres were covered by Spartina. According to an analysis of LANDSAT data by the Department of Natural Resources, however, coastal wetlands and estuaries amount to 626,921 acres.[16]

The Soil Conservation Service (SCS) analysis of coastal marshlands is presented in Table 1. The SCS estimates that coastal Georgia includes 410,788 acres of tidal and nontidal marshes, 175,500 acres of tidal water, and 9,487 acres of dredged borrow pits and filled spoil sites for a total of 595,775 acres. The best indication of marshlands conversion trends available from these data is the acreage figures for borrow pits and spoil sites (9,487 acres) which most likely were marshlands but have been dredged and filled and no longer support marsh life. Thus, dredge and fill activities account for the conversion of a minimum of 10,000 acres of marsh to date. In addition, some portion of the open water is converted marsh but it was beyond the scope of this study to determine this acreage.

Table 1. Coastal marshland acreage by county*.

County	Tidal Marsh	Nontidal Marsh	Dredge Tidal Water	Spoil Sites	Borrow Pits
Bryan	22,624	59	8,520	217	217
Camden	83,915	2,392	36,602	1,749	366
Chatham	99,571	168	48,955	2,431	573
Effingham			455		247
Glynn	71,708	20	29,474	2,333	119
Liberty	45,852	158	16,991	761	109
Long					178
McIntosh	81,613	2,708	35,503	128	59
Total	405,283	5,505	175,500	7,619	1,868

*USDA Soil Conservation Service Land Use Inventory.

MATERIALS AND METHODS

This study included a legal, historical, and scientific literature review related to the coastal marshes of Georgia. In addition, interviews were conducted with Coastal Marshland Protection Committee members, DNR staff, Justice Department staff, academics, and members of interest groups. DNR files were reviewed to ascertain the number of permits issued, denied, or held in abeyance and to quantify acreage of marshland converted. To help determine the potential for increased aquaculture along the Georgia coast, a site visit was made to the Waddell Mariculture Center in Bluffton, South Carolina.

RESULTS

Since the passage of the Coastal Marshlands Protection Act in 1970, the Coastal Marshlands Committee has received 248 permit applications for marsh-altering activities, of which 202 permits (81.5%) were granted. Between 1972, when the program became fully implemented, and 1986, the last full year of records, the Committee issued an average of 12.6 permits per year. The Committee has denied 29 permits (11.7%) in the 17 years of the program for an average of less than 2.0 denials per year. Department of Natural Resources records also indicate that 19 permit applications (6.8%) have been held in abeyance for various reasons.

Although total acreage figures for marshes converted since 1970 are not available, information obtained from DNR suggests that less than 450 acres of marsh have been altered under the permitting program. Over half of these 450 acres (256 acres) were converted during the first two years following the passage of the legislation. This does not include acreage converted by those activities exempted from the permit requirement. Most of the marsh acreage converted to other uses during the 17 years since the passage of the Act were dredged and filled under the direction of the Corps of Engineers (COE) for navigation purposes and the Department of Transportation for highway construction. Construction of Interstate 95, for example, resulted in the conversion of 3,976 acres of wetlands.[17] Total acreage converted by exempted activities is not known.

These figures do not indicate the effect that DNR technical staff have on marsh protection by working with permit applicants to minimize the impact of proposed activities on the coastal marshes. Prior to an applicant's submitting an application, the Marsh and Beach staff conducts site visits and makes recommendations to the applicant on what steps should be taken to increase the probability that state and federal permits will be issued. Some activities, which would have such impacts that obtaining permits is not likely, are not pursued further. Other activities may be moved to higher ground or altered in various ways to mitigate their impacts on the marshes and increase the likelihood that permits will be issued by DNR and the COE. Still other activities are proposed for locations which are not under the jurisdiction of the Coastal Marshlands Protection Committee and, therefore, do not require a state permit. For the three years for which records were available, the Marsh and Beach staff made a total of 801 site inspections for an average of 267 per year. Recent estimates indicate that the staff may make between 500 and 1,000 site visits each year.[18] The value of early site visits is apparent in the relatively high percentage (81.5%) of permits granted by the Coastal Marshlands Protection Committee with a minimum conversion of marsh acreage.

DISCUSSION

The Coastal Marshlands Protection Act appears to be an effective mechanism for protecting coastal marshes while allowing usage of the marshes that are not contrary to the public interest. Although amending the Act is not currently necessary, recommendations were designed to: (1) enable the law to better address concerns that have appeared or increased since the original bill was enacted, (2) incorporate constitutional changes and judicial decisions that have occurred since the passage of the Act in 1970, and (3) incorporate "housekeeping" changes. Following are some of the more significant recommendations made for amending the Coastal Marshlands Protection Act.

Legislative Purpose

To better describe the intent of the legislature in passing the Coastal Marshlands Protection Act, it was recommended that a section on legislative purpose be added to the Act. States generally assert their ownership of tidelands up to the boundary lines established by the high tide. The public trust doctrine describes this state ownership of tidelands as being subject to a trust obligation to maintain them for certain public uses. Traditionally, these uses have included navigation, commerce, and fishing. Recently, the public trust doctrine has been expanded to include the public's environmental and recreational interests in the tidelands.

The Georgia Supreme Court in State v. Ashmore established the state's ownership of the tidal foreshore as well as submerged lands below the low tide mark.[19] Although placing title to the tidal foreshore in the state, the Court did not specify the extent of the public's interest in the area. At least in its most traditional form, the public trust doctrine was adopted with the common law in Georgia.[20] Thus, at a minimum, the public's interest in coastal fishing, navigation, and commerce is held in trust by the state. Additionally, the Georgia Supreme Court has seemed to imply the existence of a protected public right of recreation in the tidelands.[21]

The 1976 "vital areas" provision of Georgia's Constitution empowers the General Assembly to restrict land use "in order to protect and preserve the natural resources, environment, and vital areas of the state."[22] The provision effectively acknowledges the existence of resources that are of equal importance to all citizens of the state and deserving of special legal treatment.[23] In 1979, the Shore Assistance Act[24] identified the coastal "sand-sharing" system as a vital area of the state. The Act's language reflects public trust concepts:[25] recognition of the delicate nature of tidal lands,[26] statewide public interest in such lands,[27] and the state's assumption of a trustee-like responsibility to maintain the integrity of the shore system.[28]

Similarly, Georgia's Assistant Attorney General, in explaining the Coastal Marshlands Protection Act of 1970, stated: "In general, the philosophy of the Act and its implementation is that coastal marshlands are an area in which the public's interest is primary . . . all marshlands are subject to the common law public trust and to the federal navigational servitude, guaranteeing public access and use."[29] The denial of permits deemed contrary to the "public interest" and the issuance of a low number of permits for marinas and similar water-dependent activities support the conclusion that the Coastal Marshlands Protection Act currently embodies implicit public trust notions.[30]

It was recommended that the Coastal Marshlands Protection Act be revised to incorporate developments in Georgia law concerning the public interest in the marshes which have occurred subsequent to its passage in 1970. Specifically, the coastal marshes could be identified as a state "vital area"; the public's interest in the marshes could be described in the more explicitly trust-like terms of the Shore Assistance Act.

Definitions

Recommendations were made to expand the definition section of the Act. Of particular note is the recommended change in the definition of "marshland." This change addresses two points. First, by specifically including mudflats and intertidal bottoms as components of the marsh, ambiguity in area coverage is decreased. Mudflats and intertidal bottoms are components of the marsh ecosystem but the fact that they are not specifically mentioned in the definition creates an unclear situation. Changing the definition as recommended clarifies the meaning of the term "marshlands." Second, the list of marsh indicator plants should be improved and expanded. Variation exists from state to state on the use of vegetation to define marshlands. Louisiana and Mississippi do not use vegetation in their definitions. South Carolina does not list the species but requires a field biologist to determine on a case-by-case basis if the area is a marsh. North Carolina lists 10 species in its definition; Alabama list 14 species; Florida lists 247 species; and Texas lists 62 species. The list of 13 species recommended is more in line with the species that are listed in the DNR regulations and the ones used in staff on-site inspections to determine if an area is covered by the Coastal Marshlands Protection Act. Also, the species of *Juncus* dominant in the Georgia marshes is *roemarianus* not *gerardi*. Consequently, it was recommended that *gerardi* be deleted from the list of indicator species. These changes thus provide a more accurate and wider variety of plants to be used as indicators of salt and brackish marsh conditions.

Activities with Major Impacts on Marshlands

The Committee is confronted by some difficulty in deciding to issue or deny a permit for certain activities that, by their nature, could potentially have a major impact on the marsh but might also be necessary or desirable. Although the authority exists to address these concerns in the existing law, there is limited direction on how the Committee should do so. Consequently, it was recommended that a new section be added to the law to specifically designate these activities as ones potentially having a major impact on the marshes and to either establish procedures in the new section or direct the Board of Natural Resources to promulgate rules and regulations to do so. The term "major impact" refers to the use of impoundments for agricultural, aquacultural, and waterfowl management purposes; the siting of marinas; the construction of public roads in the marshes; and other activities deemed by the Board of Natural Resources to potentially have a major impact on the marshes.

The use of Georgia wetlands for agricultural production began in Colonial times. During this period, little was known about the value of wetlands which were generally perceived as sites of pestilence and roadblocks to development. Conversion of wetlands to economically productive use of any type was viewed as an improvement. Crop production, principally rice, occurred in impounded tidally influenced freshwater wetlands where water levels could be manipulated. Georgia rice production reached its peak between 1850 and 1860. Chatham County was the leading producer of rice followed by Camden, McIntosh, Glynn, Liberty, and Bryan.[31]

Interest in aquaculture has appeared more recently. Aquaculture, depending on the species involved, may utilize either fresh or saltwater. Current interest in upgrading former dikes to reimpound coastal marshes relates, in part, to their potential agricultural and aquacultural uses, including the management of impoundments for waterfowl purposes. Generally, conversion of marshes for agricultural or aquacultural purposes will require the use of impoundments to regulate the depth, salinity, and flow of water. The major exception to the use of impoundments for agricultural and aquacultural purposes, however, is the culture of clams and oysters which requires the improvement of beds but does not require dredging and filling of marshlands.

Clams and oysters naturally occur in the Georgia marshes. No impoundments are necessary for commercial production of these shellfish, but production can be increased by managing the beds to improve growth conditions. This requires an investment by individuals to culture these organisms. Clams and oysters, however, are also sought after for recreational harvesting by the public. Governmental interest in clams and oysters, thus, relates to two factors: (1) water quality conditions necessary to protect public health and (2) the rights to harvest shellfish.

The Georgia DNR is currently sampling water quality in the coastal marshes to determine which areas have water quality conditions sufficient to allow shellfish harvesting. As of July 1987, 162,288 acres met the water quality standards and were approved for shellfish harvesting. Shellfish harvesting was prohibited from 132,244 acres which did not meet the water quality standards. It is estimated by the DNR staff that with further study, an additional 50,000 acres could be approved for shellfish harvesting. There is considerable potential to improve conditions for both public and commercial shellfish harvesting in Georgia, but this depends, in part, on ensuring the quality of water in the marshes.

A second concern with shellfish harvesting in the coastal marshes involves the rights to the shellfish. If the shellfish resources are adjacent to upland property, the 1902 Riparian Rights Law gives the exclusive harvest rights to the upland property owner.[32] If no adjacent upland is present, however, the right to control shellfish harvest is vested with the state unless a crown or state grant specifically conveyed that right. Where no exclusive claims exist, DNR may lease the area for commercial harvest or designate the area for public harvest. Consequently, amending the Coastal Marshlands Protection Act to address the rights to shellfish harvesting is not necessary.

Use of impoundments for aquacultural purposes differs from other marsh-converting activities. In most cases, permitted activities convert coastal marshes to dry land uses (e.g., marinas, roads). Impoundments are generally utilized to convert one type of wetland to another type of wetland. This is especially true when impoundments are used for waterfowl management purposes.

In passing upon the application for a permit (including permits for agricultural or aquacultural purposes), the Coastal Marshlands Protection Committee considers the public interest which, as defined by the Act, includes the impact of the activity on the natural flow of water and navigation; its impact on erosion, shoaling, and stagnating water; and its impact on the conservation of marine life, wildlife, and other natural resources. Based on these considerations, it would be most likely that the Committee would rule that impoundments generally are not in the public interest, especially in cases where former dikes have deteriorated. Upgrading or rebuilding these dikes to impound marshlands would have a greater impact on the marshes than upgrading dikes that are essentially intact and serviceable over most of their extent. Research conducted on the impacts of impoundments on marshes supports the contention that impoundments have a major impact on the marshes, but more research is necessary to understand the total effect.[33]

It is estimated by the DNR that between 35,000 and 40,000 acres of marsh were formerly diked along the Georgia coast. Long estimated that in 1859 alone, 23,000 acres of rice were cultivated in Georgia.[34] These estimates are probably not inflated since, by comparison, South Carolina has nearly 150,000 acres of formerly impounded coastal marshlands.[35] Reimpounding these marshes for waterfowl management or aquacultural production would most likely have a significant impact on the coastal marsh system. To protect the public interest in Georgia marshes and to ensure that impoundments are managed to minimize their adverse impacts on the marshes, it was recommended that the use of impoundments be regulated under the Coastal Marshlands Protection Act. Utilization of impoundments for these purposes should require a permit that is issued only after careful consideration of the impacts of the proposed impoundment use and the development of a management plan that specifies what management steps are required of the applicant. Monitoring of the management efforts to ensure that the plan is being implemented is necessary. In addition, to ensure that adequate funds are available to return the marsh to prepermit conditions, a bond sufficient to cover the cost of restoration is necessary.

The expansion of tourism and recreation in coastal areas creates a demand for marina facilities. Marina facilities, however, may have a number of negative impacts on the coastal environment, including water quality degradation, loss of aquatic habitat, and navigation impairment. Of primary concern is the water quality impacts caused by marinas which may include changes in turbidity, increased coliform bacteria populations, depletion of dissolved oxygen, and increased concentrations of metals and hydrocarbons. Sources of these impacts include marina sanitation devices, bilge water, wastewater disposal, dredging, fueling, boat washing, and boat exhaust. To avoid or minimize the impacts on coastal resources, marina development requires careful consideration. As more is learned about the impacts of marinas and how to minimize the impacts, requirements will change. Currently, DNR rules and regulations include guidelines for marina construction. Consequently, amending the Coastal Marshlands Protection Act to address marina siting is not necessary, but language could be added that identifies marinas as an activity with potentially major impacts on marshes and that specifically directs the adoption by the Board of Natural Resources of rules and regulations designed to minimize the impact of marinas on marshes.

The Coastal Marshlands Protection Act is silent on the construction of public roads in marshlands. The rules and regulations adopted by the Board of Natural Resources state that "roads placed on pilings rather than constructed as solid field causeways" would likely be considered not contrary to the public interest.

In April 1984, a draft policy was prepared by staff to govern the construction of public roadways across coastal marshlands to island and hammock lands. The policy would not apply to private roadways or driveways which are generally prohibited. The policy would also not apply to activities of the Georgia Department of Transportation since such activities are exempt from the requirements of the Coastal Marshlands Protection Act. Public roads included would most commonly be those constructed by developers and dedicated to local governments. The draft policy included 15 criteria which would have to be met before public roadways could be constructed. No roads would be allowed in the more productive coastal marshes.

The Board of Natural Resources has not adopted this draft policy. Under the existing law, the Board has the authority to do so. Adding the construction of public roads to this proposed section on activities with major impacts on the marshes identifies it as such and directs the Board to specifically adopt rules and regulations controlling the construction of public roads in the marshes.

Exempted Activities

A number of activities are exempted from the requirements of the Act. Clearly, these exemptions have resulted in the conversion of many times more acreage of marsh than have been converted under the Coastal Marshlands Protection Act. Increased dredging and filling activities by the Navy at Kings Bay and increased dredging and filling for channel and port activities will result in the conversion of considerable marsh acreage.

Insufficient data were available to determine the overall impact of exempted activities on the coastal marshlands. To ascertain the nature and extent of these impacts, it was recommended that the legislature request each exempted agency to provide it pertinent information on the extent of the agency's activities in the coastal marshes.

CONCLUSIONS

The coast of Georgia encompasses over 13 percent of the marshlands between North Carolina and Louisiana, more than any other state on the east coast. These marshes are a valuable resource that require protection from those activities that would cause them to be converted to nonwetland uses. The 1970 Coastal Marshlands Protection Act was enacted for this reason and has been an effective mechanism for protecting the public interest in the coastal marshes while allowing for use of the marshlands. Although amending the Act is not currently necessary, recommendations have been made that will: (1) enable the law to better address concerns that have appeared or increased since the original bill was enacted, (2) incorporate constitutional changes (vital areas designation) and judicial decisions that have occurred since the passage of the Act in 1970, and (3) incorporate "housekeeping" changes.

The Coastal Marshlands Protection Act deals specifically with those activities that would occur within the marshes. Other threats to the marshes exist that are not addressed by the Act. First, the estuarine system represents a sink for contaminants carried by rivers to the coast. Abuses of the hinterland are reflected in the quality of the water that flows into the marshes. Although considerable effort has been placed on control of discharges from municipal and industrial sources, nonpoint runoff into the state's streams can have a significant effect on water quality. Of particular concern for the coastal marshes is the concentration of nitrates in stream water. Nitrates are not generally removed by waste water treatment facilities and may be found in high concentrations in runoff from lawns and agricultural fields. Nitrates may cause the eutrophication of the estuarine system and, therefore, may have an adverse impact on the marshes.[36]

A second concern not addressed by the Coastal Marshlands Protection act is the effect on coastal marshes of alterations in adjacent or interconnected freshwater wetlands. These wetlands are afforded no protection under state law. Their condition, however, may have a direct impact on the functioning of the coastal marshes.

A third concern not addressed by the Act relates to the lack of any form of buffer zone around the coastal marshes. To protect water environments from activities that occur on land, it is generally necessary to provide at least some form of vegetative buffer zone to filter nonpoint contaminants before they reach the water resource. To ensure the health of Georgia's coastal marshes, it may be necessary to provide a buffer zone around the marshes.

An over-riding consideration related to the Coastal Marshlands Protection Act is the impact of global sea level rise on the coastal region of Georgia. The increase in carbon dioxide and other gases from anthropogenic sources in the atmosphere is resulting in an increase in the global temperature.[37] This so called "greenhouse effect" is causing the polar icecaps to melt with a resulting rise in sea level. During the past century, the sea level along the east coast of the United States has risen 30 centimeters (11.8 inches) or three centimeters per decade. This rise in sea level along the Georgia coast is predicted to continue and possibly increase. Although the rate of rise is slow, the impact of higher sea levels and storm damage resulting from the rise may be significant. The Coastal Marshlands Protection Act provides a partial mechanism for considering the potential impact of sea level rise on private facilities and public infrastructure investments. Consequently, decisions made under the Coastal Marshlands Protection Act should be made in light of the fact that the sea level will continue to rise slowly for the foreseeable future.

ACKNOWLEDGEMENTS

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LITERATURE CITED

[1]A. Sydney Johnson, Hilburn O. Hillestad, Sheryl Fanning Shanholtzer, and G. Frederick Shanholtzer. An Ecological Survey of the Coastal Region of Georgia, National Park Service Monograph Series, No. 3., 1974; L. R. Pomeroy and R. G. Wiegert. The Ecology of a Salt Marsh, Springer-Verlag, New York, 1981; J. E. Kundell and S. W. Woolf. Georgia Wetlands: Trends and Policy Options, Carl Vinson Institute of Government, University of Georgia, Athens, Georgia, 1986.

[2]Savannah Morning News-Evening Press, February 28, 1971, at B1.

[3]See: O.C.G.A. 12-5-283, Ga.L. 1970 p. 939; Ga.L. 1972 p. 991 1.

[4]See: Ga.L. 1972 p. 991, 1015; Ga.L. 1984, p. 466 3.

[5]O.C.G.A. 12-5-285(a), 281(2), 1982.

[6]O.C.G.A. 12-5-281(4).

[7]O.C.G.A. 12-5-281.

[8]O.C.G.A. 12-5-292.

[9]O.C.G.A. 12-5-282 (Supp. 1984).

[10]O.C.G.A. 12-5-285(e).

[11]O.C.G.A. 12-5-285(f).

[12]O.C.G.A. 12-5-288.

[13]O.C.G.A. 12-5-283.

[14]O.C.G.A. 12-5-286.

[15]Johnson, *supra* n. 1.

[16]Georgia Department of Natural Resources, unpublished Landsat analysis, 1979.

[17]Georgia Department of Natural Resources, unpublished data.

[18]Georgia Department of Natural Resources, *Managing Georgia's Natural Resources in the 1990's: A Five Year Strategy and Summary of Programs*, 1987.

[19]236 Ga. 401, 224 SE 2d 334, *cert. denied*, 429 U.S. 830, 1976.

[20]*Johnson v. State*, 114 Ga. 790, 40 SE 807, 1902; *West v. Baumgarten*, 124 Ga. App. 318, 184 SE 2d 213, 1971, *rev'd on other grounds*, 228 Ga. 671, 187 SE 2d 665, 1972.

[21]*Smith v. Bruce*, 241 Ga. 133, 224 SE 2d 559, 1978; *Lines v. State*, 245 Ga. 390, 264 SE 2d 891, 1980; *Goodyear v. Trust Co. Bank*, 247 Ga. 281, 276 SE 2d 30, 1981.

[22]Ga. Const., Art. III, section 6, para. 2(a)(1).

[23]Note, "A Tidelands Trust for Georgia," 17 Ga. L. Rev. 874, 1983.

[24]O.C.G.A. 12-5-230.

[25]Note, Ball, "Environment, Natural Resources, and Land Use," Annual Survey of Georgia Law, 35 Mercer L. Rev. 147, 164, 1983.

[26]O.C.G.A. 12-5-231 (legislative purpose): "It is recognized that the coastal sand dunes . . . are the fragile product of shoreline evolution, they are easily disturbed by actions harming their vegetation or inhibiting their natural development."

[27]Id.: ". . . the General Assembly declares that the management of the sand-sharing system has more than local significance, is of equal importance to all citizens of the state, is of state-wide concern, and consequently is properly a matter for regulation under the police power of the state."

[28]Id.: "The General Assembly further finds that this natural resource system . . . is important to conserve for the present and future use and enjoyment of all citizens and visitors to this state. . ."

[29]Barmeyer, "Requirements for Construction and Other Activities Affecting Beaches, Marshlands and Coastal Navigable Waters," in Institute of Continuing Legal Education in Georgia Program Materials for Seminar on Basics of Environmental Law 9-1, 9-11, 1980.

[30]Note, supra n. 25 at 874, 875.

[31]Johnson, supra n. 1.

[32]Ga. L. 1902 p. 108.

[33]M. Richard DeVoe and Douglas S. Baughman. Ed. South Carolina Coastal Wetland Impoundments: Ecological Characterization, Management, Status, and Use, Volume II, South Carolina Sea Grant Consortium, p. 593, 1987.

[34]F. L. Long. "Chemical Properties and Fertility Requirements of some Brackish Marsh Soils of Georgia," M.S. Thesis, University of Georgia, Athens, Ga., 1958.

[35]M. R. DeVoe and D. S. Baughman. "Coastal Wetland Impoundment Project: Problem Identification and Project Description," supra n. 36.

[36]Richard A. Smith, Richard B. Alexander, and M. Gordon Wolman. "Water-Quality Trends in the Nation's Rivers," Science, 235, pp. 1607-1615, March 27, 1987.

[37]National Research Council. Responding to Changes in Sea Level: Engineering Implications, National Academy Press, Washington, D.C., 1987.

ANATOMY OF A HIGH MARSH RESTORATION EFFORT

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ABSTRACT

In April 1987, a real estate development firm began filling a high saltmarsh on Long Island without Department of the Army permit authorization. Approximately 2.2 hectares were filled before the illegal activity was discovered. The Army issued a cease-and-desist order, quickly followed by a restoration order and fill removal began twenty-six days after the material was placed. The Army directed the violator to remove the fill as rapidly as possible while minimizing damage to the underlying vegetation and root zone. The rationale was that swift and careful removal would enable at least some of the underlying vegetation to recover. The developer first used a bulldozer to remove most of the fill to within 10 to 20 cm of the underlying vegetation, then a combination of a tractor towing a rake and hand raking to remove more of the fill and expose the underlying vegetation. By the end of the 1987 growing season much of the original vegetation was growing vigorously. The importance of coordination, decisive timely action, and momentum are illustrated as essential elements of successful enforcement work.

INTRODUCTION

The purpose of this paper is to describe requirements, on both administrative and technical levels, for a successful enforcement effort by the New York District of the U.S. Army Corps of Engineers (Corps) against a violator of Section 404 of the Clean Water Act.

The Group for the South Fork, a local environmental group in Southampton, Long Island, observed unauthorized filling of wetlands on April 13, 1987 in the hamlet of East Quogue, Suffolk County, New York. They reported the matter to the New York Department of Environmental Conservation (NYDEC).

NYDEC has an extremely effective state regulatory program. However, its jurisdiction is defined by published wetlands maps. Since the subject site was not included on these published maps, it had no regulatory jurisdiction over the site. On April 15, 1987, both the Group for the South Fork and the NYDEC called the Corps to report a possible violation of Clean Water Act requirements. The Corps inspected the site on April 16, 1987 and concluded that unauthorized fill had been discharged into wetlands. Since this discharge did not have prior Department of the Army authorization, it had been placed in violation of Section 404 of the Clean Water Act.

STUDY SITE

Figure 1. View of the restoration site from the southeast in May 1988 about one year after restoration.

The study site was a 2.2-hectare (5.5-acre) tract on part of the barrier beach system that extends along the south shore of Long Island, New York. It was bordered on the north (estuarine) side by a macadam road, beyond which was a saltmarsh cordgrass tidal marsh of Shinnecock Bay. The south (seaward) side adjoined a primary dune system that fronted on the Atlantic Ocean. Undisturbed high saltmarshes abutted the east and west borders of the site. Prior to placement of fill, the violation site also had been a high saltmarsh.

Since we did not see the site before it was disturbed, we cannot describe pre-disturbance species composition with complete confidence. Based upon our analysis of pre-fill photographs provided by NYDEC, however, it appears that saltmarsh hay (*Spartina patens*), spikegrass (*Distichlis spicata*), and groundsel-tree (*Baccharis halimifolia*) were the dominant species. The site also contained common reed (*Phragmites* australis) and scattered individuals of sea-lavender (Limonium carolinianum), salt-spray rose (Rosa rugosa) and probably marsh-elder (Iva frutescens).

We examined the undisturbed high saltmarsh that abutted the eastern edge of the study site, because it looked very similar to the NYDEC photos of our site. With the exception of some groundsel-tree along the southern boundary, this eastern marsh consisted almost entirely of a very dense cover of saltmarsh hay and spikegrass.

Figure 2. Eastern boundary of site about one year after restoration effort: right side of photo is undisturbed high saltmarsh; left side is restored area.

Other species such as three-square (Scirpus americanus) and seaside goldenrod (Solidago sempervirens) were present in only very minor amounts, none of them constituting more than one or two percent of the areal cover.

Common reed on the study site was limited to two small stands, one of approximately .01 hectares (.02 acres) along the southern border near the central part of the site and a second of approximately 0.1 hectares (0.25 acres) in the southwestern corner of the site, where old fill had once been placed.

Detailed tidal surveys were not available for the study site. However, using survey data for a site less than 100 meters to the east, we estimated that, before the site was filled, tidal waters from

Shinnecock Bay overtopped the road and inundated the site approximately 75-100 times each year.

MATERIALS AND METHODS

Administrative

There are three discrete phases associated with administration of a successful enforcement action: discovery, coordination, and enforcement. These phases are sequential and dependent: deficient work in discovery or coordination often dictates failure in enforcement.

"Discovery" begins with learning of an alleged violation and ends when you are reasonably certain you have all the facts documented in the file. It is characterized by an intensive work effort in a short period of time. The information that must be collected consists of:

1. Specific documentation of Corps of Engineers jurisdiction over the site based upon three-parameter determination data (soils, hydrology and vegetation) obtained at the site, published data (NWI maps, SCS soil survey maps, etc.), and site survey data if available.

2. Documentation of property ownership to establish that the alleged violator owns the site.

3. Documentation of the illegal act through statements of eyewitnesses, photographs, and statements of contractors testifying that they did the work at the site at the direction of the alleged violator.

4. A legal opinion as to the adequacy of the evidence for prosecuting the case.

5. Collection and documentation of additional evidence required for successful prosecution.

"Coordination" involves obtaining written agency recommendations and justifications for resolution of the violation. The objective of this phase is to develop and document a unified "government position" on the violation and justify that position from a scientific/legal basis. Detailed recommendations are not necessary at this phase. All that is required is reasonable agreement on conceptual resolution such as removal or development of a mitigation proposal.

