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PROCEEDINGS OF
THE TWENTY SECOND ANNUAL CONFERENCE ON ECOSYSTEMS RESTORATION AND CREATION

May 1995

Sponsored by

HILLSBOROUGH COMMUNITY COLLEGE
INSTITUTE OF FLORIDA STUDIES

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INTRODUCTION

Formerly the Annual Conference on Wetlands Restoration and Creation, this years Conference expanded its scope to not only include topics addressing wetlands but transitional and upland areas as well. Because of this expanded scope this conference has changed its title to the Annual Conference on Ecosystems Restoration and Creation. The Annual Conference on Ecosystems Restoration and Creation provides a forum for the exchange of results of scientific research in the restoration, creation, and management of: freshwater and coastal systems, uplands and transitional areas. 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 the restoration, creation and management of lands.. These Proceedings are a compilation of papers presented at the Twenty Second Annual Conference and are intended to represent these presentations.

As in years past, 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 twenty years ago. We are grateful for his help and participation. Appreciation is also extended to Charles Duesner for providing administrative support for the conference.

The following people also deserve acknowledgment for contributing to the conference and assisting in the preparation of the proceedings for publication: Elaine Baskin, Candy Bryant, , Janet Giles, Charles Mason and the staff from the HCC Bursars Office, and Sandra Upchurch. Special thanks to Johnnie Hurst for her untiring assistance in handling the many details and to Patrick Cannizzaro for his assistance in coordinating this year's Conference.

Thanks are extended to the **Don Schmitz** (FL Department of Environmental Protection) for presenting the Keynote Address and staff of the **FL Department of Environmental Protection** and **Southwest Florida Water Management District** for arranging and conducting very successful field trip.

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

To all these people, thank you.

RESPONSIBLE DEVELOPMENT INCORPORATING A HOLISTIC APPROACH TO ECOSYSTEM PRESERVATION AND CREATION

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ABSTRACT

In 1991 the Collier family of Naples began the development of a 182 hectare (450 acre) site that had remained undisturbed since the early 1920's. Prior to any construction, a complete inventory of the plant and animal communities was conducted. The site, in its natural state, included scrub and pine uplands, freshwater wetlands and 1.9 kilometers (1.2 miles) of the Cocohatchee River, a tidal stream with mangroves and saltmarsh vegetation. The development and management plan consisted of five major components, habitat enhancement, wildlife conservation, water conservation, waste management and energy efficiency. The design was prepared in accordance with the principles of sustainable resource management outlined by the New York Audubon Society's Cooperative Sanctuary Program.

Clearing for the golf course and housing was selective to avoid wetlands and was kept to minimal levels. Lakes and wetlands were created throughout the property to treat stormwater and provide habitat diversity. Areas on the golf course that could be replanted with native grasses and vegetation were restored with species similar to those found in the surrounding natural areas. All of the housing lots, clubhouse and common areas along the roadways were planted with native vegetation and future residential landscaping must utilize native plants. The irrigation system for the golf course was designed to throw water away from the natural areas. Only 35.6 hectares (88 acres) of the golf course is planted with turfgrass. The project includes 21.5 hectares (53 acres) of preserved uplands and wetlands and an additional 15.8 hectares (39 acres) of created lakes and wetlands. More than 600,000 native plants have been planted around this site, not including the residential properties, none of which were required by permits. Wildlife utilization has been enhanced by preserving or creating a variety of vegetative communities.

INTRODUCTION

The Cooperative Sanctuary Program was developed by the Audubon Society of New York State to promote the principles of sustainable resource management.

The program was designed to encourage golf courses to integrate wildlife conservation, habitat enhancement, efficient use of water and energy, and effective waste management into their construction and operational practices. Meeting those goals requires planning and commitment, from preconstruction surveys through long term maintenance.

STUDY SITE

In 1991 the Collier family began the development of a 182 hectare (450 acre) tract of land in Naples, Florida. The property had remained undisturbed during the seventy years that they had owned it. In accordance with Audubon guidelines, the first step in the development process was to survey the site and prepare a comprehensive inventory of the plant and animal communities. A permanent record of each plant species was established, with photographs and pressed specimens. The site, in its natural state contained scrub oak forests, pine flatwoods, a pond cypress dome, and 1.9 kilometers (1.2 miles) of the Cocohatchee River. The Cocohatchee is a tidal ecosystem with mangroves, saltmarshes, and some freshwater tributaries.

DISCUSSION

The clearing in the development plan was routed around on-site wetlands, which were incorporated into the green belt areas of the project. The initial clearing for the golf course was limited to in-play areas, leaving undisturbed areas along the fairways or between tees and landing areas. In areas where clearing of out-of-play areas was unavoidable, native vegetation was reestablished to blend with the preserved habitat. Several lakes were constructed and planted with native aquatic vegetation, often with a transition from littoral shelves, through native grass slopes and into preserved upland forests. Berms were constructed along some property lines to provide natural screening, and were also heavily planted with native trees, shrubs and grasses. Following construction of cart paths, sidewalks and roadways, native plants were reestablished next to the impervious surfaces, reclaiming construction accessways. Over 600,000 native planted were installed on and around the golf property, none of which were required by any permits.

Shoreline plantings included several species of aquatic plants; pickerelweed (Pontederia cordata), spikerushes (Eleocharis cellulosa and E. interstincta), fire flag (Thalia geniculata), Canna lilies (Canna flaccida), soft rush (Juncus effusus), blue flag (Iris hexagona), white water lilies (Nymphaea odorata), bulrush (Scirpus californicus), swamp lilies (Crinum americanum) and cordgrass (Spartina bakeri).

Cordgrass and coastal dropseed (Sporobolus virginicus) were used extensively on slopes and berms. Uplands areas were planted with muhly grass (Muhlenbergia capillaris), wiregrass (Aristida stricta), saw palmetto (Serenoa repens), scrub oaks (Quercus myrtifolia and Q. geminata), slash pines (Pinus elliotti), Walter virburnum (Virburnum obovatum) and coontie fern (Zamia pumila). Some damaged areas in the Cocohatchee River ecosystem were restored by planting black mangroves (Avicennia germinans), needlerush (Juncus roemerianus), sawgrass

(Cladium jamaicensis) and leather fern (Acrostichum danaeifolium). More than 15.8 hectares (39 acres) of lakes and wetlands were built on this site. The golf course only contains 35.6 hectares (88 acres) of turfgrass, compared to an average of 50.6 - 60.7 hectares (125-150 acres). The goal at Collier's Reserve is to systematically convert some turfgrass to native vegetation each year reducing the turf areas to 30.4 hectares (75 acres).

During the grow-in phase of the golf course, no herbicides were used to control weeds. Instead, unwanted plants were removed by hand until the turf had grown in. The irrigation system was designed to deliver water only to those areas that need it, not into the native plant areas, impervious surfaces, lakes or wetlands. The pumps have variable frequency drives and the whole cycle can be completed in less than six hours, allowing operation during Florida Power & Light's off-peak hours. By utilizing integrated plant management (IPM) techniques, herbicide and pesticide use can be kept to minimal levels and pests are controlled by natural means, where practical.

Wildlife utilization of the golf course and surrounding natural areas is encouraged in a variety of ways. Brush piles were left in wooded areas, dead trees were left as snags for birds and insects, and sixty bird houses have been constructed around the site, targeting ten species. Wildlife cover and vegetated corridors exist throughout the property, surrounding the golf course, houses and common areas.

The maintenance complex incorporates state of the art equipment and procedures for safe and efficient operation. Fuel storage facilities are double vaulted and above ground. Wash water is captured, cleaned and recycled. Chemical areas have in-ground sumps for containment and recycling. The shop, equipment storage and chemical storage areas are kept clean and orderly so leaks and spills can be easily detected and corrected. Recycling and energy conservation are built into the daily operations of the facility.

In order to facilitate continued innovation and success in the sustainable development program a resource committee was formed which meets occasionally to discuss ideas and offer suggestions for improving habitat values and increasing energy and water efficiency. The members include an ornithologist, a biologist, an energy expert, a member of the golf club, a recycling expert, a member of the Water Management District, and the golf course superintendent.

CONCLUSION

The result of these efforts is a golf course community that dispels the common misconception that such developments are devoid of natural plant communities, use excessive amounts of water, pesticides and fertilizers, and are poor environmental stewards. Collier's Reserve has enhanced the natural features of this site and preserved the character and integrity of the land and water. Wildlife utilization has increased over the pre-construction levels identified in the faunal surveys. Wading birds, raptors (bald eagles, ospreys, three species of hawks), scrub jays, red-wing blackbirds, and woodpeckers are common throughout the site.

Plant diversity has been increased through planting efforts, while the on-site plant communities have preserved and in some cases, expanded. The mammalian residents include red and gray foxes, marsh and cottontail rabbits, deer, opossums, raccoons, grey squirrels and fox squirrels. Gopher tortoises, aquatic turtles, alligators, and several snake species make up the reptilian population. The wetlands and lakes contain a diverse array of fish, amphibians, crustaceans, and other invertebrates. Over the next few years, members of the Natural Resource Committee and other biologists will update the post development biological survey.

DISCUSSION

Since it is unlikely that growth and development will cease in the immediate future of Florida, the prudent course of action would seem to be incorporate that growth into the native environment in a responsible manner. Rather than eliminating wildlife habitat, well-planned projects can preserve and even enhance areas that will provide food, shelter, and nesting sites for indigenous animals and can act as corridors between large tracts of publicly-owned wildlife preserves. The Cooperative Sanctuary Program is gaining momentum and support from both the environmental and developmental communities.

BALD MOUNTAIN, A SCRUB AND SANDHILL RESTORATION

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ABSTRACT

This 80.94-hectare (200-acre) restoration project at IMC-Agrico included hydroseeding 19 species of forbs and grasses, direct seeding palmettos and acorns, and containerized planting of scrub and sandhills acres. Sixty-five species were planted in the 26.3-hectare (65-acre) sandhills area, including 67,000 grasses from 8 species planted at 2.0 meters (6 feet, seven inches) apart. Forty-three species were planted in the scrub area at a density of 550 plants per .4 hectare (1 acre). After 9 months and the first dry season, over 95% of the plants were alive and there was much spring growth. The hydroseeding produced a good cover of partridge pea and many other forbs. Many forbs and grasses flowered and produced seed the same fall during which they were planted. Hydroseeded plants also fruited. The authors believe that mixing the overburden, which had a high clay content, with sand tailings was an essential step. The augers with large bits prepared excellent planting holes and roots grew out from the container media very rapidly. The authors thought watering the plants well after planting was important, even during the rainy season, but additional waterings may not have been as necessary. The plantings on the sand tailings areas are surviving well; those areas on high, exposed slopes have the highest losses. Losses are also greater where the soil has a higher clay content and the subsequent weed growth is greater. Each planting event was planned for optimal seasonal advantage: hydroseeding in late November and early December saw palmetto in early summer for high soil temperatures, and container planting in late summer for most consistent rainfall and active growth.

INTRODUCTION

This project was conceived by IMC-Agrico and Brewster Phosphate to restore two sand tailings piles to sand scrub and sandhill systems in an area south of their Kingsford Mine that is west of Bradley Junction, Florida. The Natives was hired to design, collect seeds, grow the plants, and implement the plan.

This project has 30.35 hectares (75 acres) of sand scrub restoration and 26.31 hectares (65 acres) of sandhill restoration on a large hill of sand tailings, and 10.12 hectares (25 acres) of sand scrub restoration on a smaller sand tailings hill. Sand tailings are a product of sorting the matrix by size in phosphate mining. Between the two hills was an acres converted to pasture and lakes in which approximately 16.19 hectares (40 acres) were planted to mixed forest.

MATERIALS AND METHODS

The large hill was covered with a 15.2-centimeter (6-inch) layer of overburden that was laid down by large pans, and the actual depth varied from 7.6 centimeters (3 inches) to over 30.5 centimeters (12 inches). The overburden had a fairly high clay content, thus making an unnatural soil horizon for a scrub or sandhill restoration. To improve soil conditions, the overburden was mixed into the sandtailings. After several attempts to mix the overburden and sandtailings by disking, a road pulverizer-mixer with 45.7-centimeter (18-inch) radii blades that set every few inches apart that thoroughly mixed the soil into a looser consistency was used.

On the large hill, 12.4-hectare (1-acre) plots were left without overburden, whereas all of the small hill remained without overburden. This feature was designed by IMC-Agrico to provide natural openings in the scrub where woody and herbaceous material would thrive, and most were not planted. Four of these occurred on what was to be the sandhill area in the western and northern portion of the big hill; they were planted in the same manner as the rest of the sandhill system. This also allowed the opportunity to contrast success and progression on the two soil types.

The 11.33 hectares (28 acres) of sandhill and 4.05 hectares (10 acres) of the scrub were hydroseeded in January 1993 with native forbs and grasses. This work was scheduled to begin at the end of November. Chances of drizzly rains, lower soil temperatures, and more time for growth before the usual spring droughts were reasons for choosing this time of year. Our previous research, research from other parts of the country and from the National Wildflower Research Center, indicate this is the optimal time to seed herbaceous species almost anywhere in the United States. Unfortunately, there was a "cease and desist" order on the whole project because of questions they had on work in the lakes area. This prohibited the necessary soil mixing preparation from proceeding, and hydroseeding was delayed until January 14th. December and January had an abundance of rains, ideal for seeding, which were missed.

The hydroseeder has the capacity of spraying .2 hectare (1/2 acre) per tankful. Each tank of water was mixed with paper mulch, blue dye, seed lots, and fertilizer that was applied 22.7 kilograms (50 pounds) of 12-8-8 per .4 hectare (1 acres). This is a relatively light application, so as not to encourage weed growth.

An estimated 165.6 kilograms (365 pounds) of seed were used, 81.6 kilograms (180 pounds) of two partridge peas (Cassia fasciculata and Cassia nictitans var. aspera) and 7.17 kilograms (158 pounds) of hand-collected seed from the fall of 1992, and 12.2 kilograms (27 pounds) of hand-collected seed came from the previous year. The 17.7 kilograms (158 pounds) of hand-collected seed came from 17 major species of sandhill and sand scrub forbs and grasses, and a small amount of many more species listed only as sandhill mix, or sand scrub mix. The major species included several species of blazing star (Liatris ssp.), elephant's foot (Elephantopus caroliniana), Palafoxia feayi, lop-sided indiangrass (Sorgastrum secundum), Florida paint brush (Carphephorus corymbosus), and yellow buttons (Balduinia angustifolia).

Table 1. Hydroseeding, Hand-collected Seed from 1992.

SPECIES	TOTAL GRAMS	SANDHILL	SCRUB
Balduinia angustifolia	16,046.1	All	All
Polygonella fimbriata	4,309.2	All	All
Polygonella polygama	680.4		All
Palafoxia feayi	19,533.2	All	All
Certatiola ericoides	56.7		Part
Smilax laurifolia	56.7	Part	
Liatris chapmanii	6,293.7	Part	Part
Carphephorus corymbosus	3,203.6	Part	
Sorgastrum secundum	6,350.4	All	
Liatris laevigata	4,536.0	All	All
Dalea pinnata	113.4	Part	
Sandhill mix	241.0	Part	
Seymaria pectinata	14.2		Part
Yucca filamentosa	1,771.9	All	All
Liatris spp. mixed	751.3	Part	
Pityopsis graminifolia	340.2	All	
Elephantopus caroliniana	1,587.6	All	
Garberia heterophylla	5,386.5		All
Scrub mix	311.9		All
17 + Species	71,583.8 Grams or 71.6 Kilograms		

Palmetto seed, unlike the forbs, germinate with high soil temperatures, 33 degrees C (2 degrees F). In June, 25,000 saw palmetto (Serenoa repens) and 30,000, scrub palmetto (Sabal etonia) were seeded and sealed in the scrub areas using hand-held dibbles. Since palmettos are slow growing and therefore, more costly to grow, an inexpensive way to introduce more material was sought.

The latter part of the growing/rainy season is the best time to plant uplands, because the rains are most consistent and the root systems are actively growing. On August 2nd, planting began in the sandhill portion using variably-shaped templates of 301.9-square-meter (3,250-square-foot) areas that each contained 15 trees, 2 shrubs, 11 forbs, and 72 grasses. All of the plant material was grown by The Natives from seeds and cuttings collected within at least 160.9 kilometers (100 miles) of the site, except the longleaf pine, which was potted on from bareroot seedlings from the Florida Division of Forestry. Hand-held augers were used with bits several inches larger than the pot size for all of the planting. A separate watering crew watered in the plants the following day and kept each area watered on a random schedule for the month following each planted area.

Grasses were planted 2.0 meters (6 feet, 7 inches) apart on average though in a normal, mature sandhill the usual spacing is 61.0 centimeters (2 feet) where clump occur as a close cover (Clewell, 1988). Eight grass species, totalling 66,827 plants, were planted from 15.2-centimeters (6-inch) tubeling containers. Thirty-one percent of the grasses were wiregrass (Artistida stricta), 30 percent were pineland dropseed (Sporobolus junceus), and 13 percent were lop-sided indiangrass (Sorgastrum secundum). The two lovegrasses (Eragrostis elliottii and Eragrostis spectabilis), were used especially because they reseed easily into disturbed areas and are found in these upland systems. The two species usually found in wetter sandy soils, Aristida spiciformis and Andropogon capillipes (A. virginicus var. glaucus), were used on the lower, wetter edges of the hill. Splitbeard bluestem (Andropogon ternarius) is another common sandhill grass.

The planted tree density was only 200 per .4 hectare (1 acre), and even less would have been suitable for an ecosystem that is basically a grassland. The 15,345 1-gallon-sized trees and shrubs were 67% longleaf pine, 17% turkey and bluejack oak, and the remainder a mix of 16 other species. Thirty-seven herbaceous species, totalling 9,300 plants were planted from quart containers (Table 2.)

Table 2. 65-acre Sandhill Area, Container-grown Species and Quantities.

Pinus palustris	10,255
Quercus laevis	1,148
Quercus incana	1,492
Prunus angustifolia	136
Crategeus flava	65
Quercus chapmanii	11
Quercus geminata	150
Quercus myrtifolia	128
Carya floridana	25
Total Trees	**13,410**
Bumelia tenax	210
Callicarpa americana	174
Diospyros americana	272
Vaccinium darrowi	140
Rhus copallina	107
Viburnum obovatum	60
Ilex glabra	60
Aster caroliniana	60
Licania michauxii	420
Yucca filamentosa	432
Total Shrubs	**1,935**
Aristida stricta	20,521
Sporobolus junceus	20,188
Sorgastrum secundum	8,524
Eragrostis elliottii	7,403
Eragrostis spectabilis	6,178
Andropogon capillipes	2,122
Andropogon ternarius	836
Artistida gyrans	0
Artistida spiciformis	1,055
Total Grasses	**66,827**

Commelina erecta	622
Chapmania floridana	4
Carphephorus corymbosus	468
Passiflora incarnata	336
Chrysopsis scabrella	320
Penstemon multiflorus	1,133
Lonicera sempervirens	482
Liatris garberii	170
Dyschoriste oblongifolia	431
Galactia floridana	326
Pheobanthus grandiflorus	121
Physalis arenicola	59
Piriqueta caroliniana	32
Crotolaria rotundifolia	232
Liatris laevigata	255
Liatris gracilis	397
Berlandiera subcaulis	664
Liatris chapmanii	829
Solidago chapmanii	285
Liatris tenuifolia	168
Polygonella fimbriata	324
Salvia azurea	588
Salvia coccinea	250
Indigofera caroliniana	35
Stillingia sylvatica	8
Ruellia caroliniensis	61
Palafoxia feayi & integrifolia	182
Penstemon australis	220
Asclepias tuberosa	50
Eryngium yuccifolium	110
Sisyrinchium atlanticum	121
Baptisia lecontei	1
Desmodium floridanum	1
Scutellaria arenicola	3
Salvia lyrata	2
Elephantopus carolinianus	1
Tephrosia floridana	8
Total Forbs	**9,299**
TOTAL PLANTS	**91,471**
65 TOTAL PLANT SPECIES	

The scrub area plantings began in early September and overlapped with the sandhill planting. The same techniques and procedures were followed for growing, planting, and watering, but at a spacing of 550 plants per .4 hectare (1 acre). Forty-one species were planted, dominated by the scrub oaks, sand and slash pine, rusty lyonia, scrub hickory, and silver buckthorn. Many rare species were used, including 919 short-leaved rosemary (Conradina brevifolia), 728 Britton's beargrass (Nolina brittoniana), scrub plum (Prunus geniculata), Florida gayfeather (Liatris ohlingerae), and pygmy fringe tree (Chionanthus pygmaeus). Acorns were also direct-seeded on the little hill in October.

Table 3. 100-Acre Scrub Area, Container-grown Species and Quantities.

Pinus calusa	4,155	Yucca filamentosa	954
Quercus myrtifolia	6,667	Calamintha ashei	388
Quercus chapmanii	6,024	Chionanthus pygmaeus	18
Quercus geminata	8,691	Prunus geniculata	38
Lyonia ferruginea	1,097	Lyonia lucida	259
Carya floridana	3,239	Vitis munsoniana	130
Bumelia lacuum	1,809	Asclepias tuberosa	236
Pinus elliottii	3,209	Liatris chapmanii	514
Asimina obovata	146	Liatris ohlingerae	4
Befaria racemosa	947	Polygonella f. var. robusta	715
Conradina brevifolia	919	Aristida gyrans	490
Garberia heterophylla	387	Palafoxia feayi	85
Hypericum reductum	1,128	Sisyrinchium solsitiale	91
Ilex arenicola	98	Commelina erecta	38
Persea humilis	715	Tradescanta roseolens	64
Vaccinium darrowi	1,498	Ximenia americana+oak	30
Licania michauxii	1,354	Scutellaria arenicola	6
Pityopsis graminifolia	6	Galactia floridana	52
Polygonella polygama	5	Andropogon capillipes	158
Nolina brittoniana	728	**TOTAL PLANTS**	**47,092**

RESULTS AND DISCUSSION

The authors believe that mixing overburden, which has a high clay content, with sand tailings was an essential step in developing a usable soil on this reclaimed land. since the pans cannot put down an even 6-inch layers, some areas had higher clay and silt content, even after mixing. Robert Hopper, a graduate student at Harvard University, has been comparing soil types with weed growth along four 91.4-meter (300-foot) transects. Preliminary results suggest that where the silt and clay component is above 10 percent, the cogan grass and hairy indigo weeds are much more aggressive.