Because various agencies have different missions and spheres of interest, it is not always possible to reach a consensus. However, the probability of a successful enforcement action is greatest when a unified government position can be reached. Therefore, it is important to make every effort to negotiate a proposed solution acceptable to all the resource agencies involved. This coordination often can be accomplished most effectively by telephone conferences. Although this may require an entire day on the telephone going back and forth between agencies with proposals, counter-proposals, and counter-counter proposals, it is faster than the regular mail.

The coordination phase should not last for more than a few days. Agencies should be asked to telefax individual transmittals of their specific recommendations and the rationale for those recommendations. Receipt of these transmittals usually concludes this phase, unless the violator counters the enforcement order with an alternative proposal reasonable enough to present to the agencies.

After the agencies' responses are added to the file, "enforcement" begins. The proposed solution to the violation case is prepared in the form of an enforcement order. Ideally, this order represents a consensus of agency recommendations. However, if no consensus can be reached in the coordination phase, the most "acceptable" solution should be implemented. The order must be absolutely clear, so there is no question in anyone's mind as to what is being required of the violator.

Upon discovery of the subject violation, the Corps first determined property ownership, which required untangling a complicated web of property transactions that had occurred over the previous six months. Once the owner of the property was identified, the Corps immediately sent the violator a cease-and-desist order (C&D) and made a quick, intensive effort to construct a well-documented file record.

The coordination phase did not last more than a day because all of the agencies involved wanted removal and restoration and most telefaxed their responses the dame day. The U.S. Environmental Protection Agency (EPA), the U.S. Fish and Wildlife Service, the National Marine Fisheries Service, the New York State Division of Coastal Resources and Waterfront Revitalization (which handle the Coastal Zone Management program for the State), the New York State Department of Environmental Conservation, and the Army Corps of Engineers unanimously recommended immediate removal of the fill.

The violator sent a team of attorneys and consultants to the initial meeting but did not attend himself. Under such circumstances, it was impossible to resolve the matter in one meeting. After two office meetings and one on-site meeting within five days, however, the violator agreed to restore the wetland.

Technical

We decided to try to save as much of the original vegetation as possible, reasoning that since the plants had been buried only a few weeks many or most should survive if we removed the fill quickly and carefully. Consequently, the restoration plan involved using a bulldozer, a tractor equipped with a rake and limited hand raking. With the bulldozer, the operator removed fill until the tops of the buried plants were visible. The operator then switched to a large-wheeled light tractor with a rake attachment to remove more of the fill material and expose more of the existing vegetation, without tearing up the established root systems. Hand raking was used in a few places, but was impracticable to use on the entire site.

Because of irregularities in terrain and the relatively large size of the site, it was not possible to do a perfect job of removing the fill. In fact, we did not require the violator to remove all of the fill, reasoning that it would be better to leave a thin layer of fill on the site than remove all of it and risk irreparably damaging the root mat underneath. Inevitably some small depressions remained covered with fill and some "high spots" were scraped bare of vegetation.

NYDEC was concerned that the common reed on old fill at the southwestern part of the site would colonize the entire site because of the disturbance in vegetative cover. However, the developer agreed to remove the old fill and excavate the common reed. We directed the bulldozer operator to excavate deep enough to remove the entire rhizome system of the reed. Because of the depth and persistence of the rhizomes, the operator excavated the area several times, stopping only after the bulldozer repeatedly became mired in the soft marsh sediments. The result was a small (0.2-hectare), shallow (approximately 30-60 cm. in depth) pond near the southwestern corner of the property.

Figure 3. Pond created on southwestern part of site; note common reed.

RESULTS

The initial appearance of the site after the rehabilitative work was finished was rather unpromising. Approximately 25% of the site had remains of old vegetation showing through the surface, but it was not obvious whether the vegetation would survive. However, the plants quickly began to show unmistakable signs of life, and, by late in the 1987 growing season, the site was already more than 50% revegetated.

By May 1988, it was apparent that the site was not coming back exactly as it had been prior to the placement of fill. Based on the photographs of the site prior to filling, most of the species there now were probably there before, but the species composition had definitely changed. The site was about 70% covered with vegetation, with new plants (particularly spikegrass), actively colonizing many of the remaining bare areas. Saltmarsh hay and spikegrass were still the clear dominants, but the presence of other species had greatly increased. The site was no longer primarily a dense saltmarsh hay/spikegrass marsh, with some groundsel-tree, some reed, and perhaps marsh-elder and salt-spray rose. In some parts of the site toward the higher southeastern end, three-square was not only a major component but even locally dominant. In other places, there were considerable amounts of sea-lavender, common glasswort (Salicornia europaea) and seaside goldenrod.

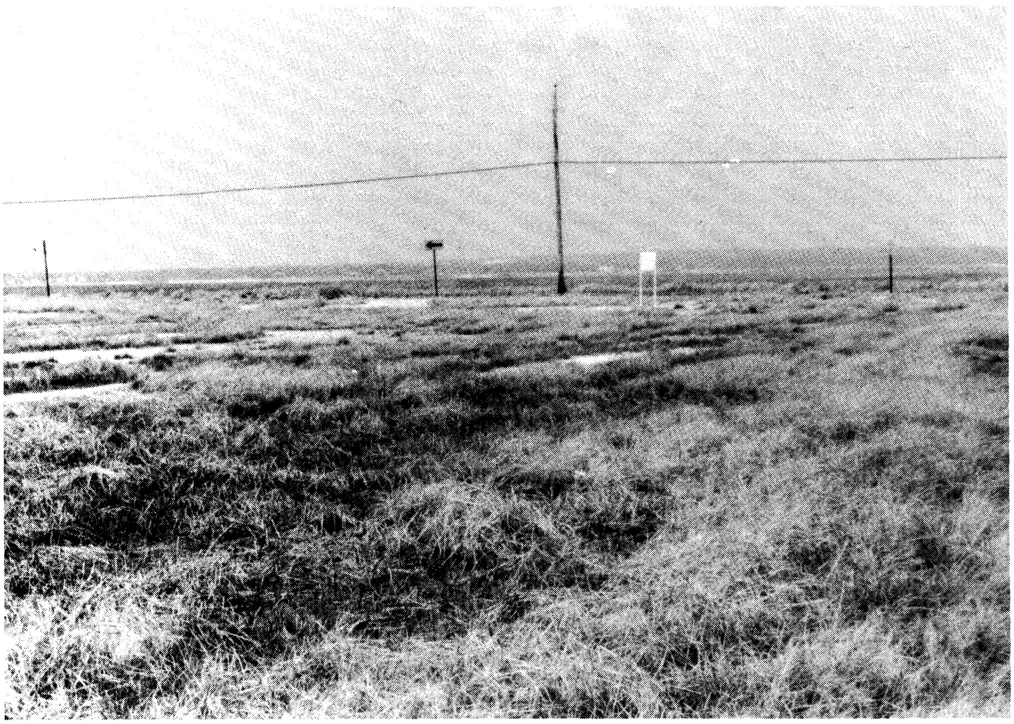

Figure 4. Local dominance of three-square on site (darker area at bottom left).

Saltmarsh cordgrass was also now on the site, although it did not seem to be present in the adjacent undisturbed saltmarsh to the east and we presume it was not present on the site before it was filled. It was the short form of the plant, occurred in only about 10 scattered locations at the site, with only a few stems at each location. We believe these plants originated from seed blown onto the site from the saltmarsh cordgrass marsh on the north side of the road.

We were not able to eliminate the common reed in the vicinity of the pond. There was still reed around much of the pond edge, although three-square was also colonizing the edges and the shallow water near the edges of the pond. The reed near the south-central boundary of the site also returned and, by May 1988, scattered stems of newly-established common reed were visible in several areas that had remained bare of vegetation through the previous growing season.

DISCUSSION AND CONCLUSIONS

Administrative

Time is critical in enforcement work, so it is important to minimize the time period between initial discovery of the violation and issuance of enforcement orders. This is particularly true if the violator is aware that the violation has been discovered. Lack of immediate action suggests indifference or approval on the part of the regulatory agency. If you delay too long, the violator may be able to make a defensive case grounded in reliance: the violator may claim that, since the regulatory agency knew about his activities but did nothing, he or she presumed there was nothing wrong and, relying on this presumption, made substantial further financial investment in the project. Long delays in initiating enforcement action also destroy program credibility.

The most time-critical period is the "crossing," the period between issuance of the order and the violator's agreement to comply with the order. We call it the "crossing" because it is best described through the analogy of a stream crossing. If there is a stream you must cross in a vehicle, a successful crossing requires fast and confident forward motion. If you slow down, falter, or decide to rethink the whole thing in mid-crossing, you lose your momentum, fail to complete the crossing, and end up in the mud.

Momentum is just as critical to an enforcement "crossing." Once the decision is made to force a violator to comply, the agency must keep the enforcement efforts moving fast and not give the violator the opportunity to consolidate his defenses, regroup, "counter-attack" or, worst of all, ignore the enforcement order and go forward with his project. The enforcement "crossing" in this case was made in five days.

The need to maintain momentum in the "crossing" cannot be empha-

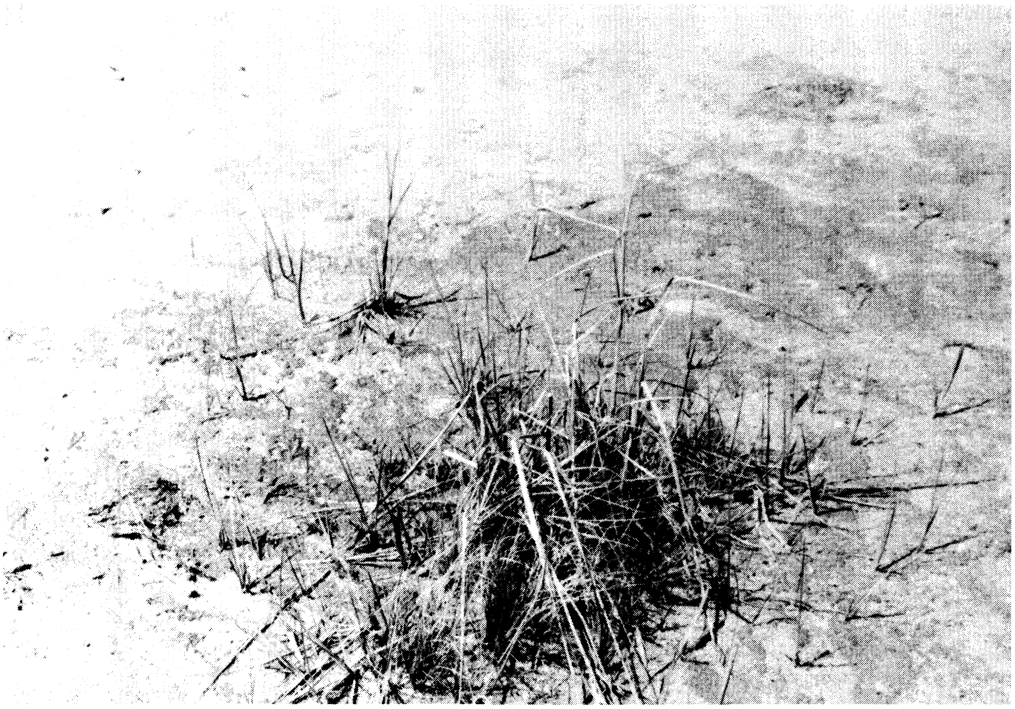

Figure 5. Saltmarsh cordgrass invading unvegetated portions of the site.

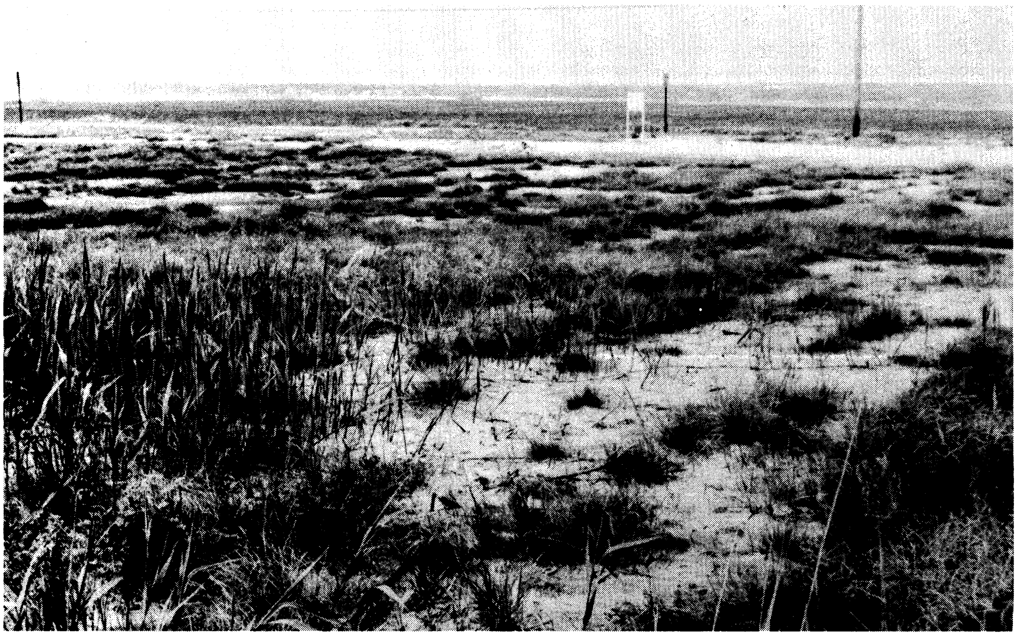

Figure 6. Common reed in south central part of the site; note rhizome "runner" colonizing bare area.

sized enough, because if you "stall out" you are likely to end up in a quagmire of litigation. For example, if you issue an order directing removal of fill within forty-five days and wait forty-five days before seeing if the order has been obeyed, the violator may very well construct a building on top of the illegal fill in that time. You are then faced with the task of persuading a judge to order destruction of a new building rather than simply removal of fill. Unless the violation is flagrant and outrageous, most judges will not order destruction of a new building.

You would have failed when you could have succeeded simply by not allowing the violator time to do any construction. No effective enforcement agency can afford to adopt a "wait and see" attitude about enforcement orders. The agency must follow the C&D order with a definitive work directive within a reasonable length of time.

What is "reasonable," of course, depends upon the "eye of the beholder." To agency personnel, thirty to sixty days to arrive at a well-considered decision resolving the violation may seem quite "reasonable." To a violator, who may have men and equipment under contract for a limited building season, even three or four days might seem totally "unreasonable," particularly if the stoppage is costing many thousands of dollars per day.

Although regulatory agency personnel often tend to be unsympathetic to violators, successful resolution through enforcement requires negotiating settlements in good faith. This includes making a sincere effort to minimize the violator's financial damages due to delays in resolving the violation.

Even when the violator is initially prepared to comply with agency direction, the case can quickly become an expensive, unproductive lawsuit simply because of delay and indecision on the part of the regulatory agency.

An order also should require that the violator notify the agency issuing the order of intent to comply within 24 hours of receiving the order. If the violator calls and indicates intent to comply, the enforcement agency should schedule a meeting to discuss specifics as soon as possible. If the violator does not call, the enforcement agency should call the violator and demand a response. If the violator refuses to meet and refuses to comply, the enforcement agency should quickly seek an immediate legal remedy such as a preliminary injunction or temporary restraining order. The latter almost always brings a resistant violator to the negotiating table.

As to administrative conclusions, there are at least two:

1. Violators cannot long resist the combined weight of Federal and State authority. A well-documented and unified "government" position on resolution will quickly destroy resistance to enforcement.

2. Persistence and follow-through are essential for a successful restoration action. As a practical matter, simply issuing a restoration order does not usually succeed. If the violator tells you to "go fly a kite," you have to seek immediate legal remedies to maintain momentum over the "crossing." Successful enforcement actions are characterized by convincing the violator that it will cost him more to resist the order than to comply. If you convince a violator early on that your agency is prepared to go to court if necessary, you will be able to settle most cases quickly without having to go to court.

Technical

The future presence and distribution of species at this site is not certain. At present, species are still actively competing to occupy the remaining unvegetated areas. Will the site ultimately stabilize at a condition similar to what was originally present, or have site conditions changed sufficiently that a significantly different plant community will ultimately prevail? Monitoring the site over the next several years should give us a strong indication as to the answer to that question.

Meanwhile, it seems intuitively evident that the area will have a somewhat greater habitat value than it had before disturbance because of the greater number of microhabitats on the site: open water and scattered bare areas as well as saltmarsh vegetation. Greater diversity in plant composition may increase foraging attractiveness for microtine rodents such as white-footed mouse (*Peromyscus leucopus*) and meadow vole (*Microtus pennsylvanicus*), thereby attracting locally-found raptors such as northern harrier (*Circus cyaneus*), American kestrel (*Falco sparverius*), and peregrine falcon (*Falco peregrinus*) (U.S. Fish and Wildlife Service 1983). Unvegetated areas may increase nesting habitat value for killdeer (*Charadrius vociferus*).

The continued presence and additional encroachment of common reed concerns us somewhat, but we are undecided as to whether to direct further excavation, hand-pulling or some other action at the site. At this time, we feel that salinity and competition from other vegetation will probably be the chief factors dictating the future distribution of common reed, assuming no further work at the site.

Although literature reports vary somewhat as to the tolerance of common reed to salinity, it appears that the most notable chemical limiting factor is chlorinity exceeding 1.2 percent (12 ppt) (Howard et al. 1978). We tested salinity at the site on two occasions and it was at or above 12 ppt at every point tested except the pond.

In the pond, salinity seemed to vary between 4 ppt to more than 20 ppt depending upon tidal and precipitation events. If soil salinity levels on the remainder of the site remain at or above 12 ppt, encroachment should not proceed much beyond what has already occurred.

Also, although common reed is a highly successful pioneer species, it normally does not invade if an area is already well-vegetated by other species. Rapid establishment of other plant species on the site should preclude pervasive colonization by common reed. The reed that is already established on the site, however, will probably persist indefinitely, even if it ceases significant advancement (Howard et al. 1978).

Another question is whether the colonizing saltmarsh cordgrass will survive. The literature suggests that it may persist. Nixon (1982) noted that many writers considered at least some of the short-form saltmarsh cordgrass to be high saltmarsh vegetation, and included short-form saltmarsh cordgrass with saltmarsh hay and spikegrass as a dominant species of unfilled high saltmarshes. Monitoring the site over the next few years should answer this question as well.

On the technical side at least three conclusions can be drawn from this case study:

1. Certain species of high marsh vegetation, such as saltmarsh hay, spikegrass, and three-square, can survive burial for some period of time. The site was filled on April 13, 1987 and removal operations began 26 days later. These species can apparently survive burial for at least this length of time during their dormant period or very early in the growing season.

2. Disturbed sites may provide an increase in local habitat diversity, at least on a short-term basis.

3. Successful revegetation of the site has been achieved perhaps more rapidly and certainly at much lower cost using the approach we selected than if we had required the violator to remove all fill and replant the site. However, we do not know how long it will take for the site to revegetate completely or how long interim successional stages will last.

On a final note, the formula for success in enforcement may be summarized: Select a well-coordinated, feasible "government" solution as quickly as possible, then "run as fast as you can and keep moving!."

ACKNOWLEDGEMENTS

We wish to thank Mr. Fred Muschake and Mr. Charles Hamilton of the New York State Department of Conservation, and Mr. Kevin McDonald of the Group for the South Fork for their assistance in gathering the information presented in this paper.

LITERATURE CITED

Howard, R., D. C. Rhodes, and J. W. Simmers. 1978. A review of the biology and potential control techniques for Phragmites australis. IWD D-78-26. U.S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi, 80 pp.

Nixon, S. W. 1982. The ecology of New England high salt marshes: A community profile. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, D.C., FWS/OBS-81/55, 70 pp.

U.S. Fish and Wildlife Service, Region 5. 1983. Fish and wildlife resource studies for the Fire Island Inlet to Montauk Point, New York, Beach Erosion Control and Hurricane Protection Project Reformulation Study, Terrestrial Resource Component, Courtland, New York, 140 pp.

EFFECTIVENESS OF MITIGATION TECHNIQUES AT THE ALAFIA RIVER CROSSING

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ABSTRACT

Mitigation activities are frequently required for highway construction projects. This paper explores the effectiveness of revegetation of black rush (*Juncus roemerianus*) marsh in Florida. The results of a six year monitoring effort are reported. Based on the results, it is concluded that elevation was the critical factor in the success or failure of this marsh revegetation effort.

INTRODUCTION

The extension of Interstate 75 from north of Tampa to Naples and on to Miami by the Florida Department of Transportation (FDOT) began in the mid 1970s. A portion of this interstate crossed the Alafia River just west of Riverview (Figure 1). Located in central Hillsborough County, this 49.6 kilometer river originates in the western part of Polk County and empties into Hillsborough Bay near Gibsonton. The interstate crosses this tidally influenced river approximately 5.6 kilometers east of its mouth. At this location the interstate is a six lane rural design. Twin concrete bridges, 473 meters long, cross the river at about 10.4 meters above mean high water. The floodplain was bridged to an elevation of 1.8 meters or more to minimize potential adverse impacts to this sensitive ecological area.

During the development of the final design for the interstate, environmental permits were required from a number of agencies: the U.S. Army Corps of Engineers, U.S. Coast Guard, Florida Department of Environmental Regulation (FDER), and the Tampa Port Authority. These permits were obtained in 1978 prior to construction and, among other things, specified: 1) no fill (temporary or permanent) to be placed in the wetlands; 2) no dredging for access of work barges; 3) the use of temporary timber mats; and 4) an on-site, post-construction inspection to determine if restoration measures would be necessary in the tidal marsh.

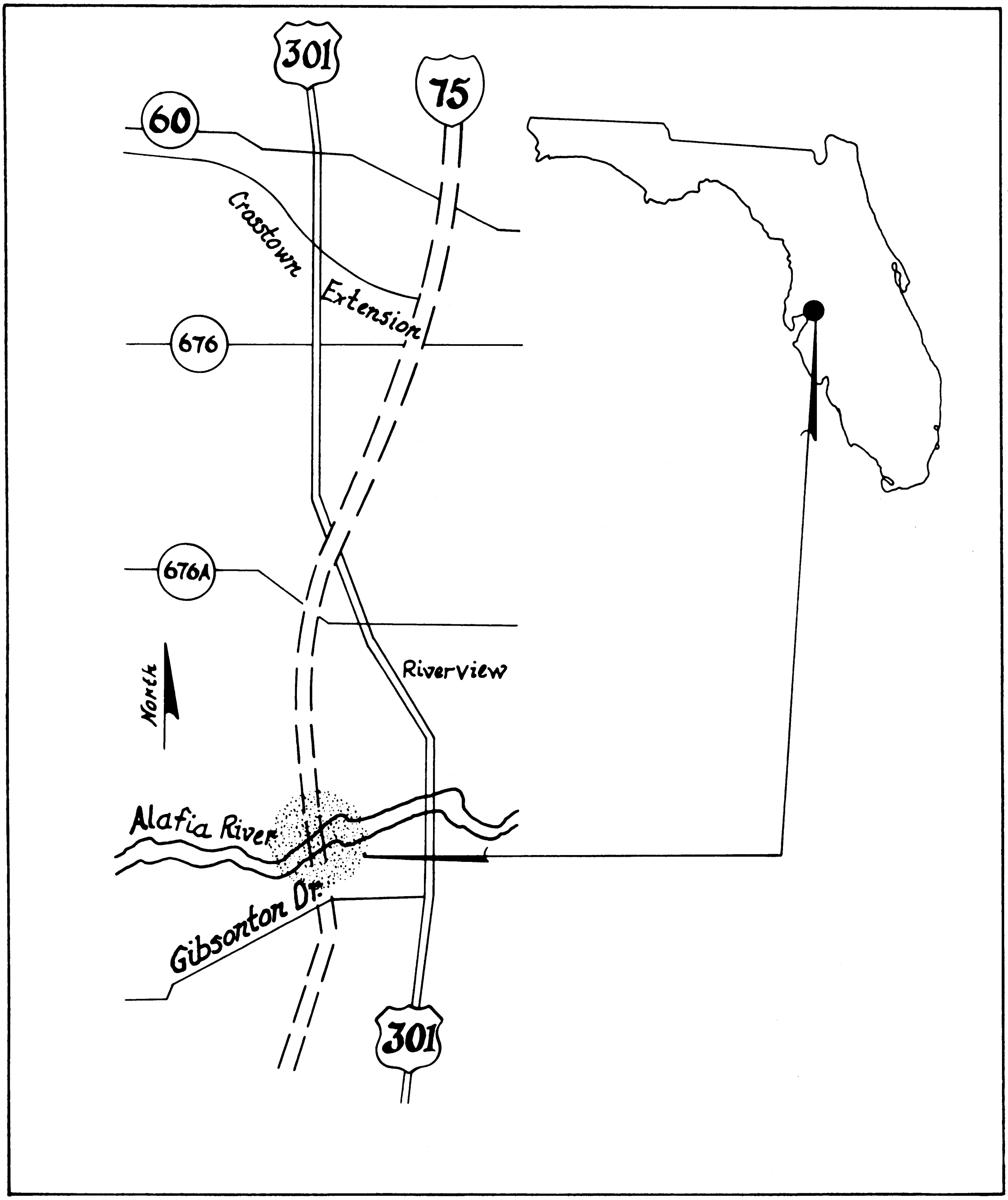

Figure 1. Project location map.

STUDY SITE

In cross section, the bridge and approaches transitioned from a pine/palmetto flatwoods north of the bridge at an elevation of 3.7 meters through a natural marsh edge habitat of palmetto (Serona repens) and cabbage palm (Sabal palmetto) (Burkhalter 1974) at 1.5 meters. Crossing a black rush (Juncus roemerianus) marsh approximately 129.5 meters wide at an elevation of 0.3 meters, the bridge finally reached a natural berm (approximately 0.9 meters high) of alluvial deposition. The berm is vegetated with palmetto, cabbage palm and grasses for about 15.2 meters before it reaches the edge of the Alafia River. Two small tidal creeks cross the black rush marsh under the bridges. On the south side of the river, the bank climbs rapidly to an elevation of 1.8 meters within 18.3 meters of the river's edge (Figure 2). This rapid transition to pine/palmetto flatwoods minimized any adverse impact on the aquatic environment south of the river. Because of this, all mitigation activities required by the permits focused on the north side of the river, specifically the black rush marsh.

As noted earlier, the original permits received in 1978 provided for temporary timber mats to be placed over the black rush marsh. The black rush was to be burned prior to the placement of the mats and the mats were to be removed after construction was completed. Any areas where culverts were to be placed had to be restored to original contour and vegetative cover. The FDOT was required to arrange an on-site post-construction meeting with FDER to determine if restoration measures would be necessary in the tidal black rush marsh. If restoration was deemed necessary, the FDOT was responsible for the development of a restoration plan that met the approval of the FDER.

MATERIALS AND METHODS

The construction contract was awarded in September of 1979 to Wiley N. Jackson Company. Work began in January of 1980 and the contractor quickly proposed several permit modifications to allow for easier, lower cost construction techniques. However, these construction methods would have resulted in greater impacts to the river's floodplain environment. The proposed modifications requested in the spring of 1980 featured a timber loading platform on the north bank of the Alafia River and a 129.5 meter long, 18.3 meter wide temporary access road across the black rush marsh, also on the north side of the river (Figure 3). Two temporary 46 centimeter culvert pipes were to be installed in the two tidal creeks to maintain the tidal flushing these creeks provided. Additional finger fills were provided east and west of the temporary access road.

The access road was to be placed on Mirafi filter fabric after the area of black rush to be covered was burned to ground level; however, wet conditions encountered during construction precluded this. As an alternative, the black rush was cut off near ground level and covered with the fabric. Approximately 1.2 meters of fill material was

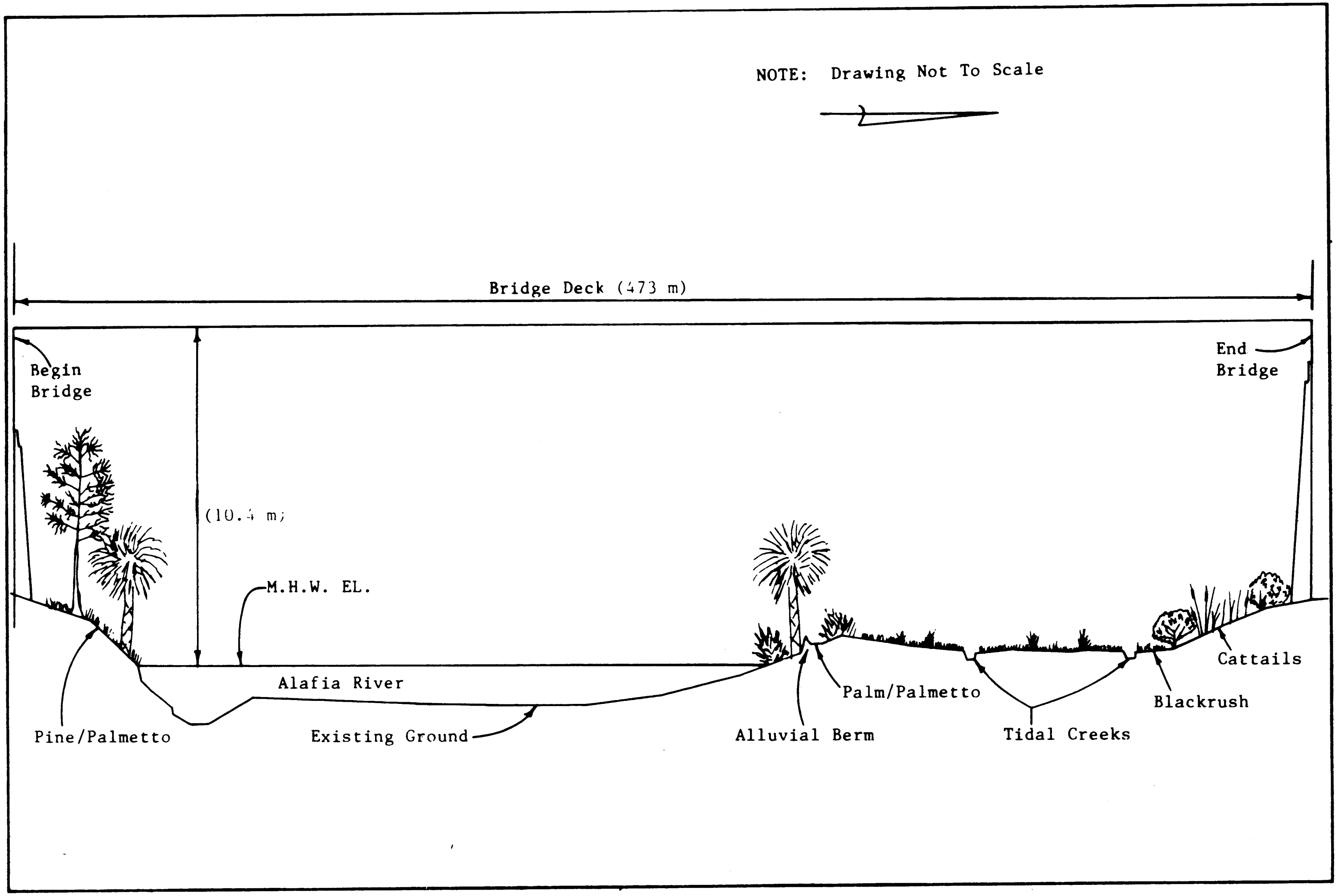

Figure 2. Alafia River floodplain - cross section view.

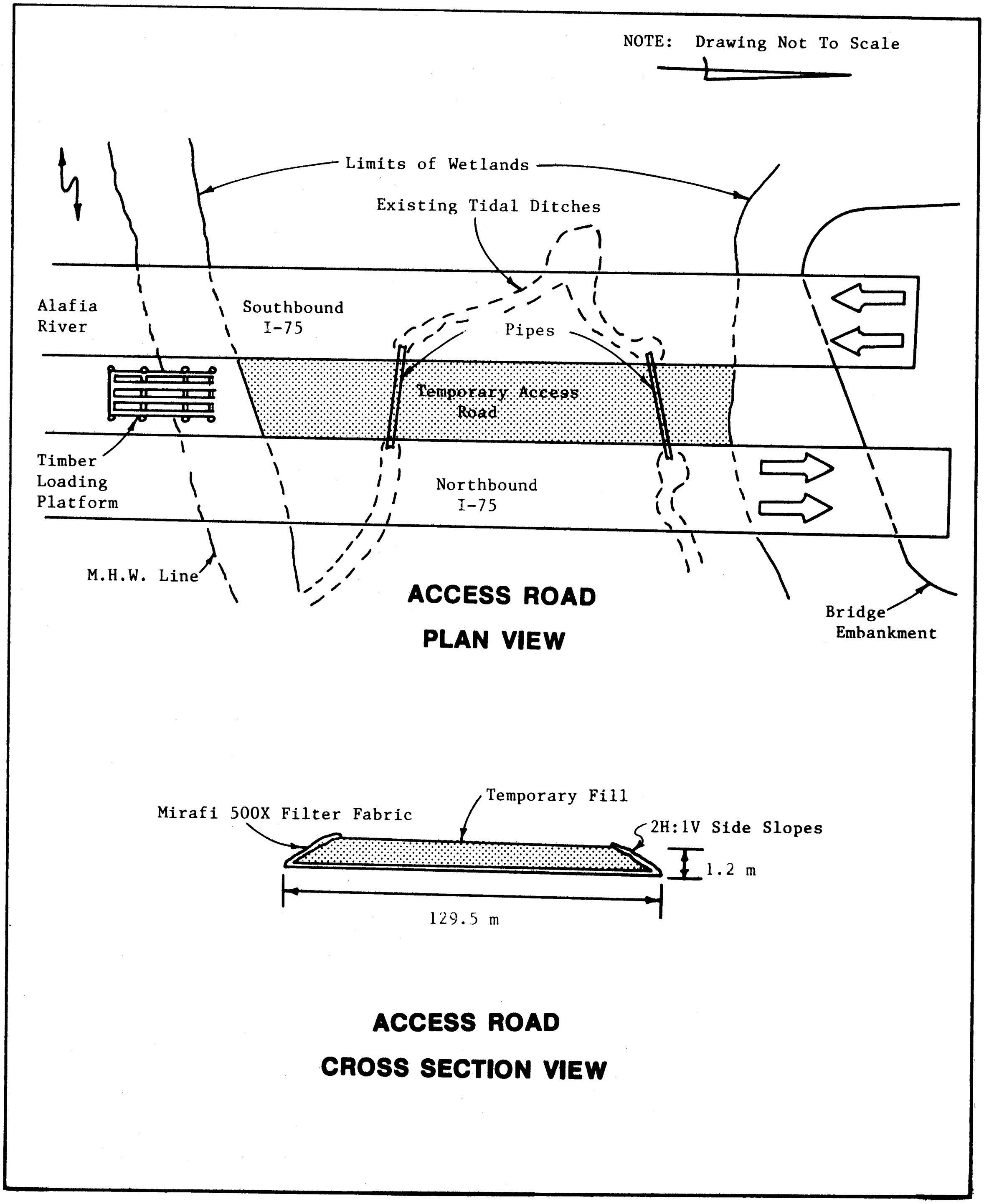

Figure 3. Modified permit sketch.

placed on top of the fabric and the edges of the fabric were rolled back to minimize soil erosion into the marsh.

Adjacent to the main access road, 11 finger fills were constructed to allow access for the bridge construction equipment and materials. The contractor used the same techniques here as on the main access road. The permit modification required that all disturbed areas were to be restored to original contour and revegetated with black rush clumps a minimum of 15 centimeters square. The revegetation plan was to be coordinated with the FDER prior to its implementation.

As construction of the bridge was nearing completion, the FDOT developed a revegetation plan that was approved by the permitting agencies (Figure 4; Table 1). While the FDOT did not guarantee any success rate, it did agree to monitor the site for at least two years and report the results to the FDER.

The contractor commenced the revegetation plan in September 1981 by first removing the fill material and the filter fabric from the finger fills as required. The contractor used a backhoe with a modified bucket to avoid tearing the fabric. The overlay of fill material was carefully removed until the filter fabric was reached. Before uncovering the filter fabric, a test hole was created on one fill pad to determine the condition of the fabric and the black rush under it. It was noteworthy to find that the fabric under the fill was in nearly original condition while the edges of the fabric exposed to the sun was brittle and easily torn. The black rush and supporting muck soil was compressed as much as 30 to 46 centimeters in some locations. As the fabric was uncovered, the edges were rolled toward the center to minimize the loss of the fill material.

Following removal of the fill and filter fabric at each individual location, the fingers were replanted as required by the revegetation plan. The first step in this process involved the restoration of the fill site according to the plan. This involved various techniques including backfilling with a variety of materials (see Table 1) and matching contours as specified in the plan.

The next step was to identify a donor site for the replacement black rush. Undisturbed areas of black rush marsh were available within existing rights-of-way. To minimize the potential impact on these undisturbed areas, the contractor was required to restrict the width of his clearing for donor plants. The contractor also used random patterns and spread the collection on donor plants over a fairly large area. To collect the plants, workers first cut a path 0.6 to 0.9 meters wide through an area of black rush up to 22.9 meters in length. A gasoline-powered weed cutter with a saw tooth blade was used to cut the upper portion of the plants off, leaving about 30 to 46 centimeters of stem. Using a hand shovel, random 15 centimeter squares of black rush were dug and transported to the revegetation areas. Here, operating from planks to avoid sinking into the muck, the workers placed the plugs of black rush into holes created by the use of post

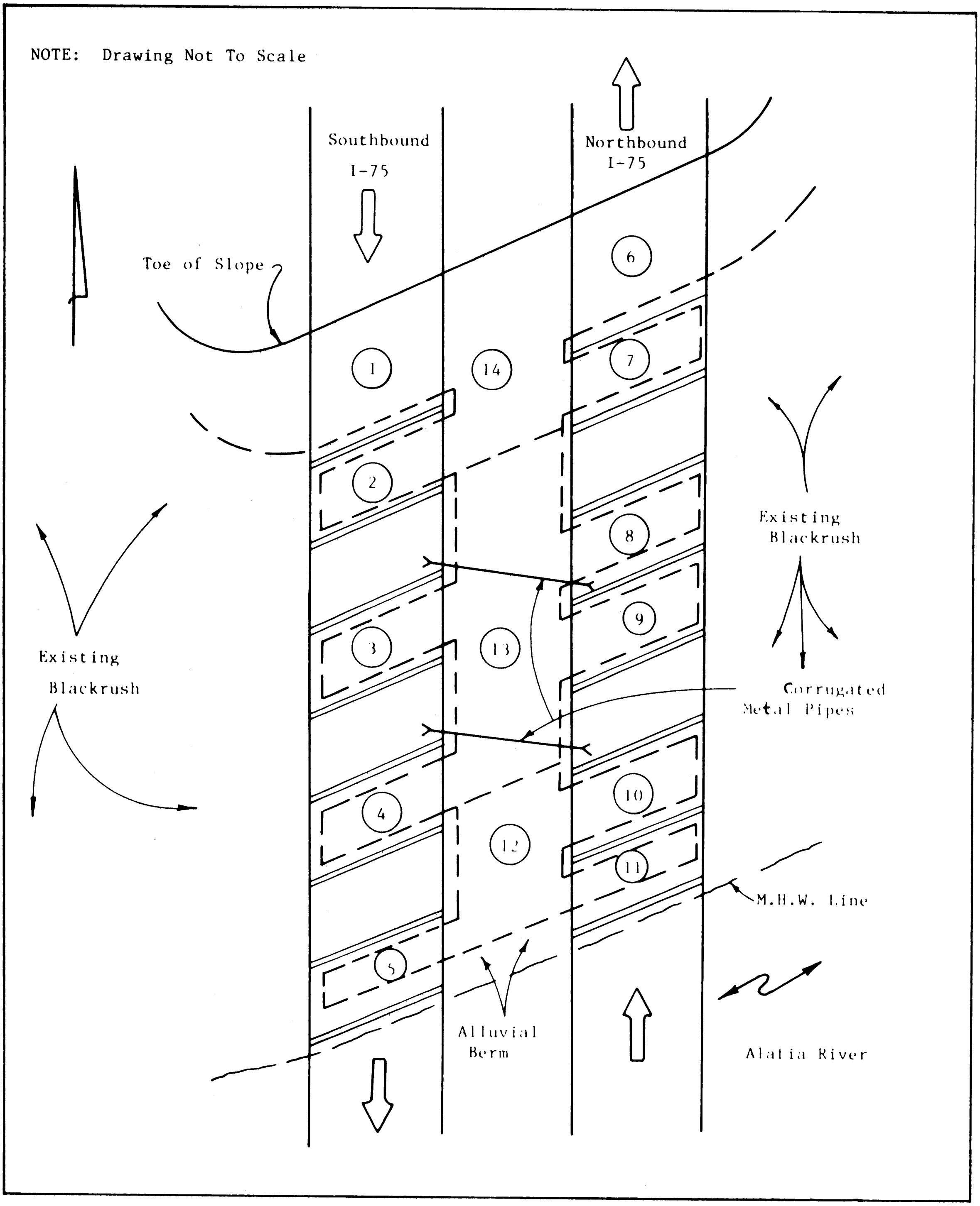

Figure 4. Revegetation plan.

Table 1. Revegetation plan.

Area*	Proposed Activity
1.	Remove filter blanket and backfill with mixture of organic material and topsoil to match natural surrounding contours. No replanting required.
2.	Leave as presently excavated and plant black rush on 0.9 meter centers.
3.	Leave as presently excavated. No replanting.
4.	Backfill with sand (low organic content) to match natural surrounding contours and plant black rush on 0.9 meter centers.
5.	Leave filter blanket with approximately 15 centimeters of existing fill remaining on top of it. No replanting required.
6.	Remove filter blanket and backfill with mixture of organic material and topsoil to match natural surrounding contours and replant black rush on 0.9 meter centers.
7.	Backfill with topsoil to match natural surrounding contours and replant black rush on 0.9 meter centers.
8.	Backfill with topsoil to match natural surrounding contours and replant black rush on 0.9 meter centers.
9.	Remove fill from filter blanket and roll back blanket without damaging existing root system. No replanting required.
10.	Remove fill from filter blanket and roll back blanket without damaging existing root system. Replant black rush on 0.9 meter centers.
11.	Leave filter blanket with approximately 15 centimeters of existing fill remaining on top of it and replant black rush on 0.9 meter centers.
12.	Remove filter blanket and backfill with sand (low organic content) to match natural surrounding contours and replant black rush on 0.9 meter centers.
13.	Remove filter blanket and backfill with topsoil to match natural surrounding contours and replant black rush on 0.9 meter centers.
14.	Remove filter blanket and backfill with mixture of organic material and topsoil to match natural surrounding contours and replant black rush on 0.9 meter centers.

*See Figure 4 for location of each area.

hole diggers. The plants were generally placed on 0.9 meter centers within the areas called for in the revegetation plan.

After the revegetation of finger fills was completed, the main access road was removed and replanted. The process followed was the same as for the finger fills, with removal starting near the river and working northward. The revegetation work was completed by November of 1981.

RESULTS AND DISCUSSION

Based on field reviews conducted in 1981, 1982, 1983, 1986 and 1987, the relative success of the revegetation plan was evaluated. As noted in Table 2, Area 1 shows generally good recovery. This area was restored to original contours but was not revegetated. A diversity of plant species transitions from north to south. On the northern edge is found grasses, dog-fennel (Eupatorium capillifolium), wax-myrtle (Myrica cerifera) and Brazilian pepper (Schinus terebinthifolius). As the area gets wetter, cattail (Typha domingenis) and black rush dominate (Godfrey & Wooten 1981). Total vegetative coverage is better than 60 percent and would probably be better if it were not for livestock creating paths and trampling vegetation.

Table 2. Summary of results.

Area*	Results Percent Coverage of Black Rush
1	25%
2	30%
3	10%
4	90%
5	15%
6	0%
7	30%
8	80%
9	10%
10	20%
11	80%
12	90%
13	80%
14	30%

*See Figure 4 for location of each area.

Area 2 contains one of the two tidal creeks with cattails, alligatorweed (Alteranthera philoxeroides), willows (Salix spp.)

(Tarver, 1978) and ferns present along with black rush. This area was not backfilled but was planted with black rush. About 30 percent of the area is covered with black rush.

Like Area 2, Area 3 also contains one of the tidal creeks and was not backfilled, nor was it replanted. Less than 10 percent natural black rush reestablishment has occurred. Some fern and alligatorweed is present but little other vegetation is found. Open water, even at low tide, occupies 80 to 85 percent of the area.

Black rush, 1.2 to 1.5 meters tall, covers better than 90 percent of Area 4. After backfilling to natural contours, the area was replanted with black rush.

Approaching the alluvial berm separating the marsh from the Alafia River, Area 5 shows less than 15 percent coverage by black rush. This area has approximately 15 centimeters of the original fill left on top of the filter fabric. No revegetation was attempted in this area. What black rush that does exist is shorter (typically 0.9 meters high) than the surrounding plants.

Transitioning from the toe of slope southward on the north-bound bridge (east side), Area 6 is very similar to Area 1. Edge plants such as dog-fennel, Brazilian pepper, wax-myrtle and palmetto are dominant. Livestock paths crisscross the area. After backfilling and contouring of this area was completed, black rush was planted. No surviving black rush could be found.

Area 7 was backfilled and contoured but not replanted. Revegetation is slow, with less than 30 percent coverage in black rush. Alligatorweed and some cattail was found.

After backfilling and contouring, Area 8 was replanted with black rush. About 70 to 80 percent of the area is now covered with black rush 0.9 to 1.5 meters tall.

Following removal of the fill, the filter fabric was carefully removed from Area 9 so that existing black rush root stock was not damaged. No additional plants were introduced. The results show less than 10 percent revegetation in this area although some young plants (0.3 to 0.6 meters high) are in evidence.

After removing the fill and filter fabric, Area 10 was replanted with black rush. No backfilling or contouring took place prior to the planting. Less than 20 percent coverage of 1.2 to 1.5 meter tall black rush has taken place.

Approaching the alluvial berm on the east side of the project, Area 11 shows better than 80 percent revegetation with 1.1 to 1.2 meters tall black rush. Like its western counterpart (Area 5), this area was left with about 15 centimeters of fill material on the filter fabric. The difference appears to be due to the fact that Area 11 was replanted while Area 5 was not.

Area 12 is one of three segments of the main access road. This area nearest to the river was backfilled and contoured before being replanted with black rush. Today a dense coverage of black rush 1.2 to 1.5 meters tall exists.

Like Area 12, Area 13 was backfilled and contoured prior to replanting. Approximately 80 percent of the area is covered with 1.2 to 1.5 meter tall black rush and bisected by the two tidal creeks.

The last area to be revegetation was Area 14. This transition area from marsh to upland at the northern end of the bridge was backfilled and contoured. Black rush marsh gives way to cattail, dog-fennel, Brazilian pepper and wax-myrtle. The area adjacent to the toe of slope is disturbed by livestock paths and contains vines and grasses.

As a final point, the donor sites were monitored to determine if any diverse impacts would result. After six years of growth, it is nearly impossible to distinguish the donor areas from the adjacent growth.

CONCLUSIONS

Based on the results of six years of monitoring this mitigation effort in a tidal marsh, several conclusions can be drawn.

1. Reestablishment of the preconstruction contours is critical to the success of revegetation of tidal black rush marsh. Backfilling (independent of soil type) and contouring before replanting seemed to be the controlling factors in a successful effort. Those areas where backfilling of some type did not take place generally resulted in less than 20 percent black rush coverage. The transition areas (1, 5, 6 and 11) also showed the effects of elevation changes. The black rush does not appear to survive as well when the elevation was increased 30 centimeters over the preconstruction level of the marsh.

2. Supplemental planting will increase the rate of coverage significantly when combined with backfilling and appropriate contouring. A comparison of Areas 2/3, 5/11, 7/8 and 9/10 illustrates this conclusion. Whether planting on 0.9 meter centers is necessary for coverage could not be determined although it is the accepted norm.

3. Removal of the filter fabric does not appear critical to the successful reestablishment of a black rush marsh if the area is contoured and revegetated. Area 11 illustrates this principle well.

4. The use of areas next to the project for donor sites did not have any adverse impact on the viability of the marsh. No indication of the removal of these donor plants is evident if care is taken in

their selection and removal.

5. After six years, the replanted black rush is generally as tall and full as those specimens found in the undisturbed areas.

6. Finally, it appears that replanting will generally be required in this type of marsh setting to insure reasonable coverage. Areas 7 and 8 illustrate this concept although temperature and rainfall may play an important part.

The use of revegetation through plugging is an acceptable method to aid in the reestablishment of a black rush marsh. Before replanting, backfilling and contouring to preconstruction conditions is critical. The use of available topsoil or fill material is adequate to provide for plant growth in this type of marsh environment. Donor sites near the project (if available) will not be adversely impacted if plugs are removed at random over a large area.

ACKNOWLEDGEMENTS

The contribution of Ms. Sherry Swinford to this paper must be noted. Her knowledge of the project and her assistance with the graphics was invaluable.

LITERATURE CITED

Burkhalter, A. P. (Ed.). 1974. Aquatic weed identification and control manual. Florida Department of Natural Resources, 100 pp.

Godfrey, R. K., and J. W. Wooten. 1981. Aquatic and wetland plants of southeastern United States. The University of Georgia Press, 712 pp.

Tarver, D. P. 1978. Aquatic and wetland plants of Florida. Florida Department of Natural Resources, 127 pp.

WETLANDS RECLAMATION USING SAND-CLAY MIX FROM PHOSPHATE MINES: RESULTS AFTER THREE YEARS

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ABSTRACT

In response to the need to establish wetlands following phosphate mining, the authors developed a method for incorporating a perched water table into a reclamation site. An almost impermeable hardpan created from a 2:1 sand-clay mix from mine tailings was covered with overburden with some areas treated additionally with pine flatwoods soil strips, hardwood swamp soil strips, marsh topsoil strips, and untreated sand-clay mix. Details of the method were reported at the 12th Annual Conference in 1985. Here we report follow-up developments and applications of the method. The perched water table proved very effective at retaining surface water. A combination of drought followed by prolonged flooding of the site devastated the plantings, strongly reinforcing the importance of an operable water control system in early stages of the wetlands creation process. The experimental plots described in our first report provided invaluable information about successes and failures which occurred. In a follow-up reclamation effort, the experimental data led to improvement of the method, species selection and some successful strategies including potential use of *Salix caroliniana* as a nurse species for accelerating succession towards a hardwood swamp and an associated stream. Our observations and quantitative data on species diversity and the growth under controlled conditions during the establishment phase seem to show that reclamation is accelerated using our techniques.

INTRODUCTION

The abundance of wetlands in headwaters of drainage basins in Florida has brought many of these areas under State jurisdiction. Although guidelines have been established for making jurisdictional determinations which control the use of the land, the regulations provide for some exceptions under very carefully considered circumstances. For the phosphate industry, the ore which is beneath the wetlands is of importance. Thus, development of a series of protocols which will result in vegetated and floristically diverse wetlands systems on mined land is of great interest to both the industry and to the people of the State of Florida represented by government agencies.

CF Industries has acreages of floodplain wetlands underlain by significant deposits of phosphate rock. Some such land is located in Hardee County along Hickey Branch, a minor tributary of Payne Creek which drains into the Peace River. The actual headwaters of Hickey Branch were mined many years ago and the land left as a series of alternating water-filled pits and spoil piles. The discharge from these pits was directed into Hickey Branch streambed at the boundary with CF lands.

To test our theories on wetland reclamation, we established a series of plots which provided a diversity of elevations, substrates, and tree species (Miller et al. 1985).

One thing that most wetlands have in common is an elevated water table which may be perched over a pervious layer of the presence of an almost impermeable stratum. Therefore, we set out to perch a water table by depositing a sand-clay mix (2:1) layer on overburden at a depth of about 6 dm below the finish elevation. The drying of the mix is relatively fast as compared to pure clay tailings. A capping layer of overburden was spread over the sand-clay to form a gentle slope as soon as equipment could operate on the surface. Five contoured plots were created which differed in their final soil treatment as follows:

1) untreated overburden;

2) hardwood swamp topsoil held in reserve storage piles for several months and then deposited in strips along contours;

3) pine flatwoods topsoil spread as above;

4) marsh topsoil was mechanically collected, immediately transported and spread as above; and

5) untreated sand-clay not contoured.

We planted tublings of red maple (Acer rubrum), pignut hickory (Carya glabra), sugarberry (Celtis laevigata), popash (Fraxinus caroliniana), loblolly bay (Gordonia lasianthus), dahoon holly (Ilex cassine), sweetgum (Liquidambar styraciflua), sweetbay (Magnolia virginiana), blackgum (Nyssa biflora), laurel oak (Quercus laurifolia), and water oak (Quercus nigra). Various species mixes were planted at all contour levels in some copses of 5-7 trees spaced about 2 m (7 ft) apart with larger distances between the copses. Such spacing was designed to create small regions of closed canopy which could provide early niches for establishment of understory wetland taxa. Corners of the planting grid were clearly marked by tall PVC stakes.

Once tublings and stakes were in place, a cover crop of *Aeschynomene* was planted to reduce weed establishment and improve soil structure. The *Aeschynomene* grew very well and weeds were few as compared to open sites in the surrounding area. The stripped-in soils carried seeds and vegetative parts of native species into the site and a diverse herbaceous flora was established during the first season. However, the anticipated rainy season did not materialize and the resulting drought stress resulted in the death of both many tublings and hundreds of volunteer red maple seedlings. The surviving plantings were heavily browsed and many were lost to rodent predation. We thought most situations in the test plots were under control. We had anticipated some possible rodent damage to the new plantings, but we had not reckoned on the amount of damage that can be done by wild hogs. The whole area was scarred with ruts from hog activity. Once the hogs were removed, recovery of the test plots continued. However, the perturbation had been so severe that isolation of success and failure criteria by quantitative methods, so carefully planned for, became impossible. Even so, we learned a great deal by observation from the original test site.

STUDY SITE

By using what we had learned from the experimental plots, a system was designed to create a riverine floodplain woodland in the headwaters region of Hickey Branch. Thus, even as the test plot was recovering we began to put a new reclamation area in place. Our goal remained to create a wooded wetlands system comprised of a diversity of native species in a minimum time. The thrust of the project was to create a new watercourse and floodplain for Hickey Branch immediately adjacent to the existing and previously artificially flooded basin scheduled for future mining (Figure 1). The first segment, designated Area I, comprised the uppermost 15 acres of the new streambed and floodplain (Figure 2).

Area I was generally contoured to between 3-6 dm (1-2 ft) below finish grade and sand-clay mix pumped in to obtain a finish thickness 1-3 dm thick. The sand-clay mix was then covered with overburden to an average depth of 4-5 dm to reach contours shown in Figure 3.

A mix of the same tree species as used in the test plots was planted in the new reclamation area in June of 1985. In addition, small clumps of para grass (*Bachiaria purpurescens*), soft rush (*Juncus effusus*), pickerel weed (*Pontederia cordata*), arrowhead (*Sagittaria lancifolia*), great bulrush (*Scirpus validus*) and cordgrass (*Spartina bakeri*) were planted to contribute to an herbaceous layer.