The hydroseeding was an attempt to mass seed a variety of species onto a site to attain more diversity and to offer an economically-viable alternative to exotic grass groundcovers in future reclamation projects. Having more native grass seed in the mix in future projects would give better erosion control. The hydroseeding produced a good cover of partridge pea and many other forbs. The partridge pea is still reseeding well in the second season. Many of the seeded forbs were flowering and producing seeds the fall of the same year that they were planted. The sand tailings areas without the overburden mix, however, did not have much germination from the hydroseeding mix, nor have many weeds entered these areas.

A year after the seeding of the palmettos (summer 1994), Robert Hopper took five samplings of 15.2 meter (50 foot) by 15.2 meter (50 foot) areas. Though he has not yet finished his research, he advises that the samples showed about a 36%

germination rate on the combined palmetto species. This costed out to about $0.25 per germinated seedling.

After the first spring dry season in 1994, it was estimated that over 95% of the plants were alive and there was much spring growth. The augers with large bits prepared excellent planting holes. Roots grew out from the container media very rapidly; after one month the root mass of the grass tubelings grew to 25 centimeters (10 inches) across. Watering the plants in well after planting was important, even during the rainy season, but additional waterings may not have been as necessary. The plantings on the sand tailings are also surviving well; those on high, exposed slopes have the highest losses. Losses are also greater where the soil has a higher clay and silt content and the subsequent weed growth is greater.

It has been one and a half years since the project has been completed. The authors have noted flowering and seeding in many species. The same fall of the planting all of the grass species began producing seeds except lop-sided indiangrass (Sorgastrum secundum), which bloomed the following fall. Most herbaceous perennials also produced some seed the year of planting and seeding. Obvious seedlings of some species such as Liatris tennuifolia and Polygonella fimbriata robusta were observed. The cogon grass has become a severe problem, especially on the western slope and will need to be treated, though extensive treatment has now been given to the surrounding areas which have been a seed source. The longleaf pines have been candling to heights of 2 meters. Many scrub plants such as little blueberry (Vaccinium darrowi) and rusty lyonia (Lyonia ferruginea), have established very well and continue to put out new growth though they have been known to be difficult to establish in home landscapes. After the second spring drought period, a few losses were seen, but mostly new spring growth.

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THE COCOHATCHEE STRAND RESTORATION PROGRAM: A UNIQUE PROJECT-SPECIFIC MITIGATION BANK

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ABSTRACT

A wetland restoration and enhancement program was designed to function as an on-site mitigation bank for a development project in Naples, Florida. The isolated 54-hectare wetland system (Cocohatchee Strand) consisted of predominantly Palustrine forested and scrub-shrub communities. Wetland functions had been adversely affected primarily by exotic plant infestations and by a severely degraded hydroperiod. The restoration program was developed based upon an extensive investigation of existing site conditions. These efforts included water table monitoring, topographic surveys, wildlife surveys, vegetation mapping, stratigraphic characterization, specific capacity and constant-rate aquifer testing, and surface and groundwater modelling using ICPR, HEC-2, and MODFLOW. The resultant program called for conversion of exotic monocultures to native forested wetlands and marshes combined with innovative hydrologic restoration of the entire Strand. Hydrologic restoration included reconnection of Strand wetlands by constructing a land bridge over a canal that severed these wetlands. It also included use of pumps to divert water from this canal and lakes into the Strand. Predictive MODFLOW models were run prior to committing to the pumping scenario to ensure target hydroperiods were attainable. The mitigation bank formed by the restoration program was established using a sliding scale of "credits" linked to restoration success criteria. A bank withdrawal system was created based on a matrix of quality ratings for wetlands which may be impacted by development.

INTRODUCTION

The Cocohatchee Strand is a 54-hectare (133 acres) primarily Palustrine forested wetland system approximately 2.4 kilometers (1.5 miles) long. It is situated within an 842-hectare (2,080 acres) development project known as the Pelican Marsh Community (PMC). Pelican Marsh is being developed by WCI Communities Limited Partnership (WCI) as a mixed-use project with primary land uses being residential homesites and golf courses. In the initial planning stages of this community, WCI recognized the Strand as an important natural resource which could benefit PMC, future residents, and the region. However, these benefits would only be realized if the degraded Strand wetland system was restored and managed. Several other isolated wetlands of much lower functional value than the Strand were scattered throughout the PMC property. WCI also recognized the need for early establishment of a wetland mitigation plan that could serve over the duration of phased development which could impact these wetlands.

To meet WCI's goals, a restoration program was developed with two main objectives. The first was to restore and enhance Strand wetlands to approximate historic wetland conditions and functions. The second was to establish an on-site mitigation bank which could compensate for necessary future development impacts to other less significant wetlands found in PMC. This paper provides an overview of factors which had adversely affected the Strand, some of the studies conducted in designing the restoration program, as well as the final design and mitigation banking system.

PROJECT SITE

The Pelican Marsh Community occupies portions of Sections 25, 27, 34, 35, and 36, Township 48 South, Range 25 East, in the northwest corner of Collier County, Florida, near the City of Naples. The freshwater wetlands comprising the Cocohatchee Strand form a narrow, linear system that bisects the western portion of PMC. The project site lies within the Cocohatchee River Basin and the Strand wetland system historically drained into the Cocohatchee River which is located approximately 1.6 kilometers (1 mile) north of the northern tip of the Strand.

PMC represents an "infill" development being bounded on the west by U.S. 41 and its adjacent commercial and residential areas, on the south by residential neighborhoods, on the north by agricultural fields slated for conversion to a regional mall, and on the east by agricultural fields and adjacent residential neighborhoods. Prior to any development, PMC primarily consisted of active and abandoned agricultural fields, pine flatwoods, palmetto prairies, cypress and mixed pine and cypress forests, disturbed lands, and exotic monocultures.

DESIGN STUDIES AND SITE CONDITIONS

Site investigations began in mid-1991 and continued through 1993. Wetland boundaries were delineated following protocols described by the Corps (USACOE, 1987). Dominant vegetation associations were characterized and mapped through field surveys, aerial reconnaissance, and interpretation of aerial photographs. Particular attention was given to mapping exotic plant infestations. Topography was determined by surveying elevations at fixed intervals along 5 transects positioned through the Strand, surveying spot elevations at key locations, and using data from previously flown aerial topographic maps.

Determination of existing Strand hydroperiods was accomplished by monitoring elevations of the water table aquifer. Fifteen piezometers were installed in Strand wetlands. Water table elevations were typically recorded on a weekly basis during the wet season and monthly during the dry season over a 2-1/2 year period. Over 30 piezometers were installed in surrounding uplands and other project wetlands with water table elevations recorded on a similar schedule.

To determine representative hydraulic parameters for the water table aquifer, specific capacity and extended duration constant-rate aquifer testing was performed by ViroGroup (ViroGroup, 1993). Testing was performed using 3 wells established outside the Strand. The wells were screened through the water table aquifer and terminated in or at the top of the underlying confining layer. In order to characterize site stratigraphy, ViroGroup drilled 3 holes in the Strand and 6 holes in surrounding uplands with boreholes ranging in depth from 8 to over 36 meters (25 to >120 ft.). Lithologic descriptions obtained from these holes were used to construct cross-sections across the site.

Site observations indicated severe degradation of Strand hydrology and hydroperiods. Review of historic aerial photographs coupled with these investigations helped explain the causes. The Strand had once been part of a much larger wetland system in which water flowed from south to north discharging to Horse Creek and subsequently the Cocohatchee River. During the period from 1952 to 1992, over 144 hectares (355 acres) were lost leaving 57 hectares (141 acres) remaining as shown in Figure 1. The extent of uplands draining into the Strand was also drastically reduced to the minimal area shown in Figure 1.

In the early 1900's a railroad was built through the Strand. Goodlette-Frank Road (GFR) was later constructed along its course in 1992. As shown in Figure 1, two culverts allowed surface water to continue flowing under the road to northern Strand wetlands but flows were restricted and flow patterns altered. A spur railroad grade and adjacent drainage ditch remained east of GFR. These features also disrupted normal Strand sheetflow patterns and the ditch overdrained adjacent wetlands. Pine Ridge Canal was built in the early 1970's. This canal and the spoil grades along its banks severed the northern Strand wetland from the two southern portions and the sheetflow they once provided. Farmers installed culverts in the Strand wetland adjacent to the canal on its eastern side (see Figure 1), with the culverts discharging into the canal. Since the canal is controlled at elevation 6.0 ft. NGVD in this area and the average Strand ground elevation is about 9.0 ft., these culverts served to overdrain Strand wetlands east of the canal.

Farmers also drained nearby agricultural fields using pumps at the locations shown in Figure 1. The discharge drained into the Strand. This pumping helped support the wetlands which were badly in need of water inputs, however the timing desynchronized the normal hydroperiod. Water was pumped for a few weeks at the end of the wet season when water levels in the wetlands would normally be declining and it was supplied sporadically in large volumes which resulted in rapid changes in Strand water levels.

Strand hydrologic conditions were obviously degraded and would only become worse given that the agricultural pumping would cease upon development of PMC. Thus restoration of hydroperiods and hydrologic patterns was identified as the primary element of the Strand restoration program. One component of this effort was to re-establish the physical hydraulic connection between Strand wetlands severed by Pine Ridge Canal which would also alleviate problems caused by the culverts draining these wetlands. Additional water inputs would still be required to offset the loss of historic water sources.