Gallon size specimens of the *Taxodium*, *Nyssa*, *Fraxinus*, and *Acer* were included as well as tublings of all woody species listed above. Water was allowed into the site and a series of monitoring stations

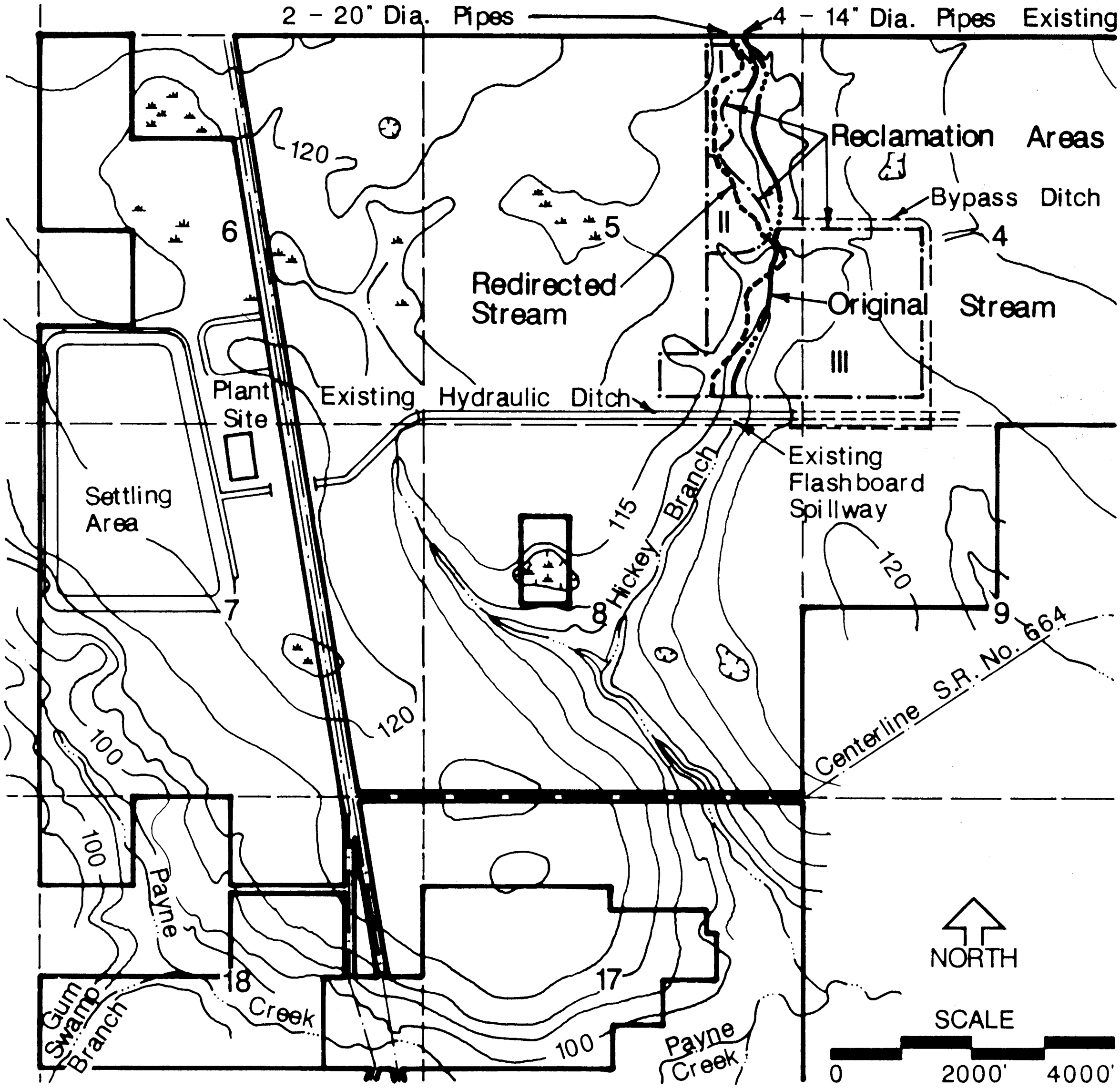

Figure 1. Hickey Branch watershed showing the location of reclamation areas.

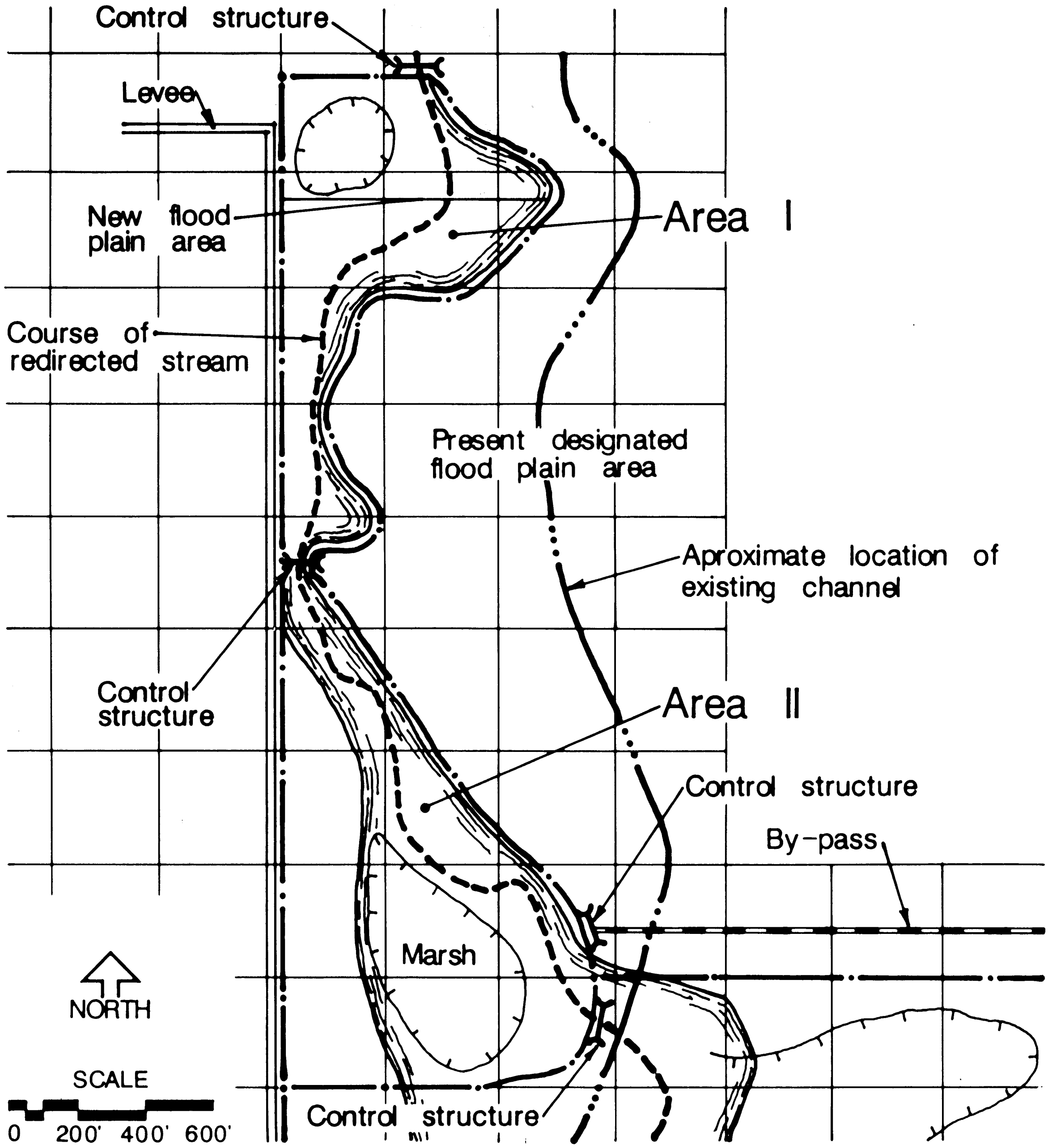

Figure 2. Newly created wetlands areas immediately adjacent to the existing watercourse and floodplain.

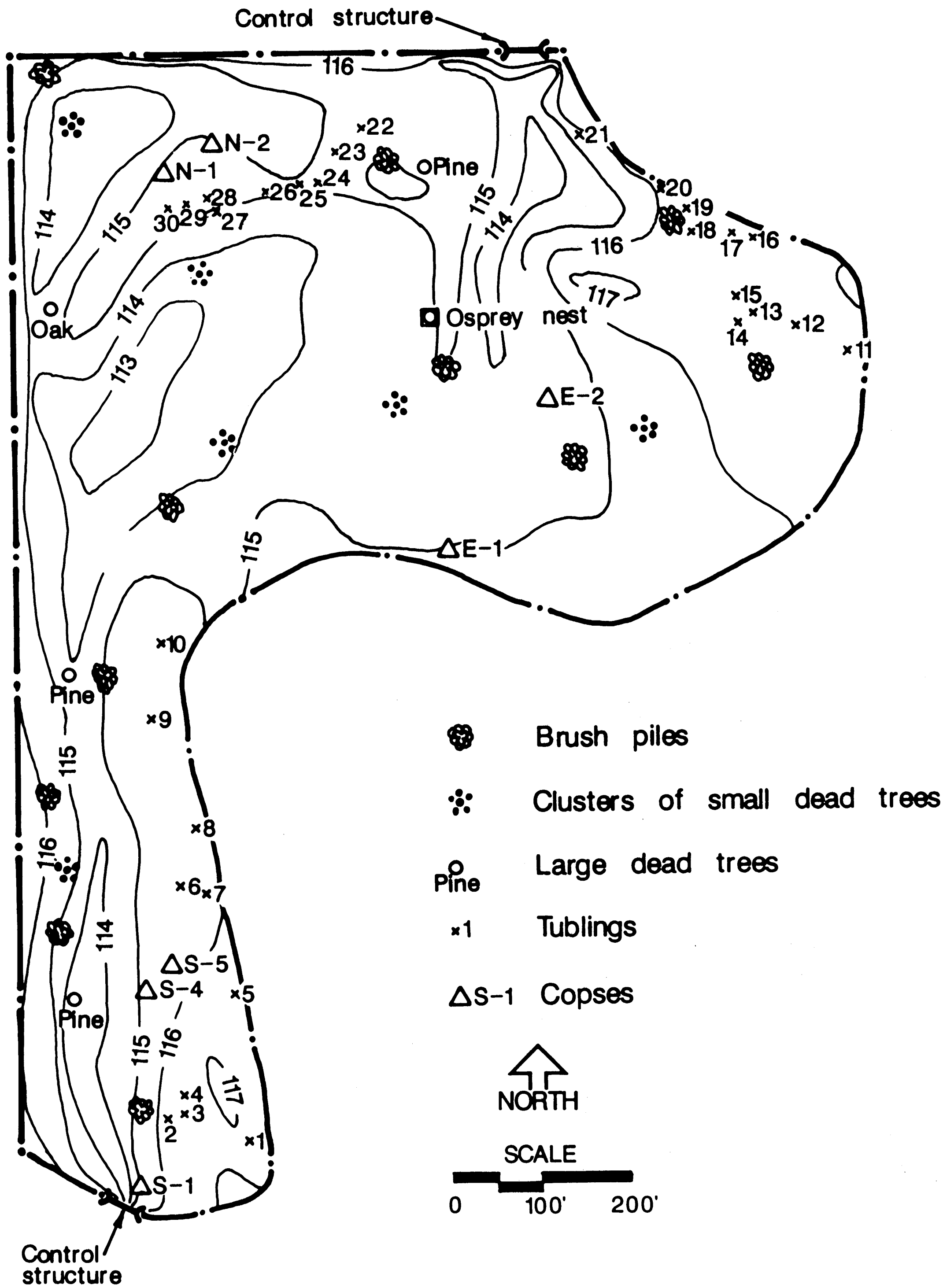

Figure 3. Finish contours in Area I showing other features installed to encourage wildlife. Note the absence of a planned channel below the outfall.

were put into place using marked, tall PVC stakes. The monitoring program was planned to incorporate early season growth status (May) and late growing season diversity and size (September).

RESULTS

The first year drought and flood proved fatal for most of the tublings. Survival was much better from the gallon sized specimens. Replanting of some areas was undertaken with gallon sized stock in spring, 1986 with good success. Natural recruitment from the upstream abandoned spoils area and the nearby floodplain was almost immediate. Diversity increased from 46 species at the end of the first growing season (1985) with the passage of time until after three years the total list of plants found on site comprises 145 species (Table 1). We have observed that some succession has occurred and that generally the biomass of exotic noxious species has diminished while that of native taxa has increased.

Even though growth results are uneven across the total area, the number of trees surviving has reached the target average of 200 per acre (494 per hectare). Generally in the "upland" sites the growth has been slower than in lower locations adjacent to the water where short-term flooding is relatively frequent. Specimens planted in areas which were soon overcome with cattails were slowed (many were thought to be lost), but we are now finding many of them rising sufficiently high in the cattail overstory to be seen.

After not quite three full growing seasons, we have produced cypress trees approaching six meters tall from gallon size transplants originally about one meter tall. Red maples, green ash and black gum have reached four meters and all three have produced fruits. The other tree species have shown less spectacular growth but seem to be well established. Some trees were temporarily overgrown and occasionally bent over by para grass but now are rising above it showing strong growth.

We have continued to follow the development of vegetation in our experimental plots which were so severely ravaged by hogs. The areas which received the different topsoiling treatments and which showed significant differences during the first year are now indistinguishable in floristic and vegetational composition. Raw overburden proved to be the least successful substrate of all with diversity slightly reduced and density of cover much reduced as compared to other soil regimes. Slightly upland from the pickerel weed and cattail assemblages the experimental plots were invaded by Carolina willow (*Salix caroliniana*) which is now four to six meters tall. Beneath the expanding willow crowns the cattails were thinned or absent in areas previously very densely covered. Willow canopy closure clearly limited the successful growth of weeds which are normally pioneer species requiring full light. Despite the hog damage to soil and the tubling population, some tublings survived and are now growing vigorously to two to three meters

Table 1. Plant species of apparently volunteer origin encountered within Area I. All species encountered during site visits are noted. x = species present in 1985; o = species not found in fall 1987 or spring 1988 monitoring.

SPECIES	SPECIES
Acer rubrum	*Emilia coccinea* (o)
Aeschynomene americana (x)	*Emilia fosbergii*
Alternanthera philoxeroides	*Eragrostis* sp.
Ambrosia artemisifolia	*Erechtites hieracifolia* (o)
Andropogon glomeratus	*Eupatorium capillifolium* (x)
Andropogon virginicus	*Euthamia minor* (x) (o)
Aster caroliniana	*Fimbristylis* sp.
Aster tenuifolius (o)	*Fraxinus caroliniana*
Axonopus affinis (x)	*Froelichia floridana*
Baccharis halimifolia	*Galactia elliottii* (o)
Bacopa caroliniana	*Heterotheca subaxillaris*
Bacopa monnieri (x)	*Hydrilla verticillata*
Bidens alba	*Hydrochloa cariniensis*
Boehmeria cylindrica	*Hydrocotyle umbellata* (x)
Boltonia diffusa (o)	*Hypericum cistifolium*
Brachiaria mutica (x)	*Hypericum tetrepetalum*
Canna flaccida	*Hyptis alata*
Carex albolutescens	*Indigofera hirsuta* (x)
Cassia nictitans (x) (o)	*Juncus dichotomus*
Centella asiatica (x)	*Juncus effusus* (x)
Cephalanthus occidentalis (x)	*Juncus elliottii*
Chamaecrista fascicularis	*Juncus marginatus*
Cicuta mexicana	*Juncus megacephalus*
Cirsium horridulum	*Juncus scirpoides*
Commelina diffusa (x)	*Leersia hexandra*
Conyza canadensis (o)	*Lemna* sp. (x)
Coreopsis leavenworthii (o)	*Leptochloa fascicularis* (x)
Cyonodon dactylon (x)	*Leptochloa uninervia*
Cyperus esculentus	*Lindernia grandiflora*
Cyperus globulosus	*Lippia nodiflora*
Cyperus haspan	*Ludwigia arcuata* (x) (o)
Cyperus odoratus (x)	*Ludwigia hirtella*
Cyperus pseudovegetus	*Ludwigia leptocarpa* (x)
Cyperus retrorsus	*Ludwigia linearis* (x)
Cyperus surinamensis	*Ludwigia maritima*
Digitaria bicornis	*Ludwigia octivalis* (x)
Digitaria sp.	*Ludwigia peruviana* (x)
Diodia virginiana	*Ludwigia pilosa* (x)
Echinochloa colonum (x)	*Ludwigia repens* (x)
Echinochloa crus-galli (x)	*Lythrum alatum*
Echinochloa walteri	*Mikania scandens* (x)
Eclipta alba (x)	*Momordica charantia* (o)
	Myrica cerifera

Table 1 (continued)

SPECIES	SPECIES
Eleocharis baldwinii	Panicum anceps
Eleocharis sp.	Panicum dichotomiflorum
Eleusine indica (x)	Panicum ensifolium
Panicum hemitomon (x)	Rhynchelytrum repens (x)
Panicum repens	Rhynchospora decurrens
Panicum rigidulum	Rhynchospora fascicularis
Paspalum acuminatum	Rhynchospora wrightiana
Paspalum boscianum	Sabatia brevifolia
Paspalum distichum	Sabatia grandiflora
Paspalum notatum	Sagittaria graminea (x)
Paspalum urvillei	Sagittaria lancifolia (x)
Phoebanthus	Salix caroliniana (x)
grandiflora (x) (o)	Saururus cernuus
Physalis sp. (x)	Scirpus californicus
Phytolacca americana	Scirpus validus
Pluchea camphorata	Scoparia dulcis (x)
Pluchea odorata	Sesbania emerus (x) (o)
Pluchea rosea (o)	Sesbania exaltata
Pluchea sp. (x) (o)	Setaria geniculata
Polygala nana (o)	Solanum americanum
Polygala rugellii	Solidago curtisii (o)
Polygonum densiflorum (o)	Solidago fistulosa (x)
Polygonum hirsutum (o)	Solidago stricta (x) (o)
Polygonum	Sonchus oleraceus (o)
hydropiperiodes (x)	Spartina bakeri (x)
Polypremum procumbens (o)	Stillingia sylvatica
Pontederia cordata (x)	Typha augustifolia (x)
Pterocaulon virgatum	Typha domingensis
Ptilimnium capillaceum (o)	Typha latifolia
Rhexia nuttallii	Ulmus americana
Rhus copallina	Urena lobata (x)

tall under the willows. We checked other sites including an experimental area with strictly a sand-clay mix top soil. The four year old naturally recruited willows in the sand-clay area formed a canopy with a very open understory and a good organic ground litter. Volunteer young Florida elms and red maples were observed up to about four meters tall growing under the willow canopy. Colonies of wetland ferns were established as well.

Wherever canopy closure was observed in any of the sites, the understory vegetation was clearly different from that of exposed areas. Weedy species were absent or few and an epiphytic bark community had started to develop.

DISCUSSION AND CONCLUSIONS

The much higher survival rate of gallon sized planting stock seems to correlate with flooding duration. Larger plants survived when tublings did not. Where flooding was shallow and the terminal bud was not covered, survival was good. Where the terminal bud was flooded for more than a few days, the plants died. This indicates that in the development of a controlled hydrologic regime during the establishment period, careful consideration must be given the duration and depth of flooding relative to the terminal bud.

Survival of gallon size specimens was much higher under conditions of drought as well. In this case, the larger root mass is probably a critical factor. Another is the presence of good water holding capacity in potting soils versus the tightly compacted but often porous overburden soil.

We suspect that succulent tublings are more desired for food by hogs and small rodents than larger stock. Deer browse on young trees has not been observed with certainty so far, but some tip damage may be from such browsing. In the early stages of recreation of wet woodlands removal of large herbivores is obviously important.

Recruitment of desirable native species from surrounding areas into the reclamation area occurred quickly. New taxa are introduced as the soil and microclimatic conditions change under the growing trees. The trends of succession to floodplain forest are clearly established once even small areas such as individual copses of native canopy species produce closed canopy. Barring catastrophic destruction of the canopy formers or a drastic change in hydrologic regime, it seems clear that a healthy and diverse floodplain forest can be expected.

In our work of restoring or recreating wetlands, especially in relation to mining, we have strived to accelerate the normal process of vegetational succession to forest. We have observed over many years that weedy herbaceous stages may linger for a long time. We suspect that is why cattails and primrose willow are considered to be undesirable invaders. Both cattails and primrose willows are obligate bright light plants. Even the comparatively light shade of willows soon causes thinning of both, and neither can compete with shade tolerant plants. Seedlings of floodplain forest hardwoods are shade tolerant, if not umbrosophilic, and seem to make more growth where evaporative stress is reduced. Inasmuch as the woody seedlings are the key to setting the trend to woodlands, their early establishment effectively eliminates several slowly changing early successional stages.

We have concluded that for peninsular Florida the common Carolina willow is a pioneer wetlands species which may allow us to bypass attenuated herbaceous phases in recreation of wetlands. By its shade it can limit the extent of cattail monocultures and primrose willow thickets. It can encourage the growth of transplanted stock of native canopy species. And, because it is not shade tolerant, it is a transitional species in the process. As a practical matter, we are planting fresh-cut willow cuttings up to 2 m or more long by 2-3 cm in diameter in cattail marshes with at least a ninety percent willow survival. As the canopy develops we expect the cattails to thin so that hardwoods can establish. We are also planning to plant hardwood copses directly in sand-clay mix and then to interplant Salix as a nurse species to discourage competing weedy species and encourage hardwoods. We expect to be able to report on the progress of our systems so established after another growing season or two.

ACKNOWLEDGEMENTS

We thank CF Mining Corporation for their continued support of the research projects. Also, staff from the Florida Department of Environmental Regulation have been both helpful and encouraging.

LITERATURE CITED

Miller, H. A., J. G. Sampson, and C. S. Lotspeich. 1985. Wetlands reclamation using sand-clay mix from phosphate mines. In F. J. Webb (Ed.), Proceedings of the 12th Annual Conference on Wetlands Restoration and Creation (pp. 193-200). Hillsborough Community College, Tampa, Florida.

EXPERIMENTAL PLOTS OF *HALODULE WRIGHTII* IN A CLOSED, AERATED CANAL SYSTEM (LITTLE TORCH KEY, FLORIDA)

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ABSTRACT

A number of canal systems in the Florida Keys are closed to surrounding waters. Such a canal system exists within the Coral Shores Estates on Little Torch Key, in the Lower Florida Keys. Primarily because of canal depths up to 8 meters, DER-mandated plugs have separated this canal system from the waters of Pine Channel. The plugs have varied in their integrity, thus, numerous fish, invertebrates and plants have entered the canal system. To eliminate stratification and improve water quality, the second author (MS) initiated a program of aeration and monitoring in 1985. Oxygen and temperature have been measured regularly; a number of other parameters have been measured at various times. Several biological surveys have been conducted, including complete seagrass counts by the first author (EM). When the plug was partially open, there was recruitment of the three major species of grass. A distinct decrease in grass density (all species) was found as one moved away from the partially open plug. Experimental plots of *Halodule wrightii* were designed to test whether the observed distribution of seagrasses was due to increased recruitment from Pine Channel or because of higher water quality near the opening. The plants survived for several months and then declined in health and numbers, along with the natural population. During this period, the plug was restored and the seagrass population fell to zero. Potential factors responsible for the loss of seagrasses are considered.

INTRODUCTION

Extensive dredging of canal systems occurred in the Florida Keys during the 1960s and 1970s. Many areas of fringing mangrove forests and adjacent shallow water wetlands were displaced by the creation of the resulting subdivisions. With a heightened environmental awareness during the 1970s, there was an increase in the enforcement of dredge and fill regulations as well as the addition of new standards. A variety of actions have been taken against developers that dredged without permits or in violation of their granted permits. Measures included canal closure, or denial of permits for opening, and mitigative restoration of bay bottom and vegetation.

Coral Shores Estates (see Figure 1) is a subdivision that has been required to backfill canals to restore bay bottom. Some 4,000 mangroves were planted in the restored area. Of the three remaining canal systems, one straight canal and a long, L-shaped canal are open to the adjacent waters of North Pine Channel. The canals of the third canal system remain separated from the adjacent waters by a plug. The work described here was undertaken within this closed canal system with comparative monitoring in the L-shaped canal (referred to hereafter as the "open" canal).

In an effort to improve water quality and reduce stratification, the second author (MS) began the installation of aerators in the closed canal system. Eleven aerators have been emplaced since 1985 and the temperature and dissolved oxygen levels monitored. Although numerous studies have examined the effect of aeration in freshwater lakes (Pastorok et al. 1980), little work has been published concerning the efficacy of aeration in saline waters.

In a further attempt to assess and improve the water quality of the closed canal system, the first author (EM) surveyed extant seagrass populations and introduced experimental seagrass plots in 1986. The success of seagrass growth can be used as one indication of water quality; their growth and active photosynthesis contributes greatly to

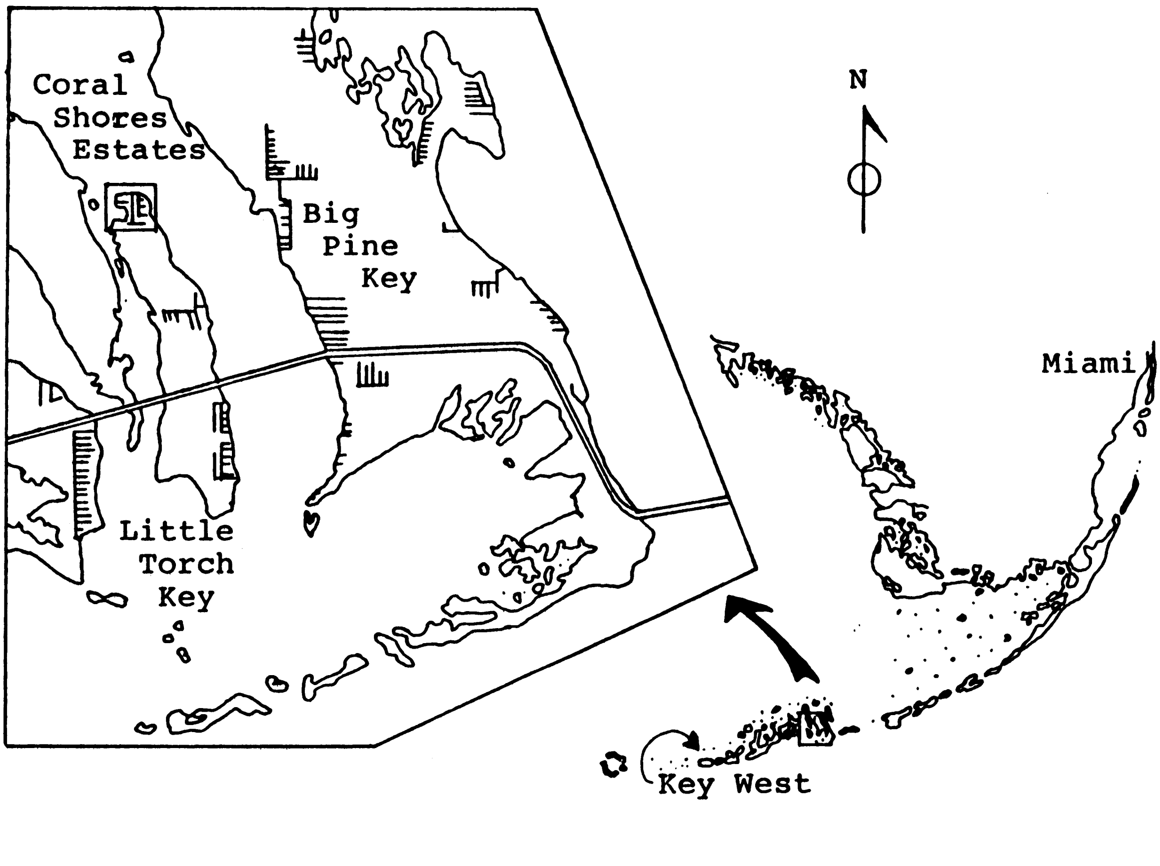

Figure 1. Site location of Coral Shores Estates.

the establishment of a stable trophic structure (Zieman 1982). *Halodule wrightii* was selected for the experimental plots because it is generally the primary colonizer in the successional sequence leading to a *Thalassia testudinum*-dominated community (den Hertog 1977; Zieman 1982) and has successfully been used in transplantation efforts (Fredette et al. 1985).

MATERIALS AND METHODS

Aeration

The aerator units were obtained from Area, Inc. of Homestead, Florida. The units have a PVC grid which supports four air stones. Weighted hoses supply air to the aerators and are supplied by an electrically-powered compressor (Model 727 AM, 1/4 hp, 3 cfm; Thomas Industries, Inc., Sheboygan, Wisconsin). One compressor was used per aeration unit. The current deployment of eleven aerators is shown in Figure 2. The first unit was installed 27 October 1985. Subsequent aerators were added on the dates shown in Figures 3a and 3b. Periodically, the compressors are serviced and the air stones recovered for muriatic (hydrochloric) acid cleaning.

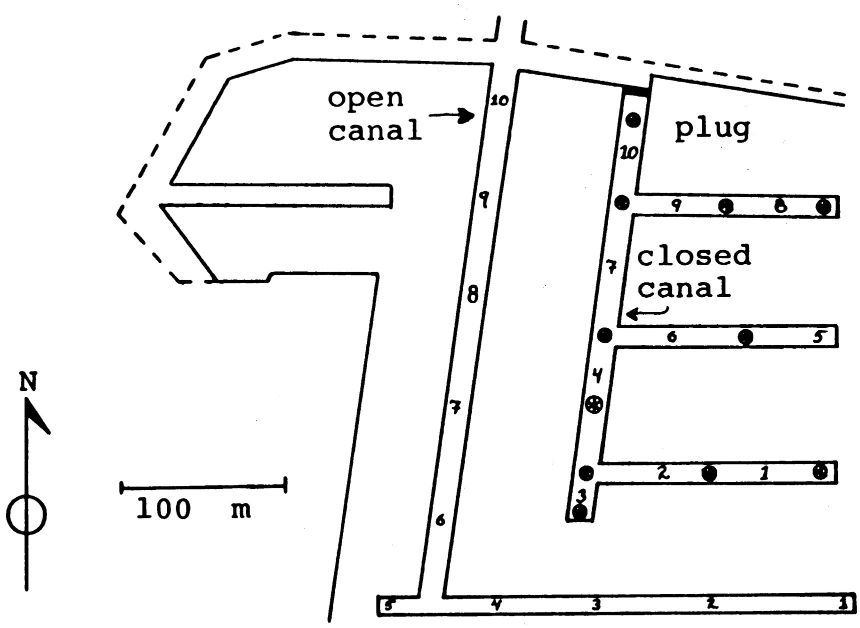

Figure 2. Canal systems of Coral Shores Estates showing monitoring stations (numbers) and aerator locations (stars). Area of scale bar has been restored to bay bottom.