A passive method of adding water to the Strand via gravity flow was determined to be infeasible. Pine Ridge canal provides drainage for off-site residential areas, thus its control elevation could not be raised enough to allow flows to enter the Strand

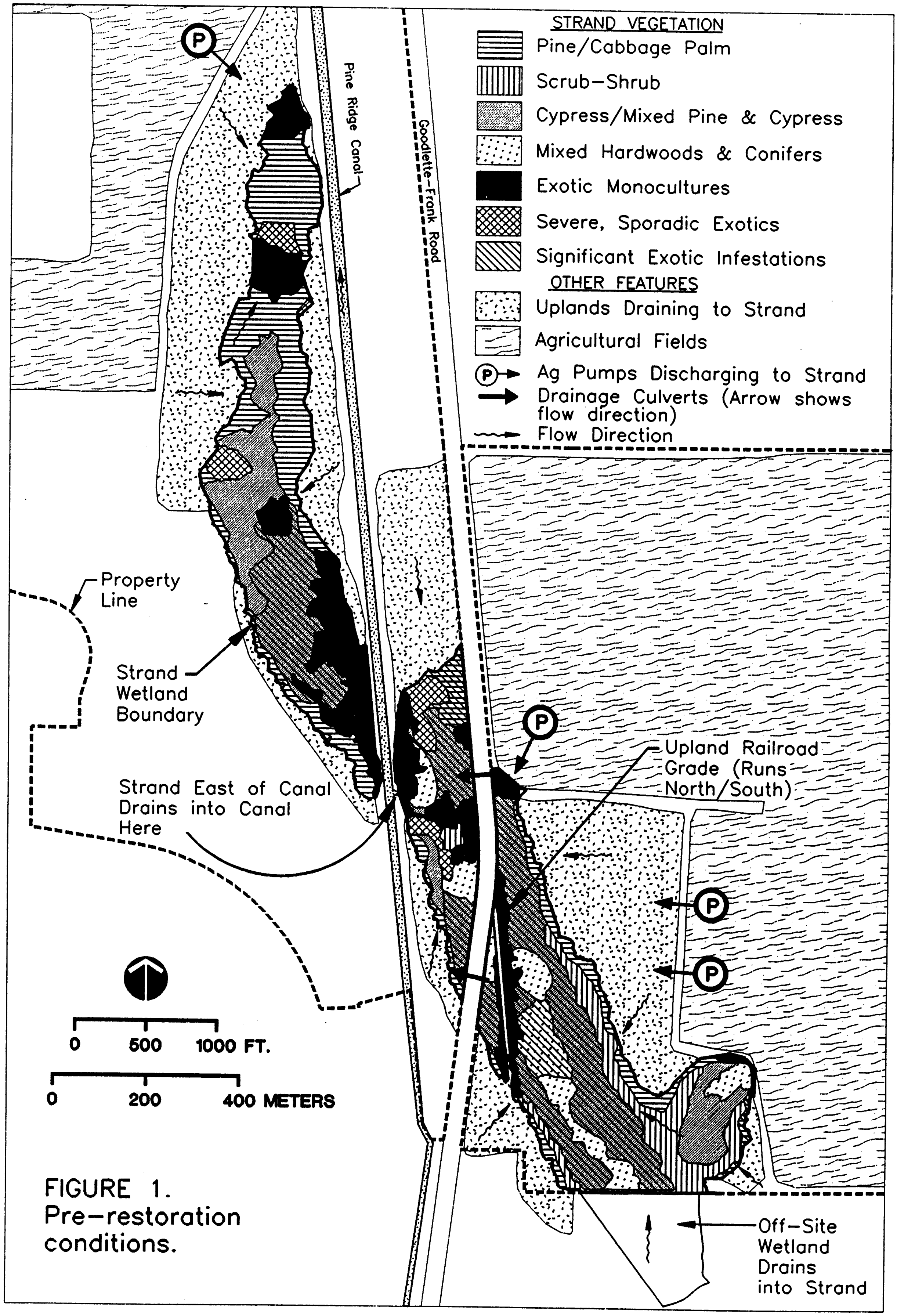

STRAND VEGETATION
Pine/Cabbage Palm
Scrub–Shrub
Cypress/Mixed Pine & Cypress
Mixed Hardwoods & Conifers
Exotic Monocultures
Severe, Sporadic Exotics
Significant Exotic Infestations
OTHER FEATURES
Uplands Draining to Strand
Agricultural Fields
Ag Pumps Discharging to Strand
Drainage Culverts (Arrow shows flow direction)
Flow Direction
Pine Ridge Canal
Goodlette–Frank Road
Property Line
Strand Wetland Boundary
Strand East of Canal Drains into Canal Here
Upland Railroad Grade (Runs North/South)
Off–Site Wetland Drains into Strand
0 500 1000 FT.
0 200 400 METERS
FIGURE 1.
Pre–restoration conditions.

Figure 1

without flooding these upstream neighborhoods. It would be desirable to construct PMC lakes using control elevations which would allow the lakes to drain into the Strand. Data gathered from the piezometers indicated that area water table elevations were too low to support such lake control elevations. Active pumping of surface water into the Strand was therefore pursued as the only available option for achieving the necessary water inputs.

Target Strand hydroperiods were first developed. It was known that PMC development would require a new roadway crossing of the Strand. Given this new roadway, the drainage divide formed by GFR, the assumption that Strand wetlands severed by the canal would be reconnected, and existing topography, three drainage basins would be formed in the Strand. Typical pre-restoration hydrographs were constructed for each of these 3 basins using the piezometer data. Target hydrographs were then developed in a manner that mimicked the general form of the pre-restoration hydrographs but increased both the depth and duration of wetland inundation. Field studies had included estimating Strand historic seasonal high water table elevations based on biological indicators such as tree buttressing and remnant tussocks. The target "hydroperiods" developed provided peak water elevations lower than these historic highs but which were 0.15 to 0.30 meters (0.5 to 1.0 ft.) above existing seasonal high water levels. Figure 3 depicts the target hydrographs developed for the three Strand drainage basins.

A groundwater modeling study was then performed by ViroGroup to determine if the target hydroperiods were attainable through pumping and to estimate the volume of water required (ViroGroup, 1993). They utilized the MODFLOW program (McDonald and Harbaugh, 1988) for this study. Model input packages included: hydraulic parameters (horizontal and vertical hydraulic conductivity, transmissivity, etc.), determined from site investigations; recharge, based on average historic precipitation; evapotranspiration, based on average historic Class A pan evaporation; and, initial water table elevations or heads, using on-site water table elevations recorded in late May of 1992. The model used a uniform grid of 4,200 cells each measuring 45.7 x 45.7 meters. It utilized 4 layers: one for the top of the water table aquifer (representing the canal and PMC lakes); one for the upper water table aquifer beneath the Strand; one for the lower water table aquifer; and, one for surface water with this layer not activated until predicted water table elevations exceeded the soil surface. The model was calibrated against the data gathered from the piezometers.

Two scenarios were modeled. Both predicted water table elevations under post-development conditions, with one model assuming the Strand hydrologic restoration program was in place and the other assuming its absence. To model the scenario with the restoration program operative, numerous iterations were first run adjusting the pumping input volumes until predicted water elevations best fit the desired target hydroperiods. The iteration with the best fit therefore estimated the volume of water required to achieve these hydroperiods. Water table elevations predicted by the models were examined for key locations in the Strand in order to determine the "best fit" iteration and evaluate the overall benefits of the hydrologic restoration program. These data were also Krieged to produce contour maps for similar review.

Surface water modeling was conducted to determine if a sufficient volume of water would be available to the pumps. The future PMC surface water management lakes and Pine Ridge Canal had been identified as the potential water sources. ICPR modeling (Singhofen, 1990) simulated post-development conditions to predict peak stages, storage, and discharge within the PMC surface water management system. This modeling allowed estimation of the volume of water available in future PMC lakes and the volume of water that would be discharged to the canal which was to be the final outfall of the PMC surface water management network. HEC-2 (USACOE, 1982) is a hydraulic analysis program for rivers, canals, etc., which predicts peak stages in such waterways. Coupled with the ICPR modeling, HEC-2 modeling was performed to determine the volume of water available in the canal under post-development conditions.

A preliminary restoration program was prepared based on our studies. It was designed following principles described by various authors (Hammer, 1992; Kusler and Kentula, 1990; Marble, 1992). This program was reviewed with representatives from various regulatory agencies to obtain their input and to develop the mitigation banking system. These agencies agreed upon a "sliding scale" method of calculating mitigation credits generated by the Strand program. This method produces a greater number of credits as mitigation success criteria are accomplished over time. A system for withdrawing credits from the bank based on the relative functional values of wetlands which might be impacted by PMC was also recommended. To develop this credit withdrawal system, all PMC wetlands were mapped and classified according to dominant vegetation associations and functional conditions.

RESULTS AND DISCUSSION

Strand Restoration Program

Figure 1 illustrates the dominant vegetation associations mapped in the Strand. Severe infestations by the exotics Brazilian pepper (Schinus terebinthifolius) and melaleuca (Melaleuca quinquenervia) were present. These exotics formed monocultures in certain areas while elsewhere exotics were dominant but restricted to relatively discrete pods leaving intermixed clusters of viable native vegetation (areas labeled "severe, sporadic exotics"). In much of the Strand, native trees dominated the canopy but there were significant infestations of exotics in the midstory. In such areas, the midstory cover accounted for by exotics ranged from 10 to over 90 percent, depending on location.

Figure 2 depicts the final Strand restoration program developed. A key element of this program is ridding the Strand of exotics and re-establishing native wetland plant communities. Areas of exotic monocultures and certain uplands are being restored to native systems by clearing, regrading, and planting the cleared/regraded areas with native wetland species. In this manner, 6 freshwater marshes totalling 4.1 hectares (10.1 acres) and 11 forests totalling 3.8 hectares (9.5 acres) are being established. The marshes are being planted with a variety of herbaceous species as well as scattered shrubs primarily along marsh perimeters. The forests are being planted with a mixture of coniferous and hardwood trees and a variety of shrubs, grasses, and

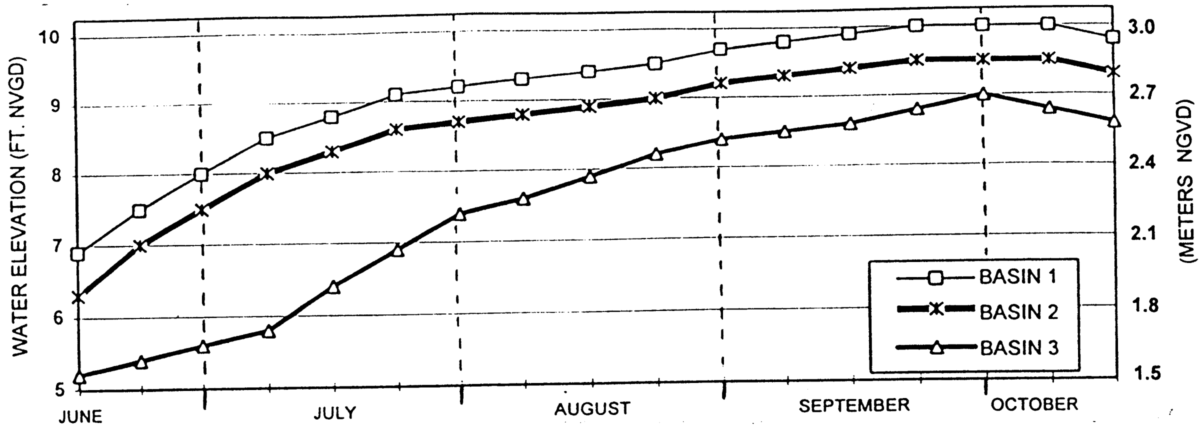


Figure 3. Target hydroperiods developed for each of the three Stand drainage basins.

forbs. The planting scheme seeks to produce an admixture dominated by forested areas with smaller pockets of wet prairies. In the disturbed areas where exotics were dominant but restricted to discrete pods, the exotics are selectively removed and the cleared portions replanted as native wetland forests using a variety of trees, shrubs, and herbaceous species characteristic of the native communities saved during the selective clearing process. Six areas totalling 2.2 hectares (5.4 acres) are being restored in this fashion.

In the areas where native trees are dominant but there are numerous exotics in the midstory, the exotics are being eradicated by physical removal and/or directed herbicide applications. Any large clearings left by the exotic removal process are then being replanted to mimic adjacent native plant communities. These intensive exotic eradication/replanting efforts involve 6 areas totalling 16.8 hectares (41.5 acres). It is noted that the Strand program includes eradication and control of exotic and nuisance plant species throughout the Strand.

To add another element of habitat diversity, the Strand program includes creation of forested hammocks. Five hammocks are being established in areas formerly comprised of exotic monocultures and uplands. Here, mounds approximately 1 meter above existing grade are formed and then planted with transitional wetland and upland species. Much of the former railway spur grade east of GFR is being removed. Other portions with large native trees present are retained and their grade partially reshaped to form 3 additional hammocks. In these, exotic and nuisance plants which dominated the understory are being removed and native trees, shrubs, grasses, and forbs added. The 8 hammocks total 0.6 hectares (1.4 acres).

One should note that the precise location and configuration of Strand restoration features shown in Figure 2 differs from that originally permitted by the regulatory agencies. The original layout of areas to be restored to marshes, forests, and hammocks had been based on field observations and photo-interpretation. Prior to any clearing or regrading activities, this original layout was staked out in the field using standard survey methods. Biologists then inspected this layout, adjusting it as necessary to save viable native plant communities and encompass additional areas of severe exotic infestations and exotic monocultures not identified in the initial design process. These adjustments were surveyed and incorporated into the preliminary design so as to maintain the desired restoration objectives. Figure 2 depicts the resulting final design which was subsequently approved by the regulatory agencies.

Hydrologic restoration, the most important facet of the Strand program, involves two components: hydraulic reconnection of Strand wetlands and other activities to enhance surface water flow patterns; and, active pumping of water into the Strand to enhance and restore hydroperiods. The groundwater modeling study indicated that the desired Strand target hydroperiods (see Figure 3) were indeed attainable as illustrated in Figure 4. To reach the target hydroperiods, the MODFLOW study predicted the required pumping volumes shown in Figure 5. The surface water modeling study predicted that PMC lakes and Pine Ridge Canal could provide sufficient water to meet these pumping demands.

The pumping system utilizes twin 1,000 gpm, 5 horsepower, submersible vertical mixed-flow pumps at 2 pump stations, P1 and P2. Pump P1 can draw water both from the canal and from one of the PMC lakes shown in Figure 2. This lake is connected to P1 by an underground 76 cm (30") reinforced concrete pipe (RCP). The PMC lakes are interconnected via culverts providing a larger water source than just the single lake. Pump P1 discharges to a feeder pond constructed in an area formerly comprised of uplands and exotic monocultures. The feeder pond serves to dissipate energy and provide water quality treatment. Flow is distributed from this pond into adjacent wetlands via four 38 cm (15") RCPs attached to grate inlets. Pump P2 obtains water from another of the interconnected PMC lakes. The discharge is routed to a shallow basin lined with rip-rap to reduce erosion and then naturally drains northward through the Strand.

Pump P2 also indirectly obtains water from off-site. A small flood containment berm was built around the entire Strand for water management purposes. The off-site wetland shown in Figure 1 once flowed northward directly into the Strand. This wetland serves as the drainage outfall for a neighboring subdivision which has a surface water management system controlled at elevation 2.9 m (9.5 feet) . Since the seasonal high water elevation targeted for the adjacent Strand drainage basin (basin 1) is 3.1 m (10.0 feet), the flood containment berm was built to hydraulically isolate this off-site wetland from the Strand in order to prevent flooding of the neighboring subdivision. To maintain a positive outfall for this subdivision, grate inlets were installed at grade outside the perimeter berm on the Strand's southern end. These inlets receive the off-site drainage and carry it through 91 cm (36") RCPs into the pump P2 lake which has a control elevation of 2.4 m (8.0 feet). Pump P2 thus draws this water and routes it back into the Strand.

The pumps are activated at the beginning of the wet season (June) and deactivated near the end of the wet season (late Sept./mid-Oct.). Pumping is used only to supplement inputs from direct rainfall and groundwater inflows, thus target hydroperiods are only attainable during years having near average rainfall. Pumping is not used during the dry season since this would not mimic historic conditions and there would be insufficient water available to the pumps. The pumps are operated by timers with automatic float shut-offs which are activated when source water elevations are excessively low or high. Presently, the number of hours the pumps are run is

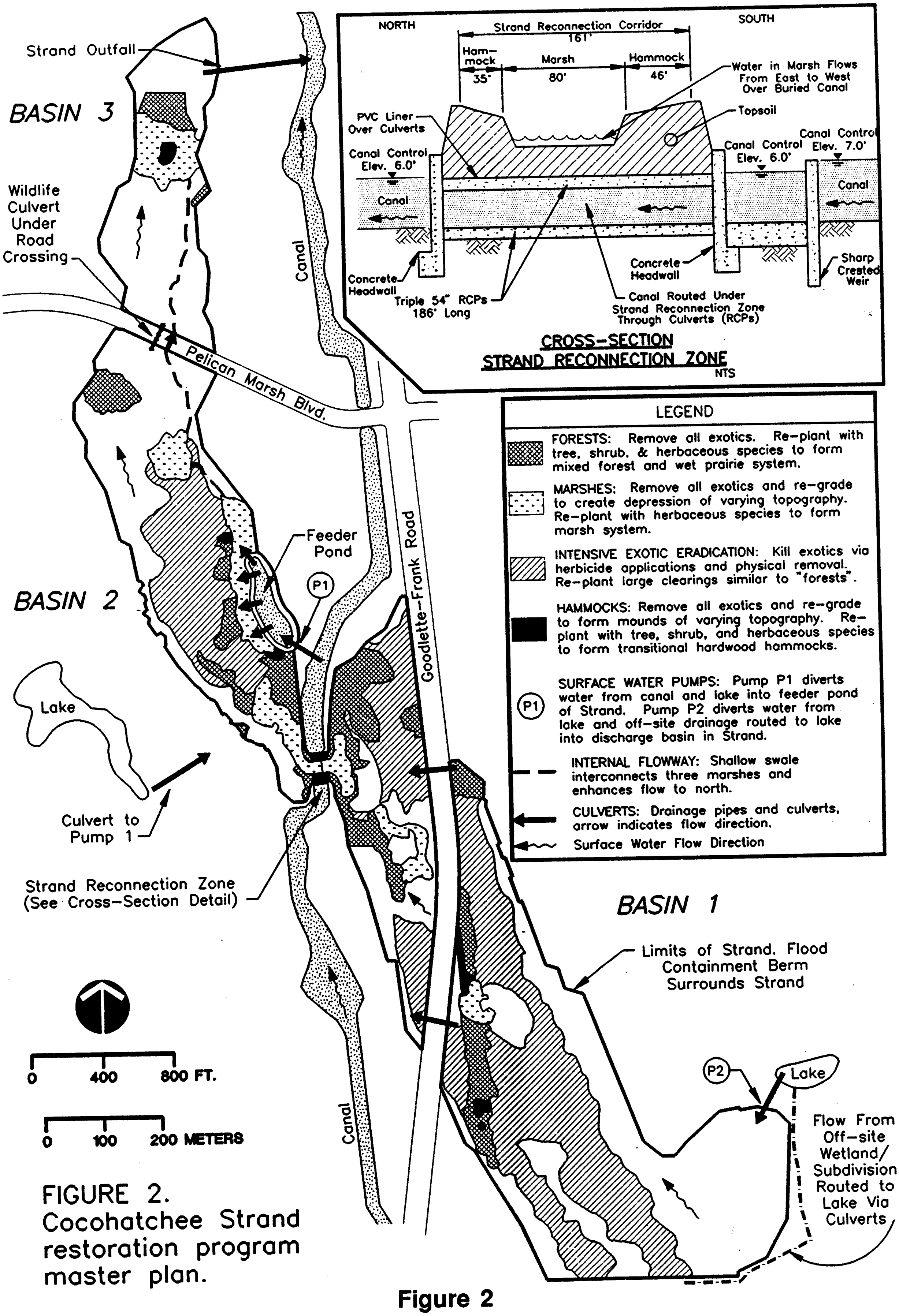


FIGURE 2. Cocohatchee Strand restoration program master plan.

Figure 2

Figure 4. Strand hydrograph predicted by MODFLOW for post-development conditions with the hydrologic restoration program in place compared with hydrographs for the measured pre-development hydroperiod and desired target hydroperiod. All hydrographs are for a point near the center of the strand.

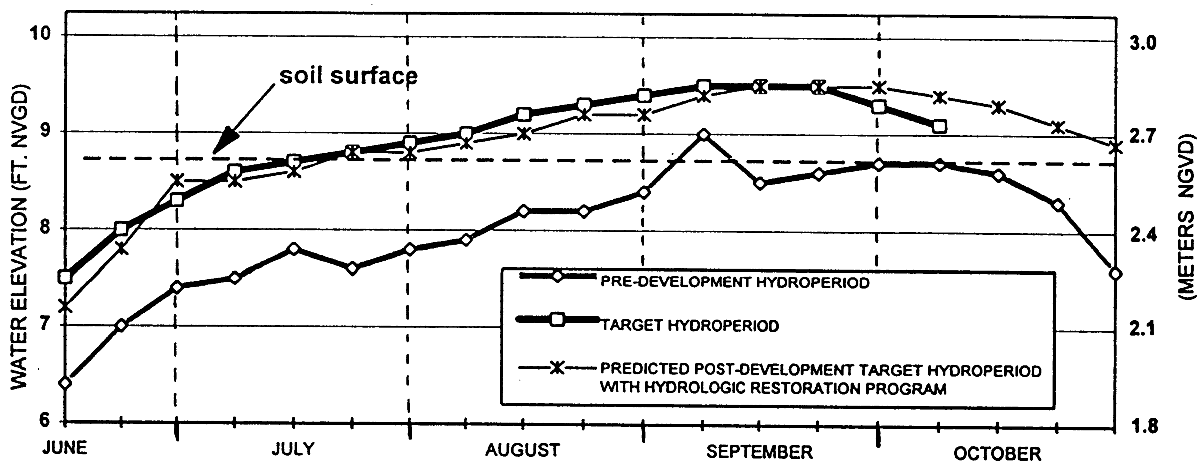
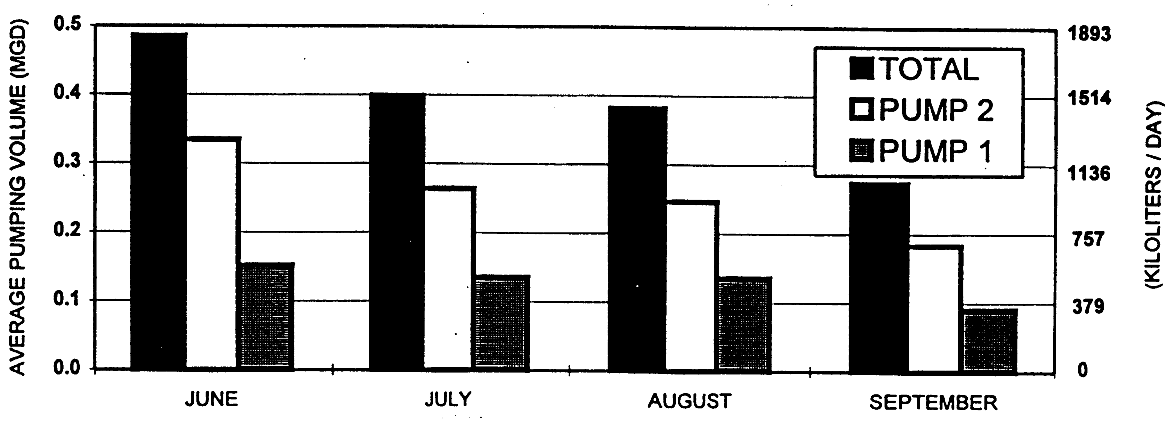


Figure 5. Average pumping volumes (million gallons per day) necessary to achieve target Strand hydroperiods predicted by MODFLOW.



determined by weekly monitoring of changes in Strand water elevations in response to pumping and rainfall.