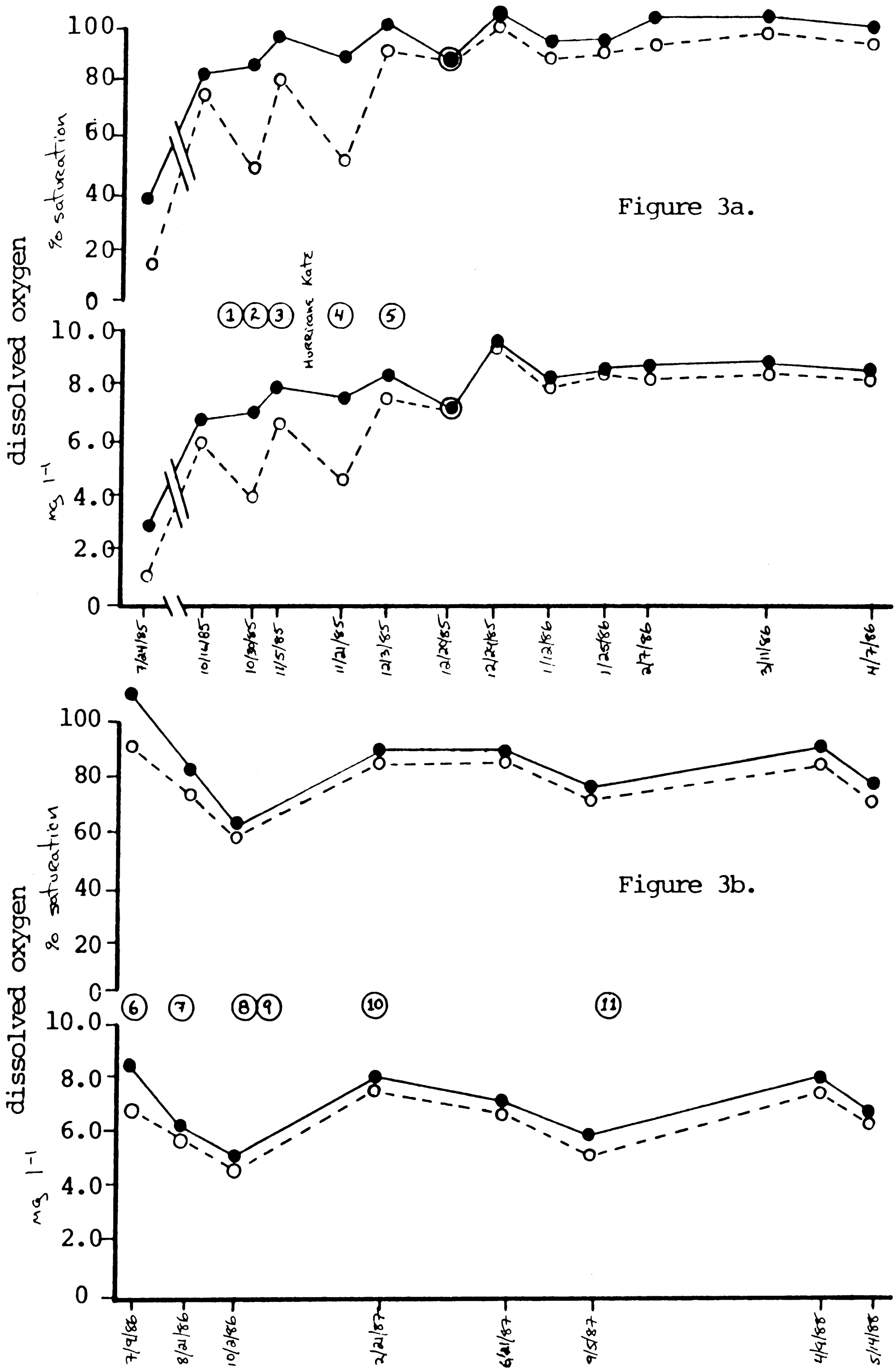

Figure 3. Oxygen levels in Coral Shores Estates closed canal system. Solid symbols indicate surface and open symbols indicate bottom values. Circled numbers represent when aerator units were added.

Oxygen and Temperature Monitoring

Monitoring has been carried out in Coral Shores Estates by state and federal agencies, private consultants and by the authors. Most of the data presented were obtained by MS using a YSI Model 57 oxygen meter. The instrument was air calibrated prior to each sampling. Measurements by others were used as a basis for adopting a standard salinity of 35 ppt. Readings were made just below the surface, at the mid-depth and about 20-30 cm from the bottom. The depths in both the open, L-shaped canal and the closed canal were 6-8 m. The surface, mid-depth and bottom figures are presented as the average of the readings from all stations in the canal system. Measurements were made halfway between aeration units (at least 15 meters from either aerator).

In Figure 3, the first baseline oxygen values (prior to aeration) were obtained by the Florida Department of Environmental Regulation (DER) on 24 July 1985 using five stations. The second baseline record was obtained by Dr. Brian Lapointe on 16 October 1985, also using five stations (three of which corresponded to those used in the DER survey). The five stations used by Dr. Lapointe were also used by the author (MS) until August 1986 when five more stations were added for a total of ten. The present station locations are shown in Figure 2.

An experiment was conducted in the secondary canal closest to the primary canal plug to assess the performance of the aerators. On 23 April 1988, the two aerators in this canal were shut down. Oxygen and temperature were monitored on a daily basis, weather permitting. The values from Stations 8 and 9 were averaged to compare with the average of all other stations. The "stratification index" (SI), used in Figure 4 and Table 1, was derived by subtracting the surface oxygen value (as percent saturation) from the bottom value. A negative value indicates lower oxygen on the bottom. The vast majority of mid-depth oxygen readings were intermediate between the surface and bottom values. The SIs were also derived from temperature records and follow the trends of the oxygen records. The variations are smaller than seen with oxygen and are not presented here.

Seagrass Survey

Naturally-occurring seagrasses were surveyed in the closed canal system on 14 September 1986. The canal system was divided into sections (Figure 5) for averaging purposes. All individuals of *Thalassia testudinum*, *Syringodium filiforme* and *Halodule wrightii* were recorded. No other seagrass species were observed. Plastic straws were used to temporarily mark plants in areas of high density to prevent double counting.

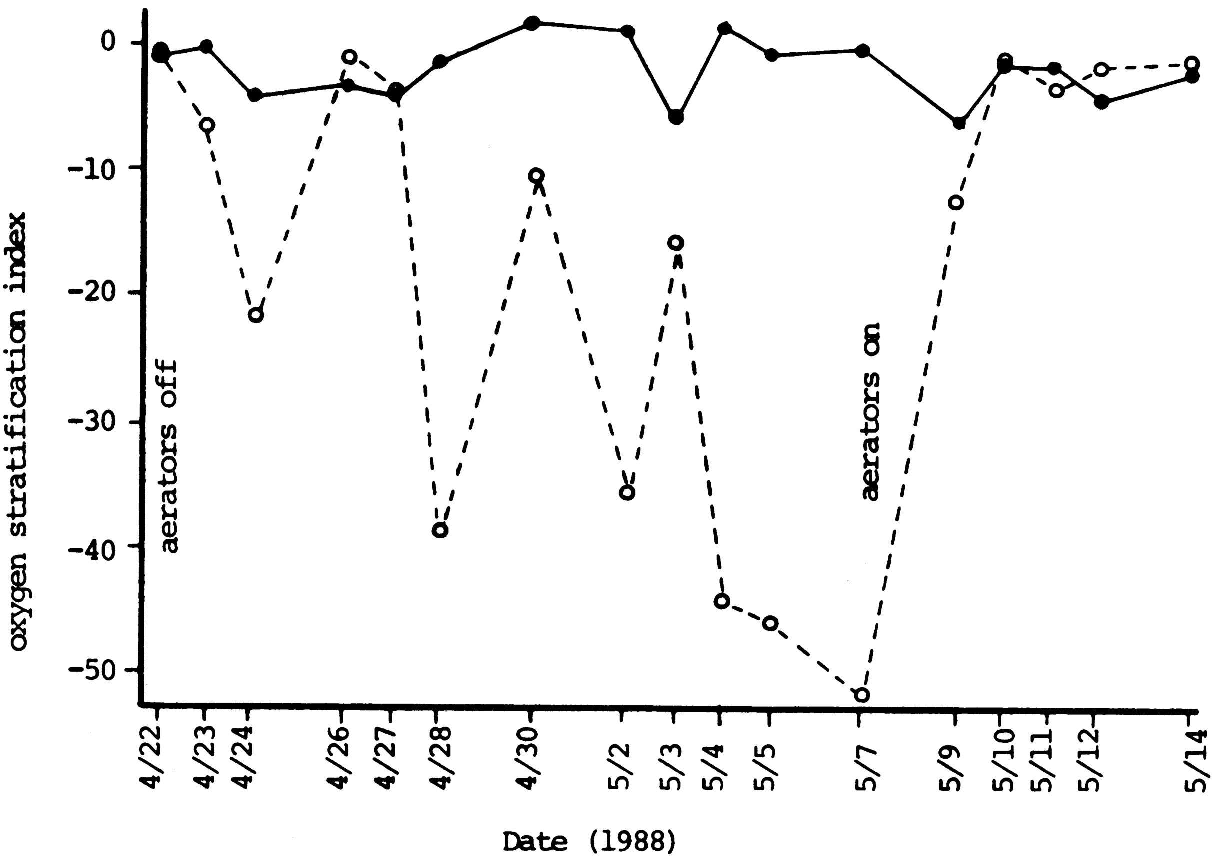

Figure 4. Aerator shut-down experiment. Closed symbols indicate mean of control stations (1-7 and 10). Open symbols indicate mean for experimental stations (8 and 9). See text for details.

Seagrass Planting

Halodule wrightii was selected for planting because of its success as a pioneer species. The method of Derrenbacker and Lewis (1982) was used. Terminal rhizomes with six shoots (turions) were collected from South Pine Channel. The turions were placed into a cooler and aerated during transport; planting occurred on the same day as collection. Two plots were established as shown in Figure 5. Plot 1 contained 36 plants in a 6 x 6 array. Plot 2 was set out in a 5 x 5 array, however, only 22 plants were emplaced. The turions were secured to the bottom with "staples" made of stainless steel wire. Staples were recovered after the roots had established themselves. The plots were reexamined at approximately monthly intervals and the plants counted.

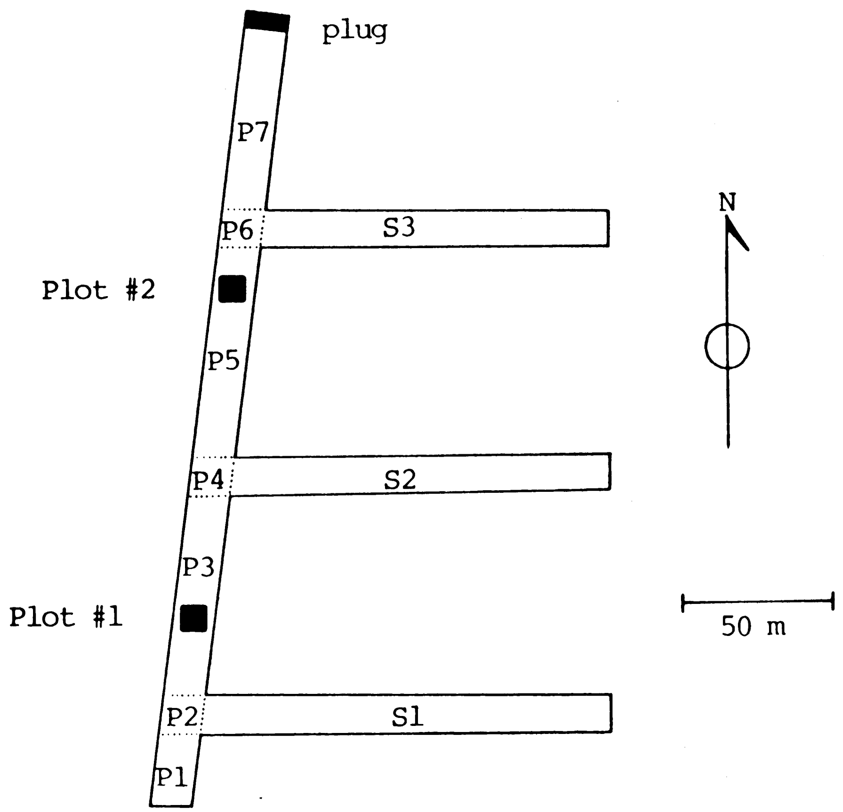

Figure 5. Subsections of closed canal system used for seagrass survey and location of *H. wrightii* planting sites.

RESULTS

Effect of Aeration on Oxygen Levels and Stratification

The percent saturation and concentration of dissolved oxygen prior to and after initiation of aeration are shown in Figure 3a. The DER survey on 24 July 1985 found bottom oxygen levels in the closed canal system to be as low as zero. The average bottom reading was 1.0 mg/l (12.9% saturation) and the average surface value 2.9 mg/l (37.2% saturation). The derived stratification index (SI; see Methods) was minus 24.3. The situation had improved by 16 October when Dr. Lapointe found a mean bottom oxygen level of 6.1 and a surface mean of 6.4 mg/l (% saturation = 77.9 and 82.1% respectively; SI = -4.1).

The first aerator was operational on 27 October 1985 and oxygen levels were recorded on 30 October. The disparity between surface and bottom oxygen readings indicates that stratification was present. The oxygen levels were higher and stratification reduced on 5 November by which time three aerators were on line. Hurricane Kate passed near the

Lower Keys on the 13th of November, disrupting power for three days and causing a breach in the plug. Stratification was again apparent in the oxygen records after the hurricane on November 21. By 3 December 1985, five aerators were established throughout the closed canal system. Although the oxygen levels varied, there is little evidence of stratification after this time.

Figure 3b illustrates the seasonal variations in oxygen levels from July 1986 through May 1988. The bottom oxygen value is usually lower than that of the surface, however, the stratification index remains less than -7.0 with one exception (9 July 1986; SI = -20.25). Compare with the SI following Hurricane Kate (when aerators were off for several days) which reached -37.32. During this period of monitoring, the plug was restored (January 1987) and aerators added to assist mixing in certain spots. The 11th aerator was installed during September 1987.

To directly test the effect of aeration, two aerators were turned off in the northernmost secondary canal on 23 April 1988. The oxygen values from that canal are compared with those of the remainder of the canal system in Figure 4. The SI's for the control areas remain fairly constant around zero (a low value of -6.03 was found on 9 May). In the experimental canal, the SI's move up and down but show a clear downward trend. On the 7th of May, the SI reached a low of -51.6. The aeration was restored to the experimental canal after taking the oxygen and temperature readings. Within three days, the SI value had returned to control levels.

Comparison of the Open and Closed Canal Systems

Temperature and oxygen were not regularly measured in the open ("L") canal. Data are presented in Table 1 for dates on which both the open and closed canal system were monitored. Six stations were monitored in the open canal on 8 February 1986, ten stations on all other dates. Ten stations were used in the closed canal on all dates. The open canal system consistently has a stratification index that is very low. The closed canal system had a low, on these dates, of -6.4. This is consistent with the general monitoring records; the SI typically varies between -7.0 and zero. The measurements listed under the "Bay" column were made in adjacent waters over undisturbed bay bottom with a depth of 1.0 to 1.5 m. The low SI observed on 23 April 1988 may have resulted from measurements made before incoming tidal waters could mix with lower layers.

Seagrass Survey

On the 14th of September 1986, all three major species of seagrasses endemic to the Florida Keys were present in the closed canal system. With the exception of section P5 (see Figure 5 for section locations), the seagrass densities in the primary canal exhibit a clear

Table 1. Percent saturation of dissolved oxygen in open and closed canal systems. S = surface, M = mid-depth, B = bottom, SI = stratification index (see Methods), ND = no data.

Date	Depth	Open	Closed	Bay
2/8/86	S	106.7	102.6	ND
	M	90.2	102.3	ND
	B	75.1	96.1	ND
	SI	-31.6	-6.4	ND
4/23/88	S	72.3	82.6	100.0
	M	49.4	81.5	ND
	B	29.3	82.3	80.0
	SI	-42.9	-0.3	-20.0
5/2/88	S	59.3	75.2	84.7
	M	47.2	74.9	--
	B	16.9	76.0	84.7
	SI	-42.4	0.8	0.0
5/12/88	S	47.7	78.9	92.0
	M	29.2	78.2	--
	B	18.6	76.6	92.0
	SI	-29.1	-2.3	0.0

gradient with the highest concentration of grasses near the plug (Figure 6). The plug was not intact at this time, having been breached by waves during Hurricane Kate some 10 months earlier. At high water, approximately 30-40 cm of water could be found over the damaged plug.

There is no gradient with respect to the plug that can be observed in the secondary canals which may be related to variations in the amount of suitable substrate; considerable amounts of concrete bridge rubble had been placed in the secondary canals. _T. testudinum_ was observed in all secondary canals, _S. filiforme_ in canals S1 and S2 and _H. wrightii_ was observed only in S1. A total of 21 plants were observed in S1, 22 in S2 and only 1 in _T. testudinum_ in S3 (which has the most concrete rubble).

The plug was restored in January of 1987, thus, completely preventing the mixing of canal waters with those of North Pine Channel. The natural seagrass population has since declined with no individuals observed in April of 1988 (243 plants were counted in the September 1986 survey). The open canal was qualitatively examined also on 23

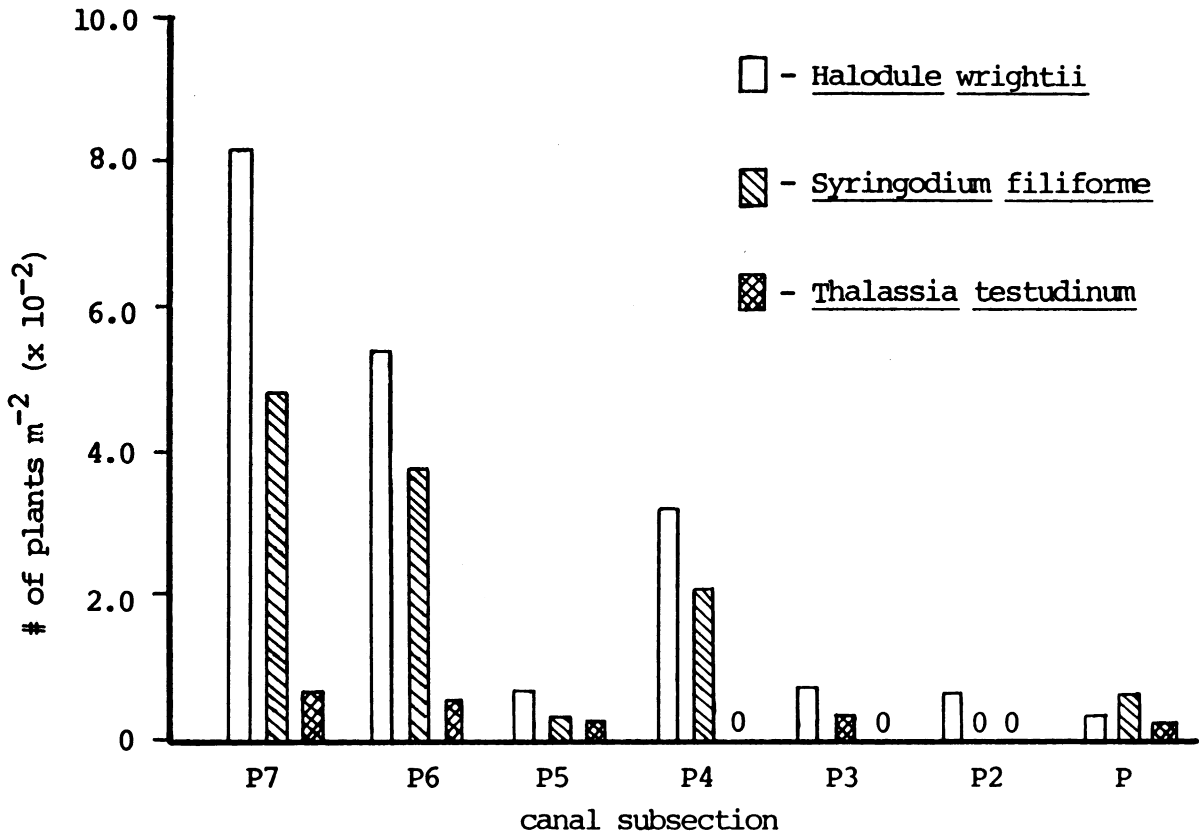

Figure 6. Densities of seagrasses along primary canal of closed canal system on 9/14/86. See Figure 5 for locations of canal subsections.

April 1988. Seagrasses were found along shallow ledges (1-2 m) for about 3/4 of the length of the primary canal. Growth on the bottom (5-7 m) was limited to within 25 m of the canal entrance. A distinct murky layer obscured the bottom below about 4-5 m and was associated with an abrupt thermocline. The mean SI based on temperature for this day in the open canal was -3.92 degrees C. Surface and bottom temperature differences of over 9 degrees C have been recorded in this canal.

Experimental Plots of Halodule wrightii

Observations of the experimental plots are shown in Table 2 (see Figure 5 for plot locations). Plot #1 lost several plants immediately after planting and then held a steady population until December. Plot #2 lost very few individuals until December when mortality and poor plant health was apparent in both plots. The decline was evident prior to restoration of the plug in January 1987 and continued thereafter; only four plants were seen in Plot #2 in June. The loss of natural seagrasses roughly paralleled that of the plots. No seagrasses were observed in April 1988.

Table 2. Survival of *H. wrightii* in experimental plots.

Date	#1	#2	Comments
8/17/86	36	22	turions collected and planted
9/14/86	30	22	plants appear healthy
10/11/86	30	21	plants appear healthy
12/6/86	14	13	reduction in shoots and health
1/87			plug restored
6/21/87	0	4	poor health; natural seagrass population reduced
4/23/88	0	0	no natural seagrasses observed

DISCUSSION

Stratification, as indicated by oxygen concentrations, in the closed canal system of Coral Shores Estates was common prior to the introduction of artificial aeration. During periods of high winds, mixing was sufficient to find little evidence of stratification. This can be seen in the baseline survey of 16 October 1985 (Figure 3) and during the shut-down experiment of April-May 1988 (for example, see data of 4/26 and 4/27; Figure 4). However, during periods of calm weather, both the open and closed canal systems can be expected to have temperature and oxygen stratification.

The artificial aeration has kept the mean temperature and oxygen differences low even during periods of high temperature and low winds when stratification might be expected to be greatest. To obtain fairly uniform results, it appears that a minimum of five aerators were required. The total canal area was 8850 square meters of 1770 square meters per aerator. There were still some "trouble" spots where stratification would be evident, therefore, aerators were repositioned and six more units added (see distribution in Figure 2). Currently, the aerator density is about one per 800 square meters and uniform results are obtained throughout the canal system. Extensive visual stratification has been eliminated, in contrast with the open canal that frequently has a very turbid layer that obscures the bottom for most of the canal's length.

The shut-down experiment (Figure 4) demonstrates the increase in oxygen stratification following cessation of aeration. More dramatically, the resumption of aeration causes a rapid increase in the dissolved oxygen concentration below the surface which is reflected by the increase in the stratification index (surface oxygen values in the experimental canal remained very similar to control areas). Thus, given sufficient aerator density, rapid elevation of oxygen concentration and improved mixing can be achieved.

On 7 April 1986, Enviropact, Inc. surveyed the closed canal system including measurements of nutrients and coliform bacteria. Water quality at all stations was above State standards. Other observations also indicate good water quality. There is a rich epibiota along the canal walls and, since aeration, little evidence of blue-green algal mats on the bottom. About 20 species of fish have been observed including a number that are considered reef species. The fish entered when the canal systems was connected to ambient waters and now remain confined. The presence of the bridge rubble appears to provide a suitable habitat for certain fish and Florida spiny lobster.

Water quality during 1986 was good enough to support the growth of seagrasses. The canal was partially open at that time to natural waters. The observation of a distinct gradient in seagrass density (Figure 6) can be explained by at least two hypotheses. There is a possibility that a parallel gradient in water quality existed with higher water quality near the area of natural water mixing. No such gradient of oxygen concentration or stratification was observed, however, nutrients, sediment quality or other water quality parameters could be responsible for the seagrass distribution. Apparently suitable sediments of sufficient depth are present throughout the canal system but limited in secondary canals S2 and S3 because of concrete rubble. Arguing against a "quality" gradient is the observation that plants found far from the entrance appeared to be as healthy as those near the partial plug.

The second hypothesis is based on a recruitment gradient and assumes uniform water quality throughout the canal system. There is no evidence of vegetative propagation of seagrasses from populations outside of the plug occurred, thus, the plants within the closed canal system have apparently originated from seeds. Limited water movement would reduce the number of seeds that reached the distal portions of the canal system resulting in the observed density gradient.

The question remains as to why the natural and planted seagrasses have not survived. The decline in health and numbers of the planted *H. wrightii* was observed about one month prior to complete reclosure of the canal system. Cold weather is one possibility, reducing the viability of plants not fully established. In the shallow bays around Little Torch Key, water temperatures occasionally reach lows of 15 to 16 degrees C, a temperature which should not be fatal to healthy seagrasses if not prolonged.

The closure of the canal system may have exacerbated the decline in seagrass numbers. Many herbivores remain trapped within the canals. Loss of more temperature-sensitive algae may have increased grazing pressure on the seagrasses. At least a dozen rainbow parrotfish (Scarus guacamaia) have been observed as recently as May 1988. Most of the individuals were 50 to 60 cm in length and have been seen eating the seagrasses when they were still present. This hypothesis can be tested by removal of the parrotfish or planting of seagrasses within protective enclosures. If seagrasses still cannot survive, then investigation of other factors will be necessitated.

ACKNOWLEDGEMENTS

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LITERATURE CITED

den Hertog, C. 1977. Structure function and classification in seagrass communities. In C. P. McRoy and C. Helferrich (Eds.), Seagrass Ecosystems: A Scientific Perspective. Marcel Dekker, New York, pp. 89-121.

Derrenbacker, J., Jr., and R. R. Lewis, III. 1982. Seagrass habitat restoration, Lake Surprise, Florida Keys. Proc. Ninth Ann. Conf. Wetlands Restoration and Creation, Hillsborough Community College, Tampa, Florida, pp. 132-154.

Fredette, T. J., M.S. Fonseca, W. J. Kenworthy, and G. W. Thayer. 1985. Seagrass transplanting: 10 years of U.S. Army Corps of Engineers research. Proc. Twelfth Ann. Conf. on Wetlands Restoration and Creation, Hillsborough Community College, Tampa, Florida, pp. 121-145.

Pastorok, R. A., T. C. Ginn, and M. W. Lorenzen. 1980. Review of aeration/circulation for Lake Management. In Restoration of Lakes and Inland Waters. U.S. Environmental Protection Agency, Washington, D.C., USEPA 440/5-81-010, pp. 124-133.

Zieman, J. C. 1982. The ecology of the seagrasses of South Florida: A community profile. U.S. Fish and Wildlife Service, Washington, D.C., FWS/OBS-82/25, 158 pp.

THE DISTRIBUTION AND AREAL EXTENT OF COASTAL WETLANDS IN THE GULF OF MEXICO, USA

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ABSTRACT

The National Oceanic and Atmospheric Administration is currently developing the first comprehensive data base describing the areal extent and distribution of coastal wetlands in the conterminous USA. These data are being developed using a systematic grid sampling procedure on wetland maps produced by the National Wetlands Inventory (NWI) of the U.S. Fish and Wildlife Service. The maps, developed from aerial photography, are generally based on 1:24,000 scale U.S. Geological Survey quadrangles and identify wetland habitats classified using the Cowardin et al. (1979) system. Fifteen habitat types are recorded by NOAA in 18.2 hectare (45 acres) cells from each map, and the data input to a microcomputer for processing and manipulation. Digitized study area boundaries can be intersected with the grid sampled data to produce data summaries and color maps for specific units of interest. This paper summarizes the distribution and areal extent of coastal wetlands of the six states (Texas, Louisiana, Mississippi, Alabama, Georgia, and Florida), 153 counties, and 23 estuarine drainage areas (EDA) in the Gulf of Mexico region, an area comprised of over 1,500 NWI maps (Figure 1).

STUDY SITE

The principal spatial unit for which the wetland data are organized is the estuarine drainage area, or EDA. The EDA is defined as that land and water component of an entire watershed that most directly affects an estuary (NOAA 1985). Figure 1 illustrates the 23 EDAs identified in Volume 1 of the National Estuarine Inventory Data Atlas for the Gulf of Mexico (NOAA 1985). Figure 2 illustrates NWI map availability for these same 23 estuaries.