The problem of re-establishing the hydraulic connection of Strand wetlands severed by Pine Ridge Canal was solved by routing the canal under wetlands recreated above the canal at the "Strand reconnection zone" shown in Figure 2. Three 1.4 m x 57 m (54"x 186') RCP's were installed in the canal which was reconfigured during the PMC development process. A 30-mil PVC liner was placed on top of the culverts to prevent seepage, followed by layers of topsoil and muck. Above the "buried" canal, a marsh was created between 2 hammocks. This marsh allows water from the Strand situated east of the canal to flow across the canal into western portions of the Strand. Flow in the canal is maintained through the triple RCP's buried beneath the reconnection zone. This hydraulic reconnection not only returned historic surface water flows to the Strand situated west of the canal, but also alleviated the overdrainage caused by the culverts which once bled water from the Strand into the canal near the reconnection zone.

The hydraulic design of the Strand employed a series of 3 cascading drainage basins. Basin 1 lies east of GFR. Basin 2 lies between GFR and Pelican Marsh Boulevard which was built to link neighborhoods within PMC. Basin 1 is that portion north of the boulevard. Control elevations of the 3 basins were set to be slightly below the target peak Strand water elevations, assuming water levels would stage high enough above control to meet these target peaks as predicted by the MODFLOW and ICPR models. The control elevations were set at 2.8, 2.7, 2.5 m (9.3, 8.7, and 8.3 feet NGVD) for basins 1, 2, and 3, respectively. The existing culverts under GFR were extended to pass under the Strand flood containment berm and their control elevations modified. A 61 cm x 34 m (24" x 110') RCP with grate inlets on either end was installed to convey water under Pelican Marsh Blvd. Note that a 1.2 m x 1.5 m (4'x 5') box culvert 90 feet long was also constructed beneath this road to allow safe passage of wildlife. Small berms built at each end prevented the passage of stormwater through this wildlife culvert. A catch basin was constructed at the northern end of the Strand. This forms the final Strand outfall with drainage routed into Pine Ridge Canal via a 61 cm (24") RCP connected to the catch basin.

A shallow flowway (avg. 2' deep x 22' wide or 0.6 m x 6.7 m) was built within the Strand to help route flows northward during low water conditions. Pelican Marsh Blvd. was built along a slight drainage divide thus the flowway helps water overcome this divide when water elevations have not yet staged high enough to naturally flow northward across it. The flowway was also designed to interconnect three of the restored marshes providing an avenue for migration of fish, amphibians, and other aquatic organisms. The Strand program also included removal of much of the abandoned railroad grade and its adjacent ditch to help re-establish normal sheetflow patterns within basin 1. The berm forming the railroad grade was excavated and the ditch filled such that the resultant grade matched that found in adjacent undisturbed wetlands. Areas formerly containing the regraded berm and ditch were included in the forest and marsh recreation efforts.

Strand Mitigation Banking System

The mitigation bank credit accrual matrix is provided in Table 1. It is based on a sliding scale of credit accrual. The number of mitigation credits provided by a particular restoration feature category increases over time as three well-defined levels of success are achieved. Success criteria are unique to each restoration feature category and require the passage of time and achievement of additional measures of success to progress from one success level to the next. For a given restoration feature category (ex., marshes) the number of credits generated at a particular success level is calculated by dividing the acreage of the restoration feature by a set mitigation ratio. This ratio decreases as greater levels of success are achieved thereby generating more credits. The principle involved is that there is limited certainty of long-term mitigation success early in the program but this certainty improves over time, thus progressive levels of success are rewarded with increased credits.

Table 1. Cocohatchee Strand mitigation bank credit accrual matrix.

STRAND RESTORATION FEATURE	TOTAL ACRES OF FEATURE	MITIGATION SUCCESS CRITERIA ACCOMPLISHED					
		1		2		3	
		Ratio	Credits	Ratio	Credits	Ratio	Credits
Marshes	10.1	1.7	5.9	1.4	7.2	1.2	8.4
Forests	14.9	3.0	5.0	2.2	6.8	1.2	12.4
Feeder Pond Littoral Zone	0.4	1.7	0.2	1.4	0.3	1.2	0.3
Intensive Exotic Eradication	41.5	5.0	8.3	4.0	10.4	3.5	11.9
Hydrologic Restoration: Basin 1	56.9	4.5	12.6	3.5	16.3	3.0	19.0
Hydrologic Restoration: Basins 2 & 3	76.0	4.0	19.0	3.0	25.3	2.0	38.0
Total Mitigation Credits Generated			51.0		66.3		90.0

Credits = (Restoration feature acreage) ÷ (mitigation ratio).
Ratio = Mitigation ratio = acres mitigation required to offset impacts to 1 acre of existing wetland which has functional qualities and capabilities equal to those of the **restored** Strand wetland system. There are 3 sets or levels of mitigation success criteria which are unique to each of the various restoration features. To progress from one success level to another requires meeting specified measurements of success which change over time.

The credit accrual matrix was set up to account for several possible permutations of mitigation success. For example, at a given point in time if half the marshes had obtained success level 3 while the other half had only reached success level 1, the total credits generated by all marshes could still be calculated by applying the 1.2 ratio to the acreage of marshes at level 3 and applying the 1.7 ratio to the acreage of those still at level 1. The sliding scale also provides an incentive to complete mitigation to the greatest success level possible before withdrawing credits. If the bank is completely depleted of credits at early success levels, no further credits can be generated or withdrawn even though further success might be obtained.

The mitigation bank credit withdrawal matrix established is provided in Table 2. To determine the number of bank credits which must be withdrawn to compensate for project wetland impacts, one first identifies the type of wetland to be impacted and its level of degradation or disturbance. A credit withdrawal value (CWV) is selected from the range of values specified in the appropriate matrix cell. The CWV is then multiplied by the acreage of wetland to be impacted yielding the number of bank credits to be withdrawn. If several wetland types and disturbance combinations are involved, withdrawal calculations are made separately for each combination and the results summed to produce the total bank withdrawal required.

Table 2. Cocohatchee Strand mitigation bank credit withdrawal matrix.

EXISTING WETLAND TYPE IN PROPOSED IMPACT AREA	Range of Credit Withdrawal Values According to Existing Degree of Disturbance Found in Wetland to be Impacted			
	High Disturbance	Moderate Disturbance	Low Disturbance	Undisturbed
Forested, Cypress	0.4 - 0.5	0.5 - 0.7	0.6 - 0.8	0.8 - 1.0
Forested, Mixed Hardwoods & Conifers	0.4 - 0.5	0.5 - 0.6	0.6 - 0.8	0.8 - 1.0
Scrub-shrub, Native Vegetation	0.2 - 0.4	0.3 - 0.5	0.5 - 0.7	0.6 - 0.9
Emergent, Herbaceous Marsh or Wet Prairie	0.1 - 0.2	0.2 - 0.4	0.4 - 0.6	0.5 - 0.9
Forested, Mixed Pine & Cypress	0.3 - 0.4	0.4 - 0.5	0.6 - 0.7	0.7 - 0.8
Forested, Pine or Cabbage Palm	0.2 - 0.4	0.3 - 0.5	0.5 - 0.7	0.6 - 0.8
Forested, Melaleuca; OR Scrub-shrub, Brazilian Pepper	0.1 - 0.2	0.1 - 0.3	0.1 - 0.3	0.1 - 0.3
Man-made, Open Water/Littoral Zone	N/A	N/A	N/A	0.2 - 0.4

Credit to be withdrawn from bank = (selected credit withdrawal value) X (acreage of wetland impacted).
For a given wetland type, credit withdrawal value is selected from range of values indicated for the level of disturbance exhibited by that wetland.Disturbance includes factors such as burns, nuisance plant infestation, drainage, degraded hydroperiod, clearing, vehicular damage, topographic alterations, etc. Exotic plant infestations automatically dictate assignment of disturbance categories as follows: Low: 10-33% exotics; Moderate: 34-65% exotics; High: > 65% exotics.

The possible types of wetlands which might be impacted by future PMC development were categorized based on the field mapping efforts. Wetland functional quality ratings were assigned based on the type of wetland, the existing degree of wetland disturbance, and principles described in the USACOE's Wetland Evaluation Technique (Adamus et al., 1987; Adamus et al., 1991). These ratings were equivalent to the CWV's of the bank withdrawal matrix. Undisturbed wetland types providing numerous functional values were assigned the highest rating (1.0) while highly disturbed wetland types providing limited functions were assigned the lowest rating. These ratings were also assigned based on the assumption that the fully restored Strand would have the highest rating of 1.0. Thus, a wetland to be impacted which is given a rating of 0.6 is deemed to provide 60% of the wetland functions and values provided by the fully restored Strand.

Note that the bank withdrawal matrix never requires the withdrawal of more than one credit for each acre of wetland to be impacted, whereas common practice usually requires a mitigation ratio exceeding 1:1 (acres of mitigation required for each acre of wetland impact). This is a result of the fact that common mitigation ratios are employed in the bank credit accrual matrix. For example, the 57 acres of Basin 1 hydrologic restoration produces only 12.6 credits at success level 1 because a mitigation ratio of 4.5:1 has been applied in the bank accrual matrix. A mitigation ratio is thus not necessary in making bank withdrawals. CWV's are used instead to account for the fact that a lower mitigation ratio would have normally been required to mitigate impacts to degraded wetlands. One should also note that withdrawals from the bank are not made on a "type-for-type" mitigation basis. If a forested wetland is to be impacted, the necessary bank credit withdrawal does not have to be deducted from credits that have accrued specifically from restoration of wetland forests. The withdrawal is simply made from the total credits which have accrued regardless of the source of these credits.

CONCLUSION

The Cocohatchee Strand restoration program was implemented in early 1994 and is now well underway. Many restoration activities have been completed and have reached the first level of success as documented by an extensive monitoring program. The mitigation banking system is operational and credits have been both generated and withdrawn.

From the outset of the Pelican Marsh project, the developer, the developer's consultants, and regulatory agency staff worked closely together which helped foster innovative solutions to complex wetland and development issues. The project was planned using a holistic approach rather than in discrete stages. Mitigation efforts focused on restoring larger integrated wetland systems rather than restoring or creating smaller isolated wetlands. We believe the resulting Strand restoration program and mitigation bank provide a good example of the benefits to both the environment and the development which can be derived from using such an approach.

ACKNOWLEDGEMENTS

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REAL-WORLD CONSIDERATIONS FOR MICROPROPAGATING Ruppia maritima L. (WIDGEON GRASS) FOR SEAGRASS RESTORATION PROJECTS

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ABSTRACT

Although laboratory techniques for in vitro micropropagation of widgeon grass Ruppia maritima L., have been established, plant-management considerations for regulating the production of large numbers of planting units of this seagrass have not been addressed. Six years of experience in testing techniques for culturing and planting Ruppia have provided us with some practical procedures that maximize both the rates of tissue propagation and the vigor of plant material used in field plantings. Turnaround time from the collection of wildstock for micropropagation to the delivery of rooted plants of an acceptable size (> 100 nodes) for use in a field planting may be as little as 12 weeks. However, delays may occur. Sterilization procedures for each new population may need to be adjusted because of new, unidentified contaminants -- additional time is often required to identify appropriate sterilants needed to clean up a new population of Ruppia. Type of culture vessel, volume of growing medium, and size of the initial explant all affect the growth, size, and quality of a field planting unit. In addition, the number, "culture age" (i.e., the number of days a plant has been in sterile culture), and growth rates of stock culture plants affect the number of field planting units that can be produced in a given amount of time. Thus, to consistently produce planting units, multiple populations of various "culture ages" should be in culture simultaneously. Monitoring the "culture age" of stock and growout plants is also critical in predicting how rapidly they will deplete the media, when they should be subdivided and transferred to fresh media, and how much time will be required for a particular cohort of plants to root. Root production is also slowed over time due to "culture age," and the health of plants under these conditions is compromised if they are held too long under culture conditions before being planted.

INTRODUCTION

In vitro micropropagation, a type of plant tissue culture, is widely used in commercial horticulture for mass-producing many species of plants -- from nursery-stock ornamentals to fruits and vegetables to field crops. Using this type of propagation system, one plant or a portion of a plant (i.e., shoot tip, stem section, rhizome tip) can be aseptically propagated to produce thousands of new plants. Wetland and coastal plants are now being commercially propagated, and as commercial production increases, it is anticipated that the use of in vitro micropropagation to produce these plants will also increase.

Micropropagation of the submerged aquatic plant Widgeon grass (Ruppia maritima L.) for the production of seagrass planting units is a relatively recent achievement (Koch & Durako, 1991, Bird et al., 1994, Durako et al., 1994, Durako et al., this volume). Over the past six years, test plantings conducted by the Florida Department of Environmental Protection (FDEP) at nine sites in the Tampa Bay and Sarasota Bay areas have provided evidence of the potential for using micropropagated Ruppia plants in seagrass restoration. Donor plants from specific field sites thought to be compatible with prospective planting sites were used in micropropagating Ruppia for these test plantings.

Although it is not an inherently complicated process, the skills and capital outlay necessary to develop a fully functional micropropagation culture facility are not small. This paper presents the estimated costs of establishing a commercial seagrass micropropagation facility and addresses some of the unreported problems we have experienced in the day-to-day laboratory culture of Ruppia. This report is based on the experiences of research staff who have worked on developing laboratory and field techniques for successfully micropropagating and for maximizing the subsequent long-term survival of cultured seagrasses planted in trial restoration projects.

BASIC FACILITIES, STAFFING, AND ORGANIZATION

The basic facilities required include an area for washing glassware, an area for preparing the growing medium, sterilization equipment (autoclave and microwave), an aseptic manipulation/transfer area, incubators or culture rooms, and a facility for acclimating plants to natural environmental conditions. The organization and set-up of a tissue-culture facility has been addressed by several experienced plant culturists (e.g., Thorpe, 1981, and Vasil, 1985).

The main equipment needs and estimated costs are outlined in Table 1. We have not attempted to list every piece of equipment needed for day-to-day operations. Such information is available in reference material (e.g., Thorpe, 1981, and Vasil, 1985) regarding the set-up of a culture laboratory. The estimated costs provided here are based on our experience over the years in operating the FDEP tissue-culture facilities. A tissue-culture facility can conceivably be established on a relatively low budget. Using old jelly jars or baby food jars as culture vessels and a pressure cooker for sterilization of the jars could save many dollars; however, such a system would require many man-hours to produce large numbers of plants. Utilization of used and salvaged equipment can be cost-effective. As has been seen in many experimental undertakings by students and research scientists, ingenuity can produce great results with minimal capital outlay.

Table 1. Basic equipment requirements and estimated costs for seagrass tissue culture lab.

Analytical-grade chemicals	$ 1,600
Air conditioning with temperature and humidity controls (heat pump)	800
Aquariums with lights, photoperiod controls, and air supply (24)	4,000
Autoclave	20,000
Bunsen burner with gas source	400
Glassware	800
Laminar-flow hood (horizontal flow)	5,000
Microwave oven	200
Peristaltic pump	800
pH meter	700
Refrigerator/freezer	600
Stir plate	175
Top-loading balance	2,200
Vacuum pump	175
Wire storage racks with lights & timers	600 ea.

The staffing requirements of a culture facility will depend on the number of plants one desires to deliver in a 30-day period. In our seagrass-culture laboratory, one person working 20 hours per week produced an average of 500 rooted, field-ready plants in a month. That individual was responsible for carrying out the required laboratory procedures as well as washing their own dishes, maintaining aquaria for acclimation of plants, and maintaining the facilities; however, the individual did not keep up with inventory, order supplies, or perform any equipment maintenance. Although one person could handle all these responsibilities, production of plants decreases as the time spent on everyday operation requirements increases.

One full-time staff person in a research facility could increase production to the level of 2000 rooted seagrass plants per month, but the number of plants will vary depending on the type of equipment and facilities available and the administrative workload.

An experienced plant or animal-tissue culturist trained in biotechnology would be required for a one-person operation. After a successful micropropagation protocol has been developed, additional staff could include trained technicians who do not necessarily have advanced educations. Responsible, methodical, detail-oriented individuals would be needed to maintain the efficiency of a tissue-culture lab and to keep production at desired levels. Of course, the over-all cost of production rises as salaries and chemical and laboratory-supply requirements increase.

Cleaning laboratory glassware and plastics used in aseptic tissue culture procedures is more labor-intense than washing regular dishes is. Laboratory glassware is expensive. A 4-liter beaker is approximately $25.00, and a 50-ml graduated cylinder is about $20.00. Careful cleaning and handling -- not speed -- is the priority in maintaining laboratory glassware.

Maintenance of aquaria, vaults, and/or pools, which are where a majority of our plants are rooted and acclimated prior to planting, includes regular cleaning and water changes, control of algal blooms and other contaminants, and cleaning of air filters. The FDEP tissue-culture laboratory maintains twenty-four 75-liter aquaria with fluorescent light banks; much time and energy are required to keep these aquaria functional for Ruppia production.

A key component of keeping a tissue-culture facility running efficiently is keeping it clean. We strive to keep bacteria and fungal spores to a minimum by regularly cleaning our laboratory and culture rooms. Dust must be kept to an absolute minimum; fungus-eating dust mites are a potential contamination nightmare. They can crawl into culture vessels with fungi on their appendages and contaminate every culture in little time. Controlling dust is another time and labor-intense requirement of staff.

Efficient organization of time, data, and operations will go a long way in converting an experimental system into a viable commercial venture. Of special consideration in our work with Ruppia has been the role of "culture age" of a population in its totipotency. "Culture age" is the number of days the plant material has been maintained as sterile tissue and reflects the number of times a plant has been subdivided in culture. "Totipotency" refers to a plant's ability to propagate and respond favorably to culture conditions. Detailed records of all field-collected plants are maintained in our laboratories; the records outline, by date, every laboratory procedure performed, as well as observations on plant vigor and response to culture conditions. As the number of plants and the number of different populations being maintained in-house increases, the ability to distinguish between groups of plants can be lost. Accurate records are vital in overseeing operations, in troubleshooting problems, and in providing staff with needed information regarding the status of the plants. For example, if a population of plants begins growing very slowly after its last subdivision, accurate records could lead to an explanation (e.g., sugar or growth hormones might not have been added to the media when it was last prepared).

MATERIALS AND METHODS

Turnaround Time

In a perfect world, Ruppia wildstock would be collected from a donor site, brought into culture and micropropagated, and then rooted plants would be delivered to the field site for planting, all in as little as twelve weeks. The following is an ideal sequence:

Week 1, Day 1. Rhizome segments with short shoots (branch-like structures that diverge from the rhizome) are collected from a donor site and brought back to the laboratory. The segments are trimmed back to three to five nodes (the point where a short shoot grows out of the rhizome), rinsed in artificial seawater, and soaked in a fungicide solution overnight, beginning the sterilization protocol.

Week 1, Day 2. All work is done in a sterile area. We use a four-foot laminar-flow hood. The explants (the trimmed rhizome segments being sterilized) are rinsed in sterile, artificial seawater to remove most of the fungicide, are soaked for 10 minutes in a 10% solution of bleach and artificial seawater (under vacuum, 40 cm Hg), are soaked in an antioxidant solution for 30 minutes to minimize browning of the tissues, and finally are transferred to individual wells with 3 ml of antibiotic solution. While in the antibiotic soak, the explants are put under vacuum (40 cm Hg) for 30 minutes and are then placed on a shaker table overnight (24-hrs total time in antibiotic solution).

Week 1, Day 3. The explants are aseptically transferred to 12-well multiwell plates (each well containing 5 ml of nutrient-enriched seawater medium) and placed on growth racks for one week.

Week 2, Day 10. The explants with no visible contamination are transferred from multiwell plates to culture tubes (25 x 150 mm) containing 35 ml of nutrient-enriched seawater medium. The plants begin to grow rapidly. Four weeks later (in Week 6-Day 38), the culture tube is filled with a large plant (more than 100 nodes) ready to be subdivided into 8 to 12 new plants.

Subdivision of stock plants is the basis of micropropagation. For example, if each large, healthy plant was subdivided into 8 small plants, one of the 8 would remain in culture and be designated a "stock plant" (to be subdivided again in four weeks). The remaining 7 small plants could be designated as "field plants," placed in fresh medium, and allowed to grow for four more weeks. Thus, if you originally had 50 large plants at Day 38, you would now have 50 stock plants and 350 field plants growing in culture conditions.

By Week 10, the field plants are large, have more than 100 nodes (the typical size of plants in our planting units), and are healthy and robust. These field plants are then rinsed in distilled water to rid the plants of medium, and then they are placed in aquaria with seawater and brighter light to root and acclimate before being placed in the field. About two weeks later (in Week 12-Day 80), the planting units are ready to install in the field.

In six years of experience, we have found that this ideal sequence does not always occur. There can be delays and problems that come in many different forms.