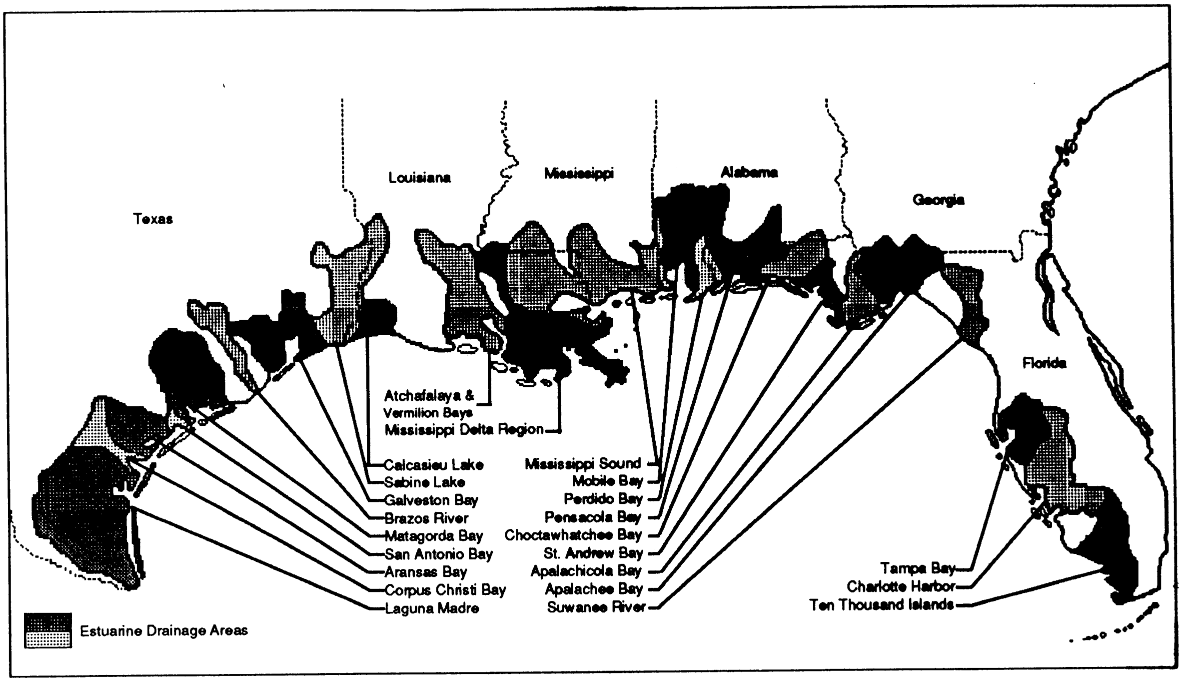

Figure 1. Estuarine drainage areas of the U.S. portion of the Gulf of Mexico.

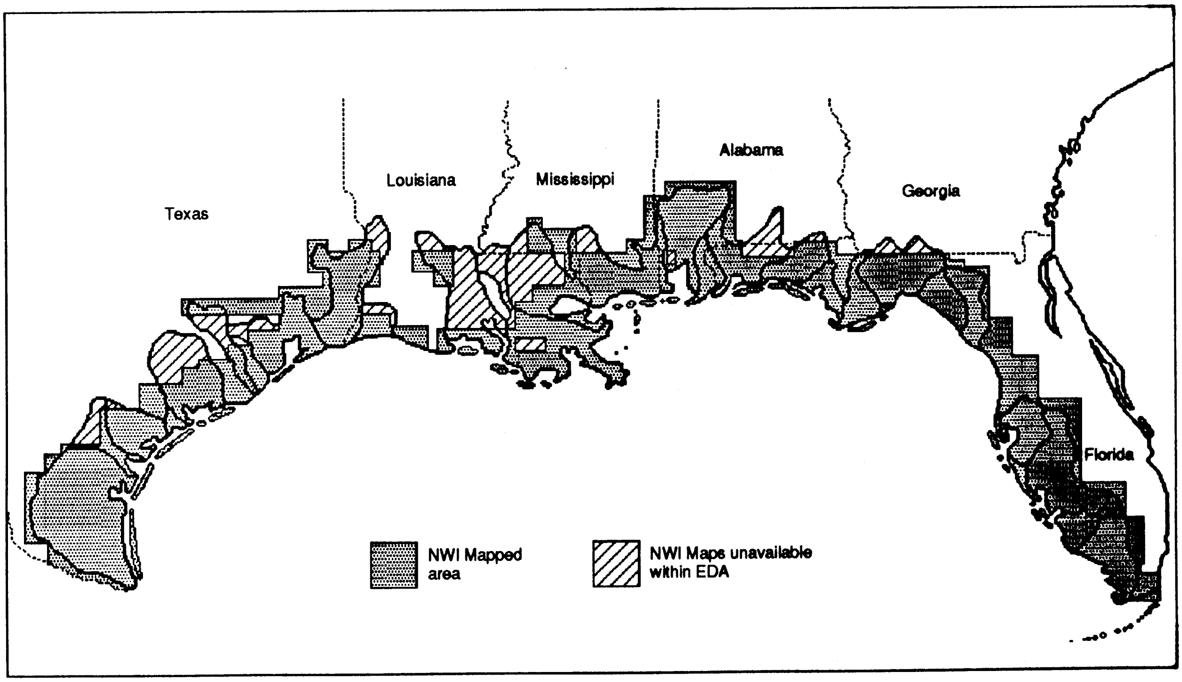

Figure 2. Wetland map availability for the 23 estuarine drainage areas on the USA Gulf of Mexico coast. Two sources of maps were utilized; U.S. Fish and Wildlife Service NWI maps, and, for most of Louisiana, wetland maps prepared for the Louisiana Department of Natural Resources by Coastal Environments, Inc., in Baton Route.

INTRODUCTION

Wetlands are transitional areas between terrestrial and aquatic systems where the water table is at or near the surface or the land is covered by less than six feet of water (Frayer et al. 1983; Cowardin et al 1979). They are vital and irreplaceable natural resources that provide a critical habitat for fish, shellfish and wildlife, filter and process agricultural and industrial wastes, and buffer coastal areas against storm and wave damage. Coastal wetlands also generate large revenues from a variety of recreational activities such as fishing and hunting.

In June 1986, the National Oceanic and Atmospheric Administration (NOAA) initiated a coastal wetland inventory project. The wetland inventory project is being conducted jointly by the Strategic Assessment Branch of the Ocean Assessment Division of the Office of Oceanography and Marine Assessment, National Ocean Service (NOS), and the Beaufort Laboratory of the Southeast Fisheries Center, National Marine Fisheries Service (NMFS), both components of NOAA.

The purpose of this project is to develop a comprehensive and consistently derived national coastal wetlands data base and to improve our understanding and management of this vital resource. The wetlands data developed from this project eventually will be incorporated into NOAA's National Estuarine Inventory and used in conjunction with other information such as land use, coastal pollution, distribution of estuarine fishes and invertebrates, and the status of classified shellfish waters to develop a national estuarine assessment capability. The goal is to build a comprehensive framework for evaluating the health and status of the Nation's estuaries and to bring estuaries into focus as a national resource base. Because wetlands provide an important habitat and food resource for coastal fisheries, their distribution and abundance are of interest to NMFS (Lindall & Thayer 1982). Development of coastal wetlands information is also an integral part of NOS's program of strategic assessments of the Nation's coastal and oceanic region (Ehler & Basta 1984).

This paper describes the areal extent and distribution of coastal wetlands within the U.S. portion of the Gulf of Mexico. This region includes six states (FL, GA, AL, MS, LA, and TX), 153 counties, and 23 estuaries. The wetlands data are based upon available National Wetland Inventory (NWI) maps for this region produced by the U.S. Fish and Wildlife Service. The wetlands data developed in this project pertain only to emergent vegetation and are meant to complement digitized NWI data by allowing rapid organization of data by EDA and county on a national basis. The grid sampling results will also represent a complete data base which extends further inland than digitized NWI data. These maps are based on aerial photography taken from 1972-84. A more detailed presentation of the wetland data for this region will be given in a data atlas to be published in the summer of 1989.

METHODS

As a first step in establishing a coastal wetlands data base, existing data on the areal extent and distribution of coastal wetlands were examined and compiled (Alexander et al. 1986). Twenty-three sources were consulted to compile acreage figures for 242 counties in 22 coastal states. Despite good geographic coverage, much of the existing data is incomplete or outdated. Variability in data quality and consistency and the lack of a unifying theme or purpose also contributed to the difficulty of consolidating the data into a single comprehensive data base. Therefore, the next step was to evaluate alternative sources of information. A key consideration was the ability to develop a data base in a timely and cost-effective manner. Multispectral scanner and thematic mapper Landsat satellite imagery have been successfully used to inventory wetland habitats (May 1986, Haddad & Harris 1985). These techniques, however, are beyond the technical resources of this project. A more timely and cost-effective alternative was to exploit a heretofore under-utilized source of wetland information, the NWI mapping program.

The NWI program was established in 1975 to generate scientific information on the characteristics and extent of the Nation's wetlands (Tiner 1984). This information was to be developed in two stages: 1) the creation of detailed wetland maps; and 2) research on historical status and trends. The maps, developed from aerial photography, are generally based on 1:24,000 scale U.S. Geological Survey quadrangles and identify wetland habitats classified using the Cowardin et al. (1979) system.

Although the NWI wetland maps represent the most reliable source of consistently derived coastal wetland information available, only approximately 1,200 of the over 5,000 maps required for complete coverage of the Nation's estuaries and other coastal areas had been converted to digital data for computer processing and mapping as of January 1986. Therefore, only a fraction of the wetlands data needed for this project were available. Since the current FWS technique for digitizing these maps is expensive and time-consuming, the FWS digitizes maps primarily on a user pays basis and a complete data base is not anticipated or planned for by the FWS in the near future (Tiner, personal communication). NWI maps remained, however, the preferred data source for this project particularly because of their availability across broad coastal regions.

Preliminary tests using a grid sampling technique on NWI maps indicated that this procedure could offer a reasonable alternative to more expensive and time-consuming techniques for quantifying NWI map information with a reasonable degree of accuracy and detail (Field et al. 1988). The grid sampling technique used to quantify coastal wetlands involves the placement of a transparent grid over a NWI map and identification of the wetland type on which each sampling point falls (Figure 3). The grid cells used in this procedure are approximately 1.78 cm (0.7 inch) on a side, corresponding to approximately

18.2 hectares (45 acres) when used on a 1:24,000-scale map. A small dot in the center of each grid cell is used as the sampling point. The exact number of sampling points varies with latitude; maps in the Gulf of Mexico contained 924-990 sampling points.

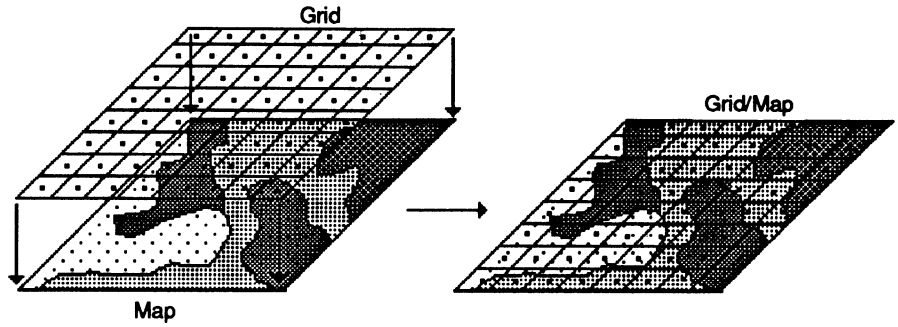

Figure 3. 1:24,000-scale map and grid.

Once the grid is aligned on top of a wetland map, sampling begins in the upper left corner and continues to the right across each subsequent row. Each cell is recorded as the habitat type on which its center dot falls. If the center dot falls exactly on the boundary between two or more habitat types, the cell is counted as the habitat type found in the middle of the square above the center dot. Grid sampled data are entered into a mapping and statistics program on a Sperry microcomputer. The program reproduces grid sampled data in matrix form on a color monitor, with each of the habitat types represented by a different color. Composites of entire estuaries or counties can be displayed using software which overlays digitized boundaries on grid sampled data and illustrates the general distribution of habitat types. Figure 4 illustrates a black and white representation of a portion of the Mississippi Delta EDA (EDA 3:13). The overlaid wetland acreage data can be aggregated by state, county, hydrologic unit and EDA. For the purposes of this technique, the numerous wetland types identified on NWI maps were aggregated into 15 habitat types (Table 1). Table 2 summarizes the FWS categories included in these 15 habitat types and gives examples of typical plant communities found in each habitat.

To determine the effectiveness of the grid sampling technique, grid sampled data for 15 NWI maps in the Mississippi Delta region were compared to NWI digital data (Table 3). These data were developed by NWI using their standard digitizing techniques. The comparisons indicate that common wetland types such as tidal fresh marsh and unspecified salt marsh are estimated extremely well (<1 and -1% difference respectively) while estimates for rare wetland types such as nontidal fresh marsh and estuarine forested and scrub-shrub are generally not reliable for this area (-42 and 59% respectively).

Table 1. The 15 habitat types identified in the grid sampling procedure.

Brackish marsh
High salt marsh
Low salt marsh
Unspecified salt marsh[a]
Nontidal fresh marsh
Tidal fresh marsh
Unspecified fresh marsh[a]
Estuarine forested and scrub-shrub
Nontidal fresh forested and scrub-shrub
Tidal fresh forested and scrub-shrub
Unspecified fresh forested and scrub-shrub[a]
Tidal flats
Non-fresh open water
Fresh open water
Upland

[a]The "unspecified" categories were added to accommodate areas for which more specific information on salinity and water regime was not available.

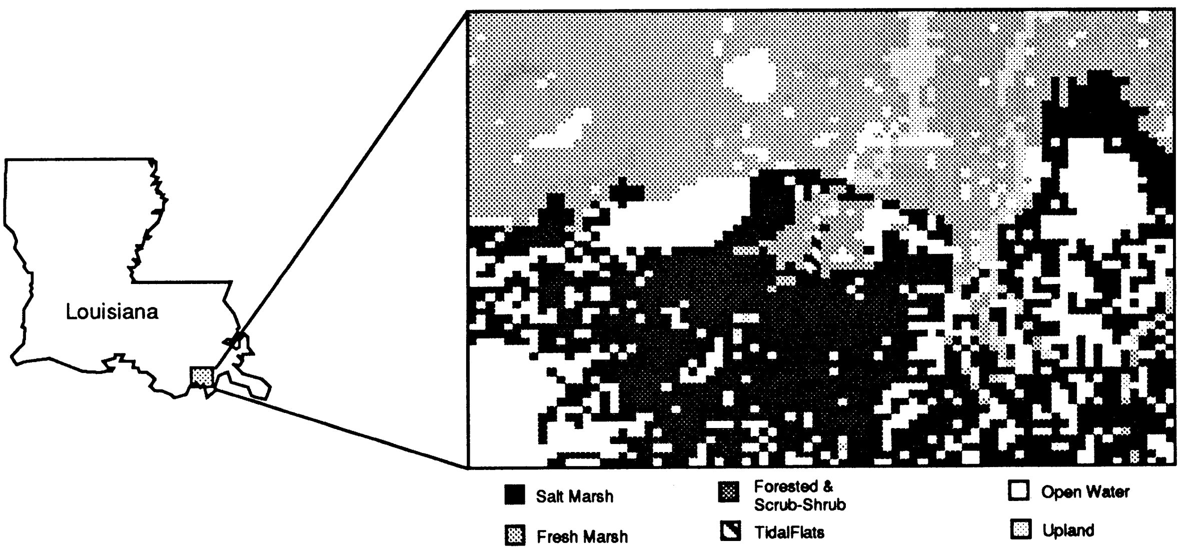

Figure 4. Composite of eight NWI maps in the coastal Mississippi Delta region of Louisiana.

Table 2. Coastal wetlands classification for the Gulf of Mexico, USA.

NOAA	FWS*	Common Plant Community
Salt Marsh		
Brackish	Estuarine intertidal emergent regularly and irregularly flooded salinity $\geq 0.5^{o}/_{oo}$ and $\leq 30^{o}/_{oo}$	black needlerush (*Juncus roemerianus*) salt hay grass (*Spartina patens*) salt grass (*Distichlis spicata*)
High	Estuarine intertidal emergent irregularly flooded salinity $\geq 30^{o}/_{oo}$	black needlerush (*Juncus roemerianus*) salt hay grass (*Spartina patens*) salt grass (*Distichlis spicata*)
Low	Estuarine intertidal emergent regularly flooded or irregularly exposed salinity $\geq 30^{o}/_{oo}$	smooth cordgrass (*Spartina alterniflora*)
Unspecified	Estuarine intertidal emergent	see "Brackish," "High" and "Low"
Fresh Marsh		
Nontidal	Lacustrine littoral emergent nontidal Palustrine emergent nontidal Riverine tidal or lower perennial emergent nontidal	bull tongue (*Sagittaria falcata*) cattails (*Typha* spp.) maidencane (*Panicum hemitomon*)
Tidal	Lacustrine littoral emergent tidal Palustrine emergent tidal Riverine tidal or lower perennial emergent tidal	spike-rush (*Eleocharis* spp.) three-square rush (*Scirpus americanus*)
Unspecified	Lacustrine littoral emergent tidal Palustrine emergent Riverine tidal or lower perennial emergent	see "Nontidal" and "Tidal"

Table 2. Coastal wetlands classification for the Gulf of Mexico, USA (continued).

NOAA	FWS*	Common Plant Community
Forested and scrub-shrub		
Estuarine	Estuarine intertidal forested or scrub-shrub	black mangrove (*Avicennia germinans*) marsh elder (*Iva frutescens*) red mangrove (*Rhizophora mangle*)
Nontidal fresh	Palustrine forested or scrub-shrub nontidal	bald cypress (*Taxodium distichum*) black willow (*Salix nigra*)
Tidal fresh	Palustrine forested or scrub-shrub tidal	same as "Nontidal"
Unspecified	Palustrine forested or scrub-shrub	see "Nontidal"
Tidal flats	Estuarine intertidal Marine intertidal (includes aquatic beds, beach/bars, flats, reefs, rocky shores, streambeds and unconsolidated shores)	saltwort (*Batis maritima*) smooth cordgrass (*Spartina alterniflora*)
Open water		
Fresh	Lacustrine limnetic or littoral Palustrine Riverine (includes aquatic beds, beach/bars, flats, open water, rock bottoms, reefs, rocky shores, streambeds, unconsolidated bottoms and unconsolidated shores)	pond weeds (*Potamogeton* spp.) water hyacinth (*Eichhornia crassipes*)
Non-fresh	Estuarine or Marine subtidal (includes aquatic beds, open water, rock bottoms, reefs and unconsolidated bottoms)	shoal grass (*Halodule beaudettei*) turtle grass (*Thalassia testudinum*) widgeon grass (*Ruppia maritima*)

*Based on Cowardin et al. 1979.

Table 3. Comparison of digital versus grid sampled data for 15 NWI 1:24,000 scale maps in the Mississippi Delta region of Louisiana.

Habitat	Area (Hectares)		
	Digital	Grid	% Difference
Tidal fresh marsh	65,510	65,108	<1
Salt marsh (unspecified)	29,930	28,512	-1
Tidal fresh forested and scrub-shrub	4,951	4,828	-2
Tidal flats	727	765	-5
Nontidal fresh forested and scrub-shrub	375	341	-9
Nontidal fresh marsh	155	90	-42
Estuarine forested scrub-shrub	221	90	-59

RESULTS AND DISCUSSION

The date of aerial photography for the maps used in this study ranged from 1972 to 1984 with 28 percent in 1979 and 42 percent occurring after 1980. The age of these maps must be taken into consideration when interpreting grid sampled data. However, because national trends indicate the abundance of most wetland types are still declining (Frayer et al. 1983), the data in this report are probably overestimates of the current resource. In addition, map availability or the lack thereof will not allow for the complete sampling of the region and therefore affect total acreage estimates. Figure 2 illustrates the NWI map availability for the Gulf at the time of this study. Louisiana was the only state where map availability was relatively poor, with only 256 maps available of the approximately 450 needed for complete coverage of the EDAs and coastal counties within the state.

Where grid sampling estimates indicate that only a small amount of a habitat is present, it does not necessarily mean that it is a rare habitat. On certain maps, due to availability of information or special needs, the FWS provided detailed water regime and water quality labels that indicate very specific wetland types. On adjacent maps, even within the same country or estuary, these labels may not have been available. Consequently, the wetland would be classified as "unspecified" when grid sampled. For example, in Louisiana grid sampled estimates indicate the presence of 2,125 hectares of nontidal fresh forested and scrub-shrub (NFFSS) wetlands and 1,574 hectares of unspecified fresh forested and scrub-shrub (UFFSS) wetlands. A large portion of the UFFSS could be NFFSS, but due to a lack of necessary labels, that distinction could not be made.

A total of 1,543 NWI maps (1:24,000 scale) covering 22.7 million hectares were sampled by NOAA for the Gulf of Mexico (Gulf). Approximately 24 percent, or 5.5 million hectares, were identified as emergent wetlands. Eight of 23 EDAs and 50 of 68 coastal counties sampled had 100 percent map coverage. Fourteen EDAs had greater than 80 percent map coverage, while 57 coastal counties had greater than 90 percent coverage. Forested wetlands were the most common habitat type accounting for nearly 59 percent of the total Gulf wetlands, followed by fresh marsh (19%), salt marsh (18%), and tidal flats (4%).

Of the six states in the region, Florida contained the most wetlands (50% of the total) (Figure 5 and Table 4), followed by Louisiana (24%), Texas (12%), Alabama (8%), Mississippi (5%), and Georgia (<1%). Texas and Florida contained the largest grid sampled areas with 37 and 35 percent of the total Gulf area sampled respectively. Louisiana accounted for only 14 percent of the total due to poor map availability, followed by Alabama (8%), Mississippi (6%), and Georgia (<1%). The central to eastern portions of the Gulf (MS, AL, FL) were dominated by forested wetlands, accounting for over 83 percent of the forested total for the entire Gulf. The coastal areas of the western Gulf (TX, LA) were dominated by salt marsh having 86 percent of the regional total, with the highest concentrations in Louisiana (69%). Texas also contained the largest amount of tidal flats in the Gulf accounting for over 54 percent of the total, while Florida contained 38 percent. Fresh marsh is found throughout the Gulf with its greatest abundance in Florida (53%) followed by Louisiana and Texas (26% and 20% respectively).

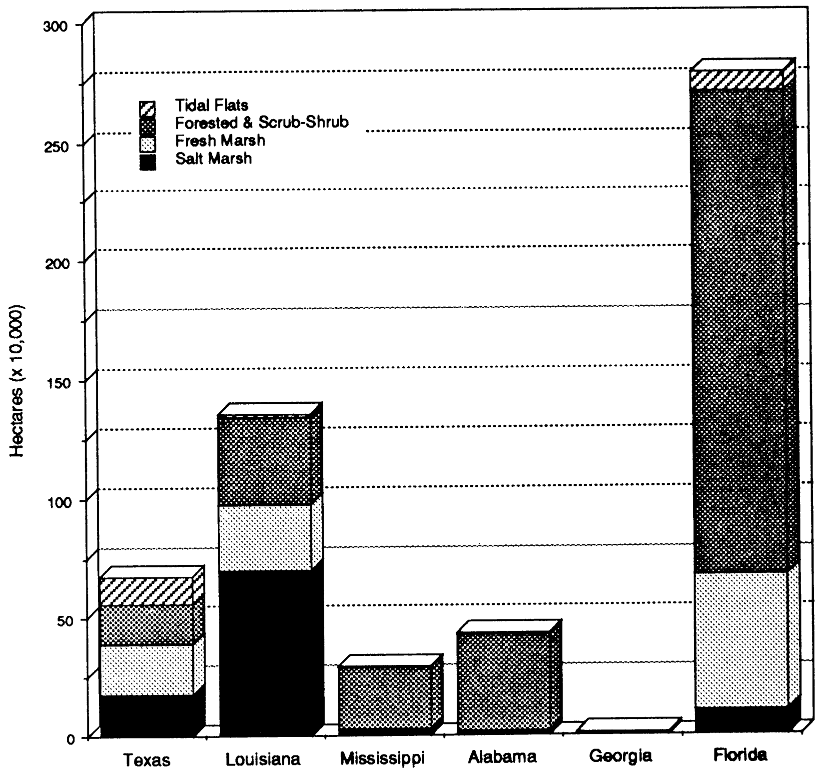

Figure 5. Coastal wetlands by state.

Table 4. Coastal wetlands by state (Hectares x 100).

Habitat	State						
	Texas Totals	Louisiana	Mississippi	Alabama	Georgia	Florida	
Salt Marsh							
Brackish	0	4,606	7	0	0	0	4,613
High	0	0	0	0	0	0	0
Low	0	0	0	0	0	0	0
Unspecified	1,748	2,366	230	103	0	1,029	5,477
Subtotal	1,748	6,972	237	103	0	1,029	10,090
Fresh Marsh							
Nontidal	2,070	308	39	52	9	5,648	8,127
Tidal	91	263	0	1	0	40	395
Unspecified	0	2,168	4	6	0	1	2,178
Subtotal	2,161	2,739	43	59	9	5,689	10,700
Forested and Scrub-Shrub							
Estuarine	11	41	7	11	0	2,484	2,551
Fresh (Unspecified)	<1	1,514	72	3	0	0	1,590
Nontidal Fresh	1,662	2,125	2,547	4,136	135	17,807	28,411
Tidal Fresh	31	19	0	9	0	74	132
Subtotal	1,704	3,699	2,626	4,158	135	20,365	32,684
Tidal Flats	1,113	129	9	17	0	781	2,049
Total Wetlands	6,728	13,539	2,915	4,337	144	27,864	55,523
Non Wetlands							
Open Water Fresh	1,384	1,474	126	206	20	1,387	4,597
Open Water Non-Fresh	4,944	6,689	127	207	0	3,323	15,291
Upland	69,698	7,399	8,379	13,187	926	47,835	147,424
Subtotal	76,026	15,562	8,632	13,600	946	52,545	167,312
Regional Acreage	82,754	29,101	11,547	17,937	1,090	80,409	222,835

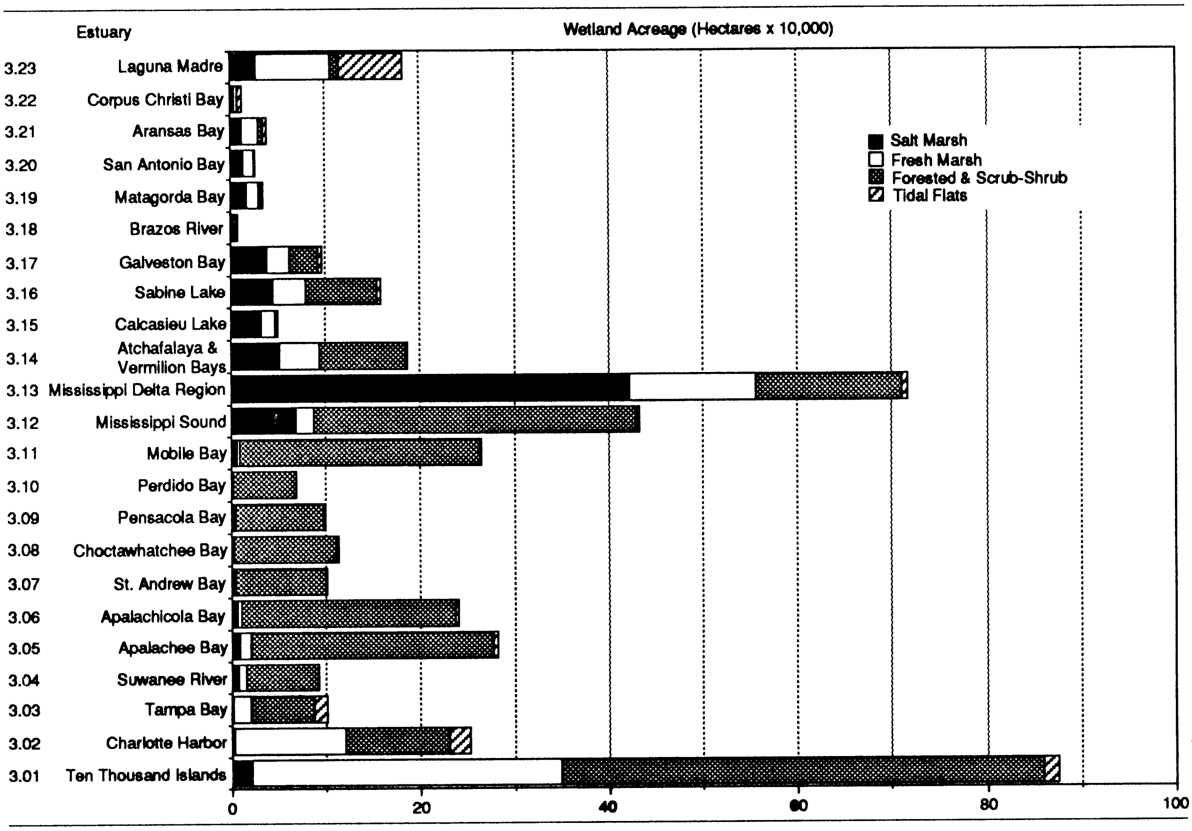

Figure 6. Coastal wetlands by estuarine drainage area.

The abundance of coastal wetlands in the counties of the Gulf follow a pattern similar to that of the states. Collier County, Florida contained the most wetlands of all Gulf counties and ranked first in total forested wetlands. Six counties in Louisiana--Cameron, Terrebonne, Plaquemines, Lafourche, St. Bernard, and Vermillion--accounted for 59 percent of the regional salt marsh total. Kenedy County, Texas ranked first in tidal flats, while Dade County, Florida ranked first in total fresh marsh.

The four major wetland habitat types are summarized by EDA in Figure 6. Laguna Madre in Texas had the largest grid sampled area with 15 percent of the total for the Gulf, however, it contained only 4 percent of the Gulf's total wetlands. It was dominated by fresh marsh and tidal flats (43% and 37% of the total wetlands in the estuary respectively), but only 7 percent of the estuary's total grid sampled area of 2.8 million hectares was wetlands. However, it contained the most tidal flats in the Gulf (40% of the total). Ten Thousand Islands in Florida contained the largest amount of wetlands in the Gulf (20% of the total) and the largest percent of wetlands (over 76%). It ranked first in the Gulf in forested wetlands and fresh marsh, 37 percent of the total fresh marsh and 20 percent of the total forested wetlands. In addition, its forested wetlands accounted for 12 percent of the total Gulf wetlands. Despite low map availability, the Mississippi Delta and Mississippi Sound EDAs combined contained 26 percent of the total regional wetlands. The Mississippi Delta EDA ranked first in total salt march (53% of the regional total). It also contained a large amount of fresh marsh, ranking second (15% of the fresh marsh total). The Mississippi Sound EDA was dominated by forested wetlands that accounted for 13 percent of the Gulf's forested total. The remaining EDAs of the Gulf had a somewhat lower abundance of wetlands due to poor map availability, areal size, and/or geographic location.

The development of these data by NOAA provides an inexpensive and relatively simple method for accurately estimating the abundance and distribution of the Nation's coastal wetlands at a level of aggregation suitable for national assessments. Products from this project will complement the FWS work and provide a useful management tool for coastal resource managers at all levels of government, particularly those Federal agencies with responsibilities for wetlands management and conservation (e.g., COE, EPA, FWS, and NOAA). Baseline data for the Nation's coastal wetlands will be a significant addition to our understanding of these systems and should improve our ability to manage them effectively. In addition, when these data are integrated into the National Estuarine Inventory data base along with other data developed as part of NOAA's Strategic Assessment Program, they will serve as an important component in assessing the overall health and status of estuarine systems.

Wetland reports for the remaining coastal areas of the coterminous USA will be forthcoming in the months ahead. Following this report, a detailed data atlas on wetlands of the U.S. portion of the Gulf of Mexico will be developed by the Spring of 1989. The next region planned for completion is the Mid-Atlantic (New York to Virginia, Summer of 1989). It will be followed by the West Coast (California to Washington, Fall of 1989) and the Southeast (North Carolina to the east coast of Florida, early 1990).

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LITERATURE CITED

Alexander, C. E., M. A. Broutman, and D. W. Field. 1986. An inventory of coastal wetlands of the USA. Strategic Assessment Branch, Rockville, Maryland, National Oceanic and Atmospheric Administration, 25 pp. (mimeo).

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. U.S. Fish and Wildlife Service, Washington, D.C., FWS/OBS-79/31.

Ehler, C. N., and D. J. Basta. 1984. Strategic assessment of multiple resource-use conflicts in the U.S. Exclusive Economic Zone. In Exclusive Economic Zone Papers. Reprint from Proceedings of Oceans '84, Washington, D.C.

Field, D. W., C. E. Alexander, and M. A. Broutman. 1988. Towards developing an inventory of coastal wetlands of the USA. Marine Fisheries Review. (in press)

Frayer, W. E., T. J. Monahan, D. C. Bowden, and F. A. Graybill. 1983. Status and trends of wetlands and deepwater habitats in the coterminous United States, 1950s to 1970s. Colorado State University, Department of Forest and Wood Sciences, Ft. Collins, Colorado.

Gagliano, S. M., K. J. Meyer-Arendt, and K. M. Wicker. 1981. Land loss in the Mississippi River Deltaic Plain. Transactions of the Gulf Coastal Association Geological Society 31:295-299.

Haddad, K. D., and B. A. Harris. 1985. Use of remote sensing to assess estuarine habitats. In O. T. Magoon (Ed.), Proceedings of the fifth symposium on coastal and ocean management - Coastal Zone '85. American Society of Engineers, New York.

Lindall, W. N., Jr., and G. W. Thayer. 1982. Quantification of National Marine Fisheries Service habitat conservation efforts in the Southeast region of the United States. Marine Fisheries Review 44:18-22.

May, L. N., Jr. 1986. An evaluation of Landsat MSS digital data for updating habitat maps of the Louisiana coastal zone. Photogrammetric Engineering and Remote Sensing (52)8.

National Oceanic and Atmospheric Administration. 1985. National Estuarine Inventory data atlas: Physical and hydrologic characteristics. Strategic Assessment Branch, Rockville, Maryland, 103 pp.

Tiner, R. W., Jr. 1984. Wetlands of the United States: Current status and recent trends. U.S. Fish and Wildlife Service, National Wetlands Inventory, Washington, D.C., 59 pp.

MATCHING TREE SPECIES TO SITE CONDITIONS IN RECLAMATION

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ABSTRACT

Diversity and succession on clay settling ponds, a landform left from phosphate mining, can be accelerated by planting small inexpensive tree seedlings. Conditions appear favorable for forested wetland communities. Pioneer species such as ash and red maple were the most successful seedlings planted. Species typical of alluvial floodplains had better survival than those common to cypress domes or bayheads. Forest litter or straw mulch applied on top of the clays at time of planting did not significantly ($P > 0.05$) improve seedling growth or survival. Forest litter collected and spread during February introduced additional tree seeds which germinated and thrived. Litter from other months was not as successful.

INTRODUCTION

Understanding and managing succession may be a practical way to restore lands to useful functions after mining. Apparently lack of adequate seeding is retarding succession in many phosphate mined areas (Wolfe 1987; Rushton 1988). Experiments were designed to test the feasibility of accelerating succession by planting eleven hydric hardwood tree species in six clay settling areas. Average water table depth and pH, both of which may affect survival, were measured for one growing season. Two mulching techniques were compared to a control.

Clay settling impoundments are a product of phosphate mining, a major industry in central Florida. Typically one ton of clay waste (dry weight) is produced for each ton of phosphate rock. The volume of clay requires above-ground storage areas ranging from 80 to 400 ha surrounded by earthen dams from 6 to 18 m in height (Haynes 1984). Approximately 60 to 75% of the land proposed for mining in central Florida is designated for clay settling areas. Over 30,000 ha of settling impoundments have already been built with 1,000 ha of new ponds constructed each year (Pittman & Sweeney 1983).

STUDY SITES

Six clay settling ponds were selected to plant eleven species of tree seedlings (Figure 1). Gardinier (area A) located at their Ft.

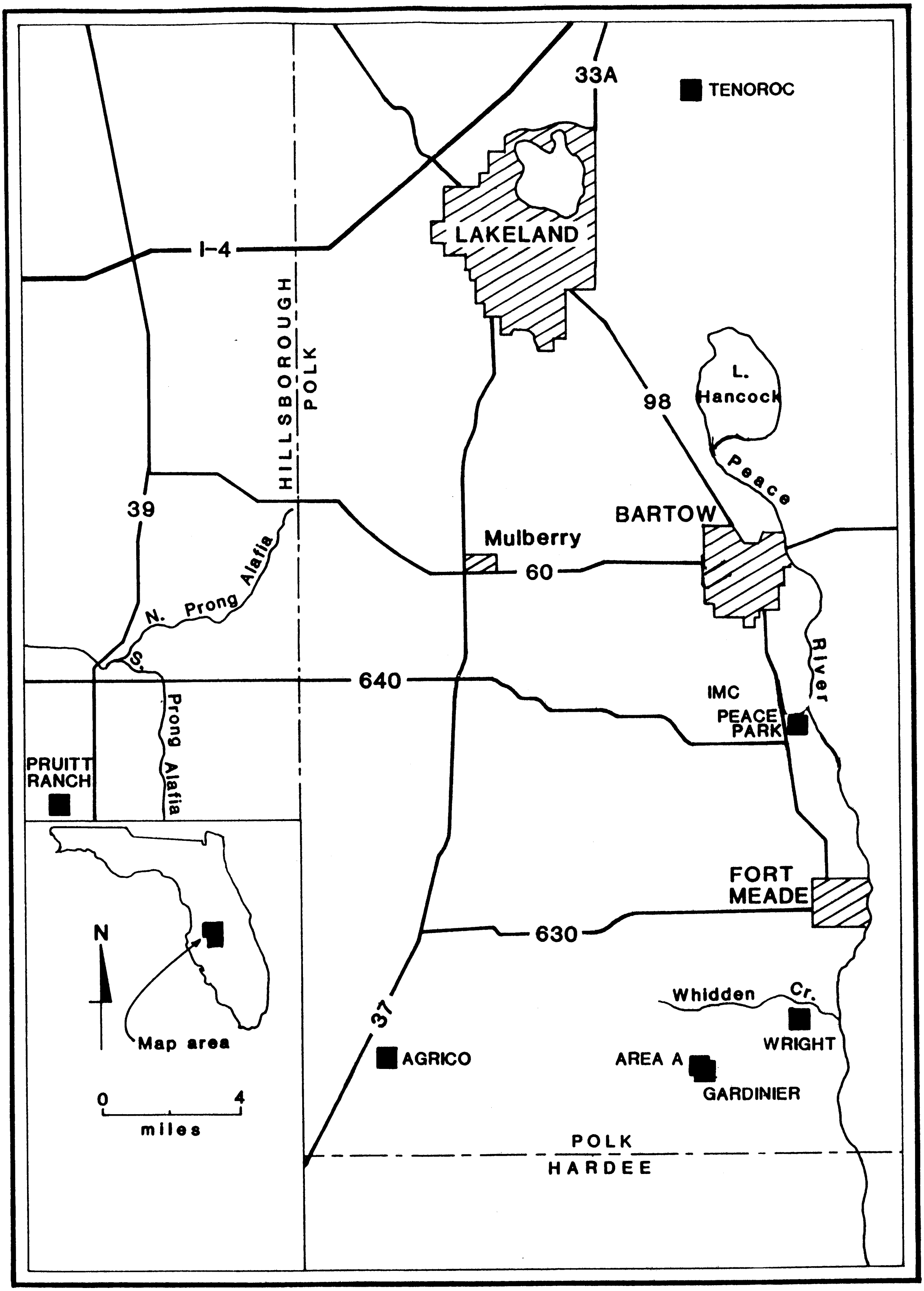

Figure 1. Location of study sites.

Meade mine, was abandoned as a spoils area of 1973. The pond built on unmined land is rectangular in area with clays approximately 10 m thick with a surface area of 130 ha (Blue & Mislevy 1982). In 1975, a perimeter ditch drained the higher ground and the dikes were lowered as a reclamation method. At the time of this study outfall pipes were above the level of the clays providing drainage only during periods of above average rainfall. Two plots were in the flooded, lower end of the pond and four plots were in drier areas. One year after planting, phosphate mining was begun adjacent to the site and the clay pond was drained for use as a disposal area. During these alterations, four of the plots were buried under spoils. Data for the remaining two plots in the wetter, lower end are included in this report.

The O. H. Wright site, also owned by Gardinier, Inc., is located adjacent to the Whidden Creek floodplain. Aerial photographs from 1957 show the old mine cuts being filled with clays. The four study plots were located at the wetter end adjacent to the floodplain. Mining activity nearby interrupted the normal hydroperiod causing both wetter and drier conditions than normal. Tenoroc (area 4A) is a large clay area located in a State Reserve under the jurisdiction of the Department of Natural Resources. Aerial photographs show the site being mined in 1958 and clays being pumped into mine cuts in 1968. Pruitt Ranch is a conventional clay settling pond deactivated 36 years ago. The clays formed a layer about 50 cm deep in the study plots. Reclamation activity by a consultant involved draining and burning the site before the seedlings were planted. Further reclamation activity included disking under three of the plots about a year after the seedlings were established. The one remaining plot, which has been grazed by cattle, is included in this report.

The IMC - Peace River Park site is a conventional clay settling pond estimated from aerial photography to have been mined from 1952 to 1956 and used as a settling area until 1968. It was leased as pasture until 1986 when it was acquired by Polk County for a park and the seedlings were planted. A fire in 1987 killed 70% of the trees. One wet plot with good survival is included in the data set.

The Agrico site is a 193 ha clay settling area located at their Ft. Greene mine. Pumping of clays was terminated in the late 1970s but the pond was used for tailings and debris until 1983. Depth of clays varies from 1 to 10 m. A successful agriculture research project was begun at the site in 1984. By 1986 the surface soils had been dried with ditches 5 to 15 m apart and alfalfa had been planted to accelerate drying further. Three of the tree seedling plots were planted on shallow clays (1 m) where alfalfa was already established and three plots were on deeper clays (> 10 m) near a drainage spillway.

METHODS

Six clay settling ponds were planted with eleven species of hydric swamp seedlings during late February 1986. Thirty-two 9 x 12 m plots were established. Plots contained eighteen individuals of 6 species for a total of 109 seedlings. Plots were cleared of all herbaceous vegetation at time of planting, but the tree canopy, where it existed, was not disturbed. Seedlings were purchased from a nursery where they had been grown in plastic flats and maintained for 6 months to a year in a shade house. They were planted in the field on 1 m centers using a KBC planting bar.

Plots were divided into three mulch treatments. Leaf litter from a nearby floodplain forest representing the same size area (9 x 4 m) was transported to the sites in garbage bags (nine 113 liter bags for each plot). One bale of hay purchased from a feed supply store was spread over a 9 x 4 m area in each plot for the straw mulch treatment, and no ground cover was used for the 9 x 4 m control section.

Hydrology measurements were made using a shallow (2.8 m) water table monitoring well with 0.5 m well screen at the bottom. Wells installed near the center of each plot were measured monthly during the summer of 1986. Average readings from April through October of 1986 were used for comparison between plots.

Soil cores were collected in each plot (except Pruitt Ranch) during a one day period in August 1986. Cores were taken from the portion of the plot with no mulch treatment using a sampler with a mud auger head. The top 5 cm was discarded and soils from the 5 to 15 cm depth were air dried and mixed for analysis. Soil pH was determined in a 1:2 soil:water suspension at the IFAS soil testing lab on the University of Florida campus.

The Duncan Multiple Range test and Chi-square tests were used to determine significant differences using a SAS program at the University of Florida Computing Center.

RESULTS

Measurements of seedling survival is presented in relation to average water table, mulch treatments and pH. Data for 1986 includes all the plots; for 1987 only those plots not impacted by fire, burial, or disking are used.

Seedling Survival

When plots are compared by year and site (Figure 2) there is over 50% survival after the first year which compares favorably with the standard set for bareroot seedlings by the Florida Bureau of Land Reclamation. Additional mortality of 8 to 35% occurred during the following year (except Pruitt Ranch which had been grazed by cattle and was more). The time elapsed since sites were abandoned as active disposal areas did not appear to enhance success but other factors, especially water table and cattle grazing, may have influenced the

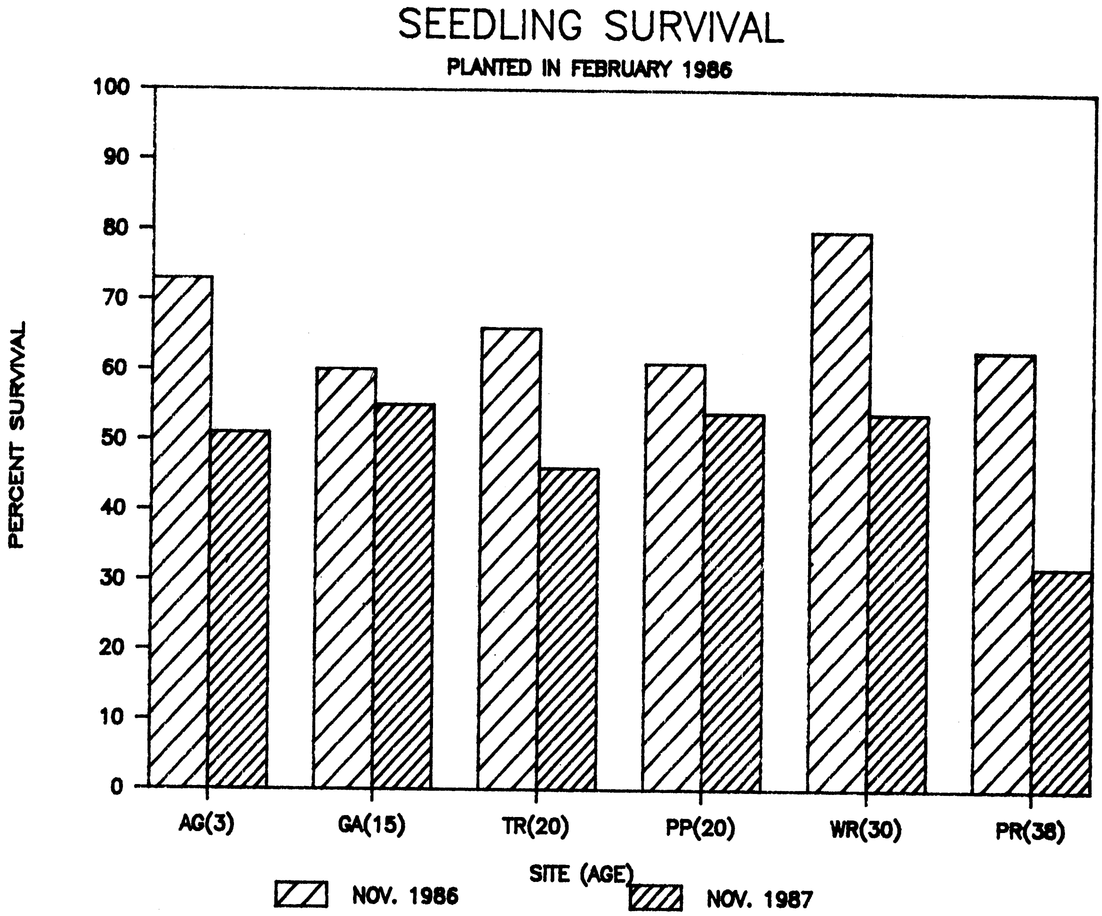

Figure 2. Seedling survival for trees planted in February 1986. Measured in November 1986 after one growing season and again in November 1987 after two years. The numbers in parentheses are the number of years since abandoned as active disposal site. Site abbreviations are in Appendix.

results. More plots are needed to test the theory that early successional vegetation prepares the way for more mature species, thus increasing growth and survival of tree species.

When measured two years after planting considerable differences are seen in survival between species (Table 1). Best survival was found for pop ash (99%) and elm (88%) which were planted at the wettest sites. Red maple planted in drier plots also survived well (70%) but did much better (91%) where the water table was near the surface. Baldcypress (63% survival) also showed its best growth and survival at wet sites (79%). Magnolia had poor survival (41%) overall, but did well (69%) at two plots where the average water table for 1986 was over a meter below the surface. Laurel oak did well at some sites in an intermediate moisture range, but could not tolerate extended periods of standing water. Only one species, loblolly bay, indicated a preference for low pH. It had poor survival (13%) but failed to become established at any site with a pH greater than 6.8 where 31% survived. Swamp bay (28% survival) was killed by standing water greater than two months but did not show a preference for low pH soils typical of bay swamps.

Soil Amendments

Plots were divided into three parts with hay from a feed supply store and floodplain forest litter introduced as two treatments to assess their value as a soil amendment. No significant difference was found in survival or growth between treatments or compared to no treatment ($P > 0.05$) (Table 2). The only advantage to adding forest litter was the introduction of some seeds which increased the diversity of planted plots. Better results would have been seen if more attention had been paid to time of collection and donor site selection. Litter collected at the end of February had the most seedlings germinate (Table 3).

DISCUSSION

Seedling Success

Forested wetlands in the south are organized along a resource gradient from small often stagnant isolated wetlands and headwater streams to strong water flow alluvial rivers and floodplains (Odum 1984). For example, evergreen bay trees grow in bogs with no nutrient or water flow except from rainfall. Isolated wetlands of upland swamps which have small drainage basins are colonized by pond cypress. As more water converges from larger areas a baldcypress strand association is typical. Finally with large inflows of water on a regional scale diverse mixed hardwood swamps predominate. Which system is the most suitable as a reclamation alternative for clay settling ponds in central Florida?

Monk (1966, 1968) divided Florida wetlands into two climax types

Table 1. Survival of tree seedlings planted in clay settling ponds arranged by water table depth.

SITE	WATER TABLE (CM)	TREE SPECIES (% SURVIVAL)											pH
		ACER	GORL	NYSS	QUEL	SABP	TAXD	LIQS	MAGG	FRAC	PERP	ULMA	
AG-1	-183	33	0	0	44	22	6	--	--	--	--	--	7.8
AG-2	-157	23	0	0	20	10	10	--	--	--	--	--	7.9
TR-4	-134	76	50	--	37	0	--	11	72	--	--	--	6.8
AG-3	-110	65	0	6	44	6	11	--	--	--	--	--	7.8
TR-1	-102	67	0	--	44	6	--	39	65	--	--	--	5.8
PR-3	-69	50	16	18	33	22	56	--	--	--	--	--	--
WR-2	-66	72	72	6	78	39	78	--	--	--	--	--	6.4
TR-3	-52	100	0	17	17	6	89	--	--	--	--	--	7.2
WR-3	-49	--	17	6	78	50	39	44	--	--	--	--	6.3
TR-2	-30	94	0	28	17	11	61	--	--	--	--	--	7.4
TR-5	-22	83	17	6	50	0	83	--	--	--	--	--	6.7
AG-4	-10	88	0	--	75	67	--	63	26	--	--	--	7.6
AG-5	-5	89	0	23	39	50	56	--	0	100	61	--	7.7
WR-1	0	--	--	89	0	--	89	--	--	100	0	100	6.3
WR-4	0	--	--	44	6	--	94	--	--	100	0	83	6.9
TR-6	0	--	--	83	44	--	94	--	--	100	88	100	7.6
GA-5	6	--	--	47	6	--	94	--	--	100	0	76	7.5
PP-5	7	--	--	0	22	--	84	--	--	100	39	82	7.1
AG-6	8	--	--	6	35	--	29	--	--	95	38	79	7.6
GA-4	16	--	--	39	0	--	100	--	--	100	0	94	7.3
AVERAGES		70	13	25	34	22	63	39	41	99	28	88	

ACER = Acer rubrum (red maple)
GORL = Gordonia lasianthus (loblolly bay)
NYSS = Nyssa biflora (black gum)
QUEL = Quercus laurifolia (laurel oak)
SABP = Sabal palmetto (cabbage palm)
ULMA = Ulmus americana var. floridana (elm)
TAXD = Taxodium distichum (baldcypress)
LIQS = Liquidambar styraciflua (sweetgum)
MAGG = Magnolia grandiflora (magnolia)
FRAC = Fraxinus caroliniana (pop ash)
PERP = Persea palustris (swamp bay)

Table 2. Percent survival and growth for two mulch treatments and a control show no significant differences between treatments. Values with the same letter are not significantly different.

Tree Species	Control	Litter	Hay	Probabilities
PERCENT SURVIVAL AFTER 2 YEARS				Chi-square
Acer rubrum	63	69	76	0.270
Gordonia lasianthus	13	18	14	0.702
Nyssa biflora	29	23	23	0.486
Quercus laurifolia	30	38	35	0.457
Sabal palmetto	19	18	24	0.584
Taxodium distichum	68	66	62	0.702
PERCENT GROWTH 1986 TO 1987				PR > F
Acer rubrum	104 A	126 A	100 A	0.462
Gordonia lasianthus	57 A	84 A	67 A	0.637
Nyssa biflora	38 A	39 A	73 A	0.361
Quercus laurifolia	43 A	73 A	80 A	0.372
Sabal palmetto	246 A	324 A	350 A	0.771
Taxodium distichum	35 A	24 A	25 A	0.463

according to pH, nutrients, and depth of maximum flooding. In general the bayhead swamp was more sterile, more acid, and not flooded as deeply as the mixed swamp habitat. Dominated by broad-leaved evergreen trees whose acid soils are high in organic matter, bayhead vegetation includes sweetbay, swamp bay, and loblolly bay. Bay swamps are typical of seepage areas and headwater streams and not floodplains of larger rivers (Gross 1987; Clewell et al. 1982). Mixed hardwood swamps are dominated by broad-leaved deciduous species (Monk 1966, 1968). They occur along high energy, nutrient enriched creeks, river, sloughs and basins that are seasonally flooded. Typical tree species are cabbage palm, ash, elm, and baldcypress. Other wetland species are generalists with a wide environmental tolerance and are common in a variety of wetland habitats. These include laurel oak, water oak, red maple, sweetgum, and blackgum.

When one compares the survival rate for the eleven species planted in clay settling ponds, the most successful seedlings were habitat generalists and those found in mixed swamps along alluvial floodplains (Table 4). None of the species common to low nutrient wetlands had greater than 27% survival.

A slightly different conclusion emerges from seedlings planted in

Table 3. Number of seeds germinating from litter transferred from wetland forest to clay settling pond seedling plots.

SITE NAME	DATE INTRODUCED	# OF PLOTS	*DONOR SITE	# SEEDS GERMINATING	SPECIES GERMINATING
Gardinier	7-12-85	2	Whidden Creek Floodplain	0	
	2-1-86	4	Ft. Meade Park at Peace River	0	
Tenoroc	7-23-85	2	Saddle Creek Floodplain	0	
	8-4-85	2	Saddle Creek Floodplain	0	
	9-28-85	2	Saddle Creek Floodplain	0	
Pruitt Ranch	8-3-85	4	Adjacent Creek S. Prong Alafia	0 0	
Peace River Park	1-24-86	5	Peace River at 640 bridge	0	
Agrico	2-8-86	2	Ft. Meade Park at Peace River	1	*Carya aquatica*
	3-11-86	4	Lake Alice Gainesville, FL	55 1	*Acer rubrum* *Celtis laevigata*
O. H. Wright	2-8-86	4	Ft. Meade Park at Peace River	20 10 8	*Ulmus americana* *Celtis laevigata* *Carya aquatica*

*Nine 113.55 liter garbage bags of litter were placed in 4 x 9 meter section in each plot.
**Description of dominant tree species in donor site (see below).

Dominant tree species in donor sites:

Saddle Creek Floodplain: *Taxodium distichum*, *Liquidambar styraciflua*, *Acer rubrum*, *Ilex cassine*, *Cephalanthus occidentalis*, *Quercus laurifolia*, *Fraxinus caroliniana*.
Pruitt Ranch Floodplain: *Quercus laurifolia*, *Sabal palmetto*, *Magnolia virginiana*, and *Pinus elliotii*.
South Prong of Alafia River: *Quercus laurifolia*, *Gledistia aquatica*, *Fraxinus caroliniana*, *Ulmus americana* var. *floridana*, and *Cephalanthus occidentalis*.
Peace River Floodplain: *Taxodium distichum*, *Acer rubrum*, *Ulmus americana* var. *floridana*, *Quercus* spp., *Gledistia aquatica*, *Carya aquatica*, and *Fraxinus caroliniana*.
Whidden Creek: *Magnolia virginiana*, *Acer rubrum*, and *Ulmus americana* var. *floridana*.

Table 4. Percent survival two years after planting compared to wetland community type.

Scientific Name	Common Name	Community Type	Edaphic* Factors	Percent Survival
FRAXINUS CAROLINIANA	Pop Ash	Mixed Swamp	HIGH	99
ULMUS FLORIDAN	Florida Elm	Mixed Swamp	HIGH	88
ACER RUBRUM	Red Maple	Generalist	ALL	71
TAXODIUM DISTICHUM	Bald Cypress	Mixed Swamp	HIGH	65
MAGNOLIA GRANDIFLORA	Magnolia	Mixed Swamp	HIGH	41
LIQUIDAMBAR STYRACIFLUA	Sweet Gum	Generalist	ALL	39
QUERCUS LAURIFOLIA	Laurel Oak	Generalist	ALL	35
PERSEA PALUSTRIS	Swamp Bay	Bay Swamp	LOW	27
NYSSA BIFLORA	Black Gum	Cypress Dome	LOW	25
SABAL PALMETTO	Cabbage Palm	Mixed Swamp	HIGH	23
GORDONIA LASIANTHUS	Loblolly	BayBay Swamp	LOW	14

*Sensu Monk 1966, 1968
Edaphic factors: pH, maximum flooding, Ca, Mg, K, and P.

several clay settling ponds by the Florida Division of State Forestry (Harrell 1988). Many pioneer species had poor success when counted after the first year. Only 45% of laurel oak survived, 46% of red maple and 17% of cottonwood. However, the sweetgum did well with 66% survival and two pine species became established, slash pine had 58% survival and loblolly pine had 63%. A cypress-gum association did well with 60% survival for baldcypress and 64% for blackgum. Loblolly bay was the only evergreen bay planted and 54% survived.

Conflicting results from the two studies indicate it is probably premature to predict the most appropriate climax community type for the new situation of clay settling ponds. However, the results of this study indicate the diverse association found along alluvial floodplains with clay soils are a more promising alternative than those from cypress domes or bayheads.

Mulch Treatments

Loamy soils are considered ideal for plant growth. Clay settling pond substrate are heavy clays (about 80% clay sized particles and 20% silt). Sharkey clay soils in Mississippi with characteristics similar to clay pond substrate demonstrate clays are not as productive as medium texture soils for silviculture (Krinard & Kennedy 1983; Johnson & Krinard 1985). Organic matter can modify the effects of clay by promoting granular-type aggregates in surface soils which increases soil porosity (Brady 1974; Ferry & Olsen 1975). In an effort to ameliorate the negative effect of clay soils, forest litter and straw mulch were introduced as treatments to clay settling pond soils. It did not significantly improve seedling growth or survival when added on top of the clay. Greater benefit might have been seen if the mulch had been incorporated into the clay surface. However, other experiments on clay soils also showed little effect with mulch treatments. For example, in one three year experiment (Unger & Jones 1981) sorghum planted on clay loam soils responded to mulch treatments only on plots with moisture stress. In another experiment (Laverdiere & De Kimpe 1984) adding organic amendments at the rate of 10 t/C · ha as peat or manure on heavy clay soils did not improve the yield of oats in growth chamber experiments.

Experiments with topsoiling have shown beneficial effects from added seeds (Farmer et al. 1982; Howard & Samuel 1979; Tracey & Glossop 1980). Fresh topsoil from 5 to 20 cm thick provide quick effective cover and the most successful species were those that spread by rhizomes. In this study mulching with forest litter during the winter introduced tree seedlings and increased diversity (see Table 3). If a layer of topsoil had also been used an even greater benefit may have been realized.

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LITERATURE CITED

Blue, W. G., and P. Mislevy. 1982. Accelerated drying of the surface of colloidal phosphate settling ponds from phosphate strip mining in Florida. Soil and Crop Science Society of Florida Proceedings 41:205-208.

Brady, N. C. 1974. The nature and properties of soils, 8th ed. MacMillan Publishing Co., New York.

Clewell, A. F., J. A. Goolsby, and A. G. Shuey. 1982. Riverine forests of the south Prong Alafia River System, Florida. Wetlands 1:21-72.

Farmer, R. E., Jr., M. Cunningham, and M. A. Barnhill. 1982. First-year development of plant communities originating from forest topsoils placed on southern appalachian minesoils. J. of Applied Ecology 19:283-294.

Ferry, D. M., and R. A. Olsen. 1975. Orientation of clay particles as it related to crusting of soil. Soil Science 120(5):367-375.

Gross, F. E. H. 1987. Characteristics of small stream floodplain ecosystems in north and central Florida. M.S. Thesis. Center for Wetlands, University of Florida, Gainesville.

Harrell, J. B. 1988. The development of techniques for the use of trees in the reclamation of phosphate lands. Florida Division of Forestry for Florida Institute of Phosphate Research, Bartow, Florida.

Haynes, R. J. 1984. Summary of wetlands reestablishment on surface-mined lands in Florida. Symposium on Surface Mining, Hydrology, Sedimentology, and Reclamation, University of Kentucky, Lexington, 40506-0046.

Howard, G. S., and M. J. Samuel. 1979. The value of fresh-stripped topsoil as a source of useful plants for surface mine revegetation. Journal of Range Management 32:76-77.

Johnson, R. L., and R. M. Krinard. 1985. Oak seeding on an adverse field site. USDA. Forest Service Research Note SO-319.

Krinard, R. M., and H. E. Kennedy, Jr. 1983. Ten-year growth of five planted hardwood species with mechanical weed control on sharkey clay soil. USDA Forest Service, SO-303.

Laverdiere, M. R., and C. R. De Kimpe. 1984. Agronomic use of clay soils from Abitibi, Quebec: 2. Effects of organic amendments and cultivation on crop production. Soil Science 137(2):128-133.

Monk, D. C. 1968. Successional and environmental relationships of the forest vegetation of northcentral Florida. American Midland Naturalist 79:441-57.

Monk, CC. D. 1966. An ecological study of hardwood swamps in north-central Florida. Ecology 47:649-54.

Odum, H. T. 1984. Summary: Cypress swamps and their regional role. In K. C. Ewel and H. T. Odum (Eds.), Cypress swamps. University of Florida Press, Gainesville, Florida.

Pittman, W. E., Jr., and J. W. Sweeney. 1983. State-of-the-art of phosphatic clay dewatering and disposal techniques. Vol. 1. A review of phosphatic clay dewatering research. U.S. Bureau of Mines for Florida Institute of Phosphate Research, NTIS #PB84 185016.

Rushton, B. T. 1988. Wetland reclamation by accelerating succession. Ph.D. Dissertation. Center for Wetlands, University of Florida, Gainesville.

Trace, W. H., and B. L. Glossop. 1980. Assessment of topsoil handling techniques for rehabilitation of sites mined for bauxite with the Jarrah Forest of western Australia.

Unger, P. W., and O. R. Jones. 1981. Effect of soil water content and a growing season straw mulch on grain sorghum. Soil Sci. Soc. Am. J. 45:129-134.

Wolfe, R. W. 1987. Mechanisms of seed dispersal and revegetation of mined lands. In H. T. Odum (Ed.), Accelerating natural processes for wetland restoration after mining. Florida Institute of Phosphate Research #83-03-041R, Bartow, Florida.

APPENDIX

LIST OF TREE SPECIES USED IN TEXT

Acer rubrum	red maple
Carya aquatica	water hickory, swamp hickory
Fraxinus caroliniana	pop ash, carolina ash
Gordonia lasianthus	loblolly bay
Liquidambar styraciflua	sweetgum
Magnolia virginiana	sweet bay
Magnolia grandiflora	southern magnolia, magnolia
Nyssa sylvatica var. *biflora*	blackgum, swamp tupelo
Persea palustris	swamp bay
Pinus elliotii	slash pine
Pinus taeda	loblolly pine
Populus deltoides	cottonwood
Quercus laurifolia	laurel oak
Quercus nigra	water oak, pin oak, red oak
Sabal palmetto	cabbage palm
Taxodium ascendens	pondcypress, pond cypress
Taxodium distichum	baldcypress
Ulmus americana var. *floridana*	American elm

SITE ABBREVIATIONS USED IN TEXT
(see site descriptions for more detail)

AG	Agrico
TR	Tenoroc
PR	Pruitt Ranch
PP	Peace River Park
WR	O. H. Wright
GA	Gardinier

WETLAND CREATION AND RESTORATION EFFORTS ASSOCIATED WITH MITIGATION BANKS

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ABSTRACT

The concept of mitigation banking was developed in the early 1980s as an innovative mechanism to obtain full compensation for unavoidable habitat losses primarily associated with the Section 10/404 permit programs for wetland development projects. Mitigation banks can involve wetland creation, restoration, enhancement, preservation, or some combination of these activities. Defining characteristics of a mitigation bank include: (1) the mitigation activities occur prior to the project impacts for which they serve as mitigation, and (2) a crediting and debiting procedure is used to define the credits available from the mitigation bank and the project-related losses that will be charged against the bank credits. Mitigation banks, once established, can be used to compensate for unavoidable and necessary losses from specific future wetland development actions and appear especially applicable to projects where individual losses are relatively small (but collectively significant) and cannot be fully mitigated on, or immediately adjacent to, the project site. Although mitigation banks can provide a timely and significant opportunity for wetland creation and restoration projects within the permit process, they are a relatively new mitigation tool. Most banks have not yet been implemented long enough to determine their final success as wetland creation or restoration efforts. In other cases, lack of provisions for long-term monitoring and evaluation have hampered assessment efforts. Of the eight mitigation banks discussed, wetland restoration apparently has been successful with one bank, there appears to have been some degree of success with restoration and creation efforts with three banks, it is unclear if any monitoring or evaluation has occurred with one bank, and it is too early to determine success for the other three banks.

INTRODUCTION

Development projects in wetland settings are regulated by a variety of complex and often overlapping State and Federal laws. Wetland development projects may be permitted when they conform to applicable laws and policies, are judged to be in the public interest, and are accompanied by an acceptable plan to mitigate unavoidable, necessary, project-related habitat losses. Compensation for impacts, which may involve habitat replacement, is part of a mitigation planning process that emphasizes impact avoidance or minimization whenever possible (U.S. Fish and Wildlife Service 1983).

Mitigation of unavoidable, necessary resource impacts may take the form of onsite enhancements or offsite activities designed to compensate for losses at the project site. Although onsite mitigation opportunities have higher priority than offsite actions, onsite mitigation opportunities may be unavailable or insufficient to meet project mitigation needs. Restoration of historic wetlands and creation of new wetlands to replace losses of existing wetlands have received increasing attention among regulatory agencies, developers, and environmental groups when offsite mitigation is being considered. Wetland creation or restoration activities have become a common feature in the permit process, often in the form of mitigation banking proposals.

Mitigation banking was developed as an innovative mechanism to obtain compensation for habitat losses primarily associated with Section 10 (Rivers and Harbors Act) and Section 404 (Clean Water Act) wetland resources development projects. Section 404 provides Federal agencies with permit authority over the discharge of dredged or fill material into waters of the United States, while Section 10 requires a Federal permit for dredging, filling, or other activities that impact the navigable capacity of any water of the United States. Mitigation banks are not intended as a short-cut to the legal requirements of either of these Acts. Defining characteristics of a mitigation bank include: (1) the mitigation activities occur prior to the project impacts for which they serve as mitigation, and (2) a crediting and debiting procedure is used to define the credits available from the mitigation bank as a result of bank establishment and management activities and to determine the project-related losses that will be charged against the bank credits.

Entities likely to be involved in the mitigation bank planning process include development interest groups and other permit applicants, Federal and State permitting agencies, county or city planning commissions or other local permitting agencies, Federal and State commenting agencies, and environmental groups. The first prerequisite is that one or more of the involved entities, usually the permit applicant, is willing to develop a bank site prior to its use as mitigation for project impacts. Formal, written banking agreements establish consensus among the planners about the characteristics and use of the bank.

A mitigation bank, once established, can be used to compensate for losses from one or more future development actions. Even though a variety of agencies may have been involved in the establishment of the bank, there usually is no commitment to approve the use of bank credits for the offsite mitigation of specific future projects. Each project proposed for debiting against bank credits during the Section 10/404 permitting process is considered on its own merits. Use of mitigation bank credits is considered only when project losses are necessary and unavoidable, onsite mitigation possibilities are unavailable or insufficient to compensate fully for project impacts, other offsite mitigation options are either infeasible or inappropriate, and the mitigation bank has the appropriate type and number of habitat credits available to offset project losses. Conceptually, mitigation banks are intended to provide a more predictable and less complex process for complying with mitigation requirements, thus improving the effectiveness of mitigation projects. Banks appear to be especially applicable to projects where individual losses are relatively small (but collectively significant) and cannot be fully mitigated on, or adjacent to, the project site.

STUDY BACKGROUND AND RESULTS

The U.F. Fish and Wildlife Service (Service) developed interim guidance on mitigation banking in the early 1980s in response to a number of requests that the Service consider banking fish and wildlife management credits for future use in mitigation. The Service currently is involved with 13 implemented mitigation banks, 12 of which are related to the Section 10/404 program. The two most prevalent types of projects for which banks are being used are highway and port development, with five banks each. The remaining three banks involve oil and gas exploration, industrial development, and a Bureau of Reclamation Federal water development project. In addition to the 13 established mitigation banks, there are at least 10 potential banks in some stage of negotiation.

Mitigation banks can involve wetland creation, restoration, enhancement, preservation, or some combination of these activities. Eight of the 13 implemented mitigation banks with Service involvement have included wetland creation activities or the restoration of historic wetlands that were either filled or drained. These eight banks are described briefly below (Short 1988).

Astoria Airport Mitigation Bank

The Astoria Airport Mitigation Bank consists of 13.4 ha (33 acres) in Clatsop County, Oregon. The land was diked brackish marsh prior to bank establishment, which involved exposing the site to tidal inundation by breaking dikes; creating islands, ponds, and new tidal channels; and building a new dike to prevent flooding of the Astoria Airport. Bank credits are available to mitigate port and harbor development projects (and possibly other types of projects) between the tip of Tongue Point to the west bank of the Skipanon River along the Oregon side of the Columbia River Estuary.

The formal bank agreement for the Astoria Airport Mitigation Bank was signed in October 1986, and it is too early to determine how effective wetland restoration activities will be at the bank site. The bank is considered a pilot project, and provisions have been made to adjust the number of available bank credits if the predicted number of credits is not substantiated by later monitoring and evaluation. There will be an interagency review of the site, which will include a complete evaluation, after 5 years.

Bracut Wetlands Mitigation Marsh

The Bracut Wetland Mitigation Marsh consists of 5.3 ha (13 acres) in Humboldt Bay, Humboldt County, California. The bay is about 8 km (5 miles) northeast of the city of Eureka; the bank site is about 1.6 km (1 mile) south of Jacoby Creek and is contiguous with the Humboldt Bay National Wildlife Refuge. The bank site was a former tidal marsh on the Humboldt Bay shoreline that had been filled with gravel, soil, and wood debris and used as a lumberyard. The wetland was restored by breaking the dike after excavating and recontouring the area. The site restoration plan also called for placing rip-rap along the outer levee and planting marsh vegetation. The bank was established to offset wetland losses from industrial development within the city of Eureka, in particular the loss of pocket marshes (marshes no larger than 0.8 ha [2 acres] in size).

The Bracut Wetland Mitigation Marsh was implemented in 1981. Because of the unique soil conditions in an area formerly used as a lumberyard, it was not possible to predict success of restoration activities in terms of production and habitat quality. In May 1987, a field study of the bank was conducted to evaluate the physical and biological characteristics of the site. A number of problems were noted, including a substantial amount of wood debris floating to the surface and drifting out into Arcata Bay; slow and poor establishment and an unexpected distribution of marsh plants; a hard, largely barren, gravel surface in some parts of the bank; and the presence of hydrogen sulfide gas in some of the tidal channels. The study also found that a number of important habitats had been established within the bank, including tidal pools and freshwater/brackish water wetlands, and that benthic invertebrate abundance was relatively high throughout the site. Habitat improvement recommendations, based on the study results, included removing the sill at the dike break, excavating the tidal channels to improve tidal drainage, and reducing the elevation of islands to alleviate acidic soil problems. Although few of the problems have been quantified, there is concern about how successful the mitigation bank has been in providing replacement habitat for the wetland sites that have been lost. The bank has certainly provided a learning experience in terms of restoration of wetland sites.

Port of Los Angeles/PacTex-Batiquitos Lagoon Mitigation Bank

The Port of Los Angeles/Pactex-Batiquitos Lagoon Mitigation Bank consists of 241 ha (596 acres) in Batiquitos Lagoon, San Diego County, California. The Batiquitos Lagoon is an elongated coastal basin that extends approximately 4 km (2.5 miles) inland and is 0.8 km (0.5 miles) in width. Tidal volume in the lagoon has been substantially reduced due to sedimentation, particularly within the last quarter century.

Seasonal freshwater inflow and elimination of tidal influence have resulted in fresh or brackish water inundation after winter rains, with subsequent evaporation during the dry season that results in high salinity levels and large salt flats. Large areas of the lagoon dry up completely in dry years. Bank establishment will involve restoring tidal influence in the lagoon by reestablishing and maintaining an open ocean entrance and creating additional marshland. Bank credits will be available to mitigate fill projects in the Port of Los Angeles.

The formal agreement for the Port of Los Angeles/Pactex-Batiquitos Lagoon Mitigation Bank was signed in November 1987. Mitigation activities have not yet occurred, and it is too early to determine how effective restoration efforts will be. Physical and biological monitoring will be done after lagoon restoration is completed in 1990.

Port of Long Beach Pier A-Newport Bay Mitigation Bank

The Port of Long Beach Pier A-Newport Bay Mitigation Bank consists of 11.7 ha (29 acres) in the Upper Newport Bay Ecological Reserve (UNBER), city of Newport Beach, Orange County, California. Bank lands were a predominantly barren, superlittoral area of degraded saltmarsh. Bank establishment involved restoration, including some dredging, of tidal influence to the area. Bank credits are available to offset losses in marine habitat from port development landfill projects in the harbor district.

The formal banking agreement for the Port of Long Beach Pier A-Newport Bay Mitigation Bank was signed in March 1984. Restoration activities at this bank appear to be successful. The site consists of subtidal and intertidal mudflats and saltmarsh of high value to coastal marine fishes and migratory birds. The restored area is a shallow, protected embayment that provides shoreline habitat and critical nursery habitat for marine fish and shellfish of commercial and recreational importance.

Port of Long Beach Pier J-Anaheim Bay Mitigation Bank

Long Beach Pier J-Anaheim Bay Mitigation Bank consists of approximately 44.5 ha (110 acres) in three separate parcels in the Seal Beach National Wildlife Refuge, located within the Seal Beach Naval Weapons Station, Orange County, California. Bank land is "weedy uplands" of low habitat value. Bank construction will involve the creation of tidally-influenced wetland and water habitat and should begin in late 1988. Specific actions will include developing some slope and islands in the area, constructing light-footed clapper rail (*Rallus longirostris levipes*) nesting mounds on the islands, and placing culverts under existing roadbeds to provide permanent unimpeded flushing of each parcel by tidal waters. Bank credits will be available to offset project losses in the Port of Long Beach and also may be transferred to other port districts in the Southern California Bight as appropriate.

The formal banking agreement for the Port of Long Beach Pier J-Anaheim Bay Mitigation Bank was signed in February 1986. Wetland creation activities at the mitigation site have not yet been completed, and it is too early to determine how effective these efforts will be. Selection of the bank site was based, in part, on the existence of tidal sloughs and saltmarsh with adjacent upland or diked areas that could be returned to tidal influence through excavation and improved tidal conduits.

Minnesota Department of Transportation Wetland Bank

The size of the statewide Minnesota Department of Transportation Wetland Bank is variable; additional credit lands can be added to the bank at any time. The habitat measures used to establish bank credits are, in order of priority, wetland restoration (particularly prairie potholes), enhancement of existing wetlands, and creation of wetlands out of upland borrow pit sites. Bank credits are available for highway projects, and debit and credit accounting occurs separately for each Minnesota Department of Transportation District.

A technical memorandum concerning the Minnesota Department of Transportation Wetland Bank was issued in January 1987. Mitigation plans are developed separately for each credit area accepted into the bank. In most cases, bank credit areas are freshwater wetlands of value to migratory waterfowl and other wetland species. It is unclear how, or if, success of wetland creation or restoration efforts at individual credit sites is being monitored or evaluated.

Goose Creek Mitigation Bank

The Goose Creek Mitigation Bank consists of 4.5 ha (11 acres) on Goose Creek, a tributary of the west branch of the Elizabeth River in Chesapeake, Virginia. The bank land was a former tidal wetland. Bank establishment involved restoring the area as a tidal coastal saltmarsh by excavating and grading an existing borrow pit, excavating a tidal flow channel between the borrow pit and Goose Creek, and planting indigenous species of wetland vegetation. Bank credits are available to offset small highway projects in the Suffolk District that affect saltmarsh wetlands.

The Goose Creek Mitigation Bank was implemented during 1982-1984 as restored tidal coastal saltmarsh. Although the Service wanted to do follow-up monitoring and assessments of the success of the habitat improvement measures, it has not had the resources to do so. In 1982, the Virginia Institute of Marine Science expressed an interest in long-term monitoring of the bank site as a developing marsh system and wildlife habitat. However, the Institute did not get the anticipated funding for their effort and monitoring plans were greatly reduced. The bank appears to be functional and manageable at 4.5 ha (11 acres), although bank design has a manmade, rather than a natural, appearance. Vegetative establishment at the bank site has been incomplete, and it is unclear if the problem will remedy itself over time or if there will be a need to replant the unvegetated areas or take some other remedial action.

North Dakota State Highway Department Mitigation Bank

There is no fixed size to the North Dakota State Highway Department Mitigation Bank; bank credit lands can be added as opportunities arise. Credit areas theoretically can be anywhere in the State, but, for all practical purposes, they are limited to areas north and east of the Missouri River. Bank establishment measures include creating wetlands, impounding wetlands, restoring drained wetlands, and developing subimpoundments. Use of bank credits is restricted to replacement of easement wetlands impacted by highway construction.

The formal agreement for the North Dakota State Highway Department Mitigation Bank was signed in August 1975. Restoring drained wetlands is the most desirable mitigation action associated with the bank. Replacement wetland habitats accepted as bank credit include constructed wetlands (both excavated and impounded) and restored drained wetlands. The agreement does not require monitoring or evaluation of the bank credit areas. However, research on constructed wetlands in North Dakota is supportive of their value as functional replacement habitats, based on the level of waterfowl use of these areas.

DISCUSSION

In recent years, there has been a proliferation of wetland development proposals that include wetland creation or restoration as mitigation goals (Golet 1986). The adoption of these concepts into the Section 10/404 permit process has significant ramifications for the Nation's wetlands. Although techniques have improved greatly in the past decade, there still is considerable debate over the advisability of using wetland creation and restoration as mitigation tools. The record of previous wetland restoration and creation efforts indicates that the technology is still experimental and unpredictable and there are no guarantees that created or restored wetlands will serve as permanent substitutes for lost wetlands (Race 1985). It is difficult, at best, to determine if replacement wetlands function as well, or provide the same values, as the wetlands they are intended to replace (Golet 1986). In addition, it may take years before replacement wetlands even begin to approach the structural and functional complexity of the lost wetlands.

Additional concerns in terms of the Section 10/404 permit process are that compliance with permit mitigation conditions in general appears to be low (Quammen 1986) and the frequent inability to monitor project-related mitigation after a permit has been issued. Unfortunately, issuance of a permit does not guarantee suitable replacement wetland habitat, even when such mitigation has been made a condition of the permit. When wetland creation and restoration techniques are combined with the permit process as mitigation for wetland losses, the risk is the loss of even more wetland area if the establishment project is unsuccessful.

Initial approaches to the use of wetland creation and restoration as mitigation have been largely intuitive (Race 1986). Although mitigation banks can provide a timely and significant opportunity for wetland creation and restoration projects within the permit process, they are a relatively new mitigation tool. Most banks have not yet been implemented long enough to determine their final success as wetland creation or restoration efforts. In other cases, lack of provisions for long-term monitoring and evaluation have hampered assessment efforts. Of the eight mitigation banks discussed above, wetland restoration apparently has been successful with one bank, there appears to have been some degree of success with restoration and creation efforts with three banks, it is unclear if any monitoring or evaluation has occurred with one bank, and it is too early to determine success for the other three banks.

All of the above factors need to be taken into consideration when a mitigation bank involving wetland creation or restoration is considered as a mitigation option. Balancing these concerns are a number of potential advantages in associating wetland creation and restoration activities with mitigation banks. Banks, where habitat mitigation credits are established in advance of project impacts, eliminate the time lag between losses of fish and wildlife habitat at the development site and compensation for those losses. Successful wetland creation or restoration efforts typically take 5 years or more to become fully functional (Riddle 1986). In a banking situation, bank credits against which project impacts can be debited are not, at least theoretically, established until the wetland values are present. This is a far different situation from the mitigation requirements generally imposed by permitting agencies, which, at best, call for concurrent initiation of mitigation and development projects, resulting in considerable delay before replacement habitat is functional.

There also may be economies of scale and more management options when wetland creation and restoration projects involve large blocks of habitat, as mitigation banks often do. In some situations, a mitigation bank may provide a better option for habitat compensation than to require small, individual mitigation projects. For example, the establishment of small, scattered wetlands may be counterproductive in areas where local or regional goals involve creation or restoration of large contiguous wetlands that contain a diversity of fish and wildlife habitats.

Three questions need to be addressed when the establishment of a mitigation bank involving wetland creation or restoration is being considered as a mitigation option (Larson 1987):

1. How well can we measure the functions of the wetland that will be lost?

2. Do we know enough about the existing wetland functions to create a functional analog at another site?

3. Where specific information is lacking, has there been enough practical field experience to provide reasonable guidance for wetland creation or restoration pending further research?

In addition, both short- and long-term assessment studies of mitigation banks are needed to determine if the wetland creation and restoration efforts are implemented as planned and to evaluate how well the new wetlands are functioning. Development of successful evaluation criteria depends both on a valid sampling design and on written documentation that clearly states the wetland creation or restoration objectives and provides sufficient detail about project design. The banking agreement should include a description of the types of wetland habitats and functions that are to be created or restored and the area of each type. Technical specifications related to where, when, and how the wetland creation or restoration activities will occur need to be clearly stated.

Implemented banks involving wetland creation and restoration efforts have met with mixed success. However, with proper long-term monitoring and evaluation, mitigation banks represent an excellent opportunity to learn more about the predicted and actual gains in habitat value with various types of creation and restoration efforts. Project monitoring and evaluation cannot be over-stressed as important components of banking efforts; both to provide information about the extent of the actual mitigation to offset losses and to develop a scientific data base that can be used in planning future wetland creation and restoration efforts. The existing knowledge base is far from complete about the length of time necessary to establish a functioning wetland, the number and extent of areas needed to provide an equivalent of the wetlands that will be lost, and whether or not all of the functions of an existing wetland can be duplicated in a mitigation bank wetland (Riddle 1986). The political reality of the situation is that our ability to mitigate for development project losses through wetland creation and restoration will have to be largely learned on the job. As Daylor (1987) aptly put it, economic development pressures will not tolerate a "stop everything until we know everything" attitude about these two mitigation possibilities.

LITERATURE CITED

Daylor, R. F. 1987. Engineering considerations in wetland mitigation. In J. S. Larson and C. Neill (Eds), Mitigating freshwater alterations in the glaciated northeastern United States: An assessment of the science base (pp. 101-114). Proceedings of a workshop held at the University of Massachusetts, Amherst, 19-30 September 1986. The Environmental Institute Publication No. 87-1, 143 pp.

Golet, F. C. 1986. Critical issues in wetland mitigation: A Scientific perspective. National Wetlands Newsletter, Sept.-Oct.:3-6.

Larson, J. S. 1987. Wetland mitigation in the glaciated Northeast: Risks and uncertainties. In J. S. Larson and C. Neill (Eds.), Mitigating freshwater alterations in the glaciated northeastern United States: An assessment of the science base (pp. 4-6). Proceedings of a workshop held at the University of Massachusetts, Amherst, 29-30 September 1986. The Environmental Institute Publication No. 87-1, 143 pp.

Quammen, M. L. 1986. Measuring the success of wetlands mitigation. National Wetlands Newsletter, Sept.-Oct.:6:8.

Race, M. S. 1985. Critique of present mitigation policies in the United States based on an analysis of past restoration projects in San Francisco Bay. Environmental Management 9(1):71-82.

Race, M. S. 1986. Wetlands restoration and mitigation policies: Reply. Environmental Management 10(5):571-572.

Riddle, E. P. 1986. Mitigation banks: Unmitigated disaster or sound investment. Paper presented at the national symposium: Mitigation of impacts and losses, 9 October 1986, New Orleans, Louisiana, sponsored by the Association of State Wetland Managers.

Short, C. 1988. Mitigation banking. U.S. Fish and Wildlife Service Biological Report 88(4), 103 pp.

U.S. Fish and Wildlife Service. 1983. Interim guidance on mitigation banking. Ecological Services, Instructional Memorandum No. 80. Fish and Wild. Serv., Div. Ecol. Serv., Washington, D.C., n.p.