Sterilization of the Plant Tissue

Because of the high level of nutrients and sucrose in the culture media, all bacteria and fungi must be eliminated from field-collected plants to be used for tissue culture purposes. We have a basic protocol in which three antibiotics, two fungicides, and a 10% bleach solution are used to sterilize the <u>Ruppia</u> explants. Our experience with 11 different populations of Ruppia from around the Tampa Bay and Sarasota Bay areas has shown that each population may require a slightly different combination of antibiotics to sterilize the plant tissue for culture purposes because of the variation in species of bacteria and fungi found in the different locations. In addition, because the species of bacteria and/or fungi present on Ruppia can vary with the seasons, the sterilization protocol for a particular population could require changes depending on what time of the year the plant is collected. If the standard antibiotic combination is ineffective, antibiotic-susceptibility tests are then required to define which antibiotics or fungicides are needed to sterilize plants from a particular population of Ruppia. Any changes in the sterilization protocol cause delays in the turnaround time of producing planting units originating from a particular donor site. Delays from two weeks to six weeks or more may occur in sterilizing plants from a particular population. Delay times can be shortened by having appropriate tissue-culture media and antibiotic-susceptibility test kits on hand, as well as a broad spectrum of antibiotics and fungicides that can be incorporated into the sterilization protocol.

Culture Vessels, Explant Size, and Media Volume

Vessels for culturing plant tissue come in many sizes and shapes. During the initial sterilization process, we prefer to use multiwell plates because they are easy to wash, can be sterilized in a microwave, require a small volume of solution per well, and require little space because they are small. We use a 24-well multiwell plate for the antibiotic soak, which requires $\leq$ 5 ml of solution per plant. For initial sterility testing, we prefer to use a 12-well multiwell plate in which $\leq$ 10 ml of medium per plant is needed to support growth. If bacteria or fungi have not been completely eliminated, any still present on or in the plant tissue will usually show up in the first week the explants are in media.

To support plant growth, we use culture tubes (25 x 150 mm) and plastic tubs (115 x 68 mm), each of which requires a different volume of medium (35 and 50 ml, respectively). We conducted experimental trials to determine the amount of growth in Ruppia over a four-week period in various volumes of medium in two types of culture vessels: culture tubes (25 x 150 mm) with 35 ml of medium and plastic tubs (115 x 68 mm) with 35, 50, 100, or 150 ml of medium. We determined that 35 ml was adequate for culture tubes and enabled us to grow a desirable-sized field-planting unit in the least amount of medium, in a space-saving vessel. However, in determining the optimal volume of medium for plastic tubs, which have a base diameter of 85 mm, we found that explants growing over a larger surface area with little liquid cover (35 ml) were more rapidly stressed. Fifty ml of medium in a plastic tub proved to be the most cost effective for the amount of growth seen in 50-ml versus that seen in 100- or 150-ml volumes.

For the production of Ruppia planting units, using plastic tubs with 50 ml of medium for stock plants and culture tubes with 35 ml of medium for field planting units proved to be the most efficient in terms of handling time and the number of dishes generated. Keeping stock plants in plastic tubs provides a sterile, wide-mouth vessel in which to directly subdivide the stock plant when it has grown sufficiently. The alternative would be to transfer the plant from a culture tube to a sterile petri plate, add sterile seawater to keep the tissues moist, cut and subdivide the stock plant, transfer the segments to fresh tubs and tubes of medium, then wash and sterilize the petri plate before another plant is placed in it. Subdividing in the vessel in which the stock plant has been growing saves time. In addition, a Ruppia plant growing in a tub has branches and rhizomes that are arranged in a wheel-like, spread-out fashion, making them easy to cut and manipulate into the smaller, desirable-sized new plants.

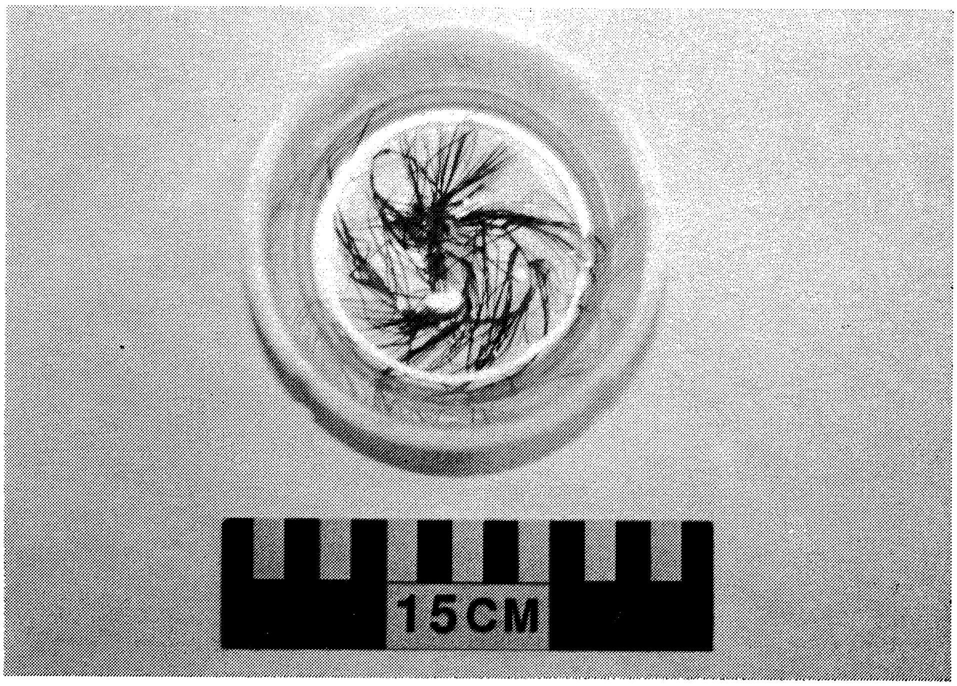

Figure 1. Cultured Ruppia stock plant in plastic tub--the wheel-like, spread-out growth of the plant facilitates subdivision in the container where the plant has been growing.

Storing the field planting units in culture tubes allows more plants to be stored in a small amount of space (40 tubes per rack). In addition, we have found that only 35 ml of nutrient-enriched medium is needed to grow a desired-sized planting unit in a culture tube. Culture tubes are also easily handled, cleaned, and sterilized.

Defining what is a "desirable size" for planting units of Ruppia may vary. During recent test plantings in which we used a newly developed cheesecloth-wire bag planting technique (Durako et al., this volume), we found that a planting unit with a surface area coverage of approximately 80 x 80 mm (or more than 100 nodes) is very successful in surviving and spreading. We have also had success with planting units of approximately 50 nodes. What seems to make the most difference in the survival and growth of plants is the number of branch tips present when the planting unit is initially placed in the field, because roots develop rapidly at the growing tips of branches and provide the initial anchoring of the planting unit.

The number of branch tips is influenced by the way in which the field planting unit is initially placed in a culture tube to grow out. The size of a subdivided explant should be more than 5 nodes but not more than 10 nodes. A larger initial explant quickly strips the medium of nutrients, becomes stressed, and is unfit as a planting unit after four weeks in culture. When working with large numbers of explants, uniform size is highly desirable, as is having plants ready to harvest on a controlled schedule. This can be done by regulating the initial plant size and the amount of medium supplied during the grow-out period.

A subdivided Ruppia plant with 5 nodes shows good growth in 35 ml of medium. However, one problem with such a small explant is that it may float to the surface of the medium when placed in the culture tube. Because Ruppia will only grow vertically (or upwards) in the culture tube, you end up with a small, short-bladed, tightly bunched planting unit if the explant is floating on the surface. We have to put two or three small units of this size together in a wire bag for one desirable-sized planting unit. This is very inefficient, considering the time and money invested in growing those two or three plants. With a slightly larger subdivided plant, preferably 7 - 10 nodes, the plant can be pushed to the bottom of the culture tube, and the branches or blades initially present will brace against the sides of the test tube and prevent the plant from rising to the surface. During the following four weeks, as the plant grows out, it occupies the entire volume of the medium, and the end product is a large, well-branched planting unit.

CULTURE AGE AS IT RELATES TO GROWTH AND ROOTING TIME

Growth Rate of Ruppia in Culture

As production of Ruppia increased, we observed that the growth rate variances in cultured plants depended on which wild population the plants came from. For instance, in comparing populations to one another and in comparing the number of explants we were able to obtain from stock plants from each population, it became apparent that some populations produced larger, more highly branched explants than other populations did. To determine this variance between the different populations' production of nodes and short shoots, we looked at the number of nodes produced by plants of each population over time and the relationship between that number and the length of time the plants from that population had been in culture (the number of days the plant material had been maintained as sterile tissue). Ten 5-node explants placed in 35 ml of medium in test tubes were monitored for four weeks, and the number of nodes produced over that time was recorded. This was done for various populations of explants of different culture ages being used in-house to produce planting unit propagules. We found that growth decreased over time (the older the culture age of a population was) despite the use of the same volume of medium; the same nutrients, sucrose, and growth regulators; and the same subdivision schedule (Figure 3).

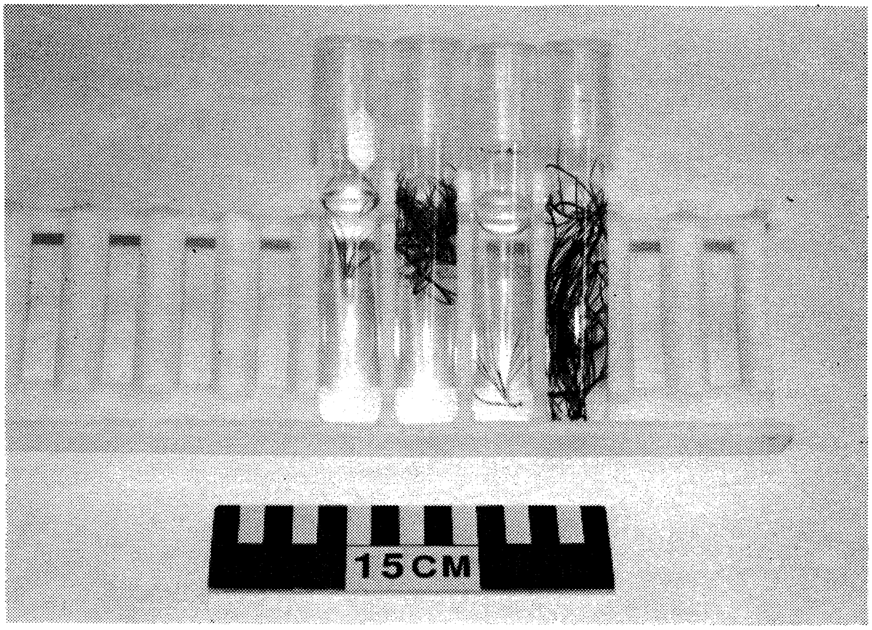

Figure 2. Where subdivided explants are positioned in the culture tube can determine the size of the explant after 4 weeks of grow-out time. A. A 5-node explant floating in top of culture tube. B. A floating explant after 4 weeks of growth; explant is short-bladed and has had minimal rhizome growth. C. A 7- to 10-node explant pushed to the bottom of the culture tube when initially placed in the tube. D. An explant that was pushed to bottom of the tube after 4 weeks of growth; explant is approximately 80 x 80 mm with large internodes in the rhizome, is multibranched, and has long blades.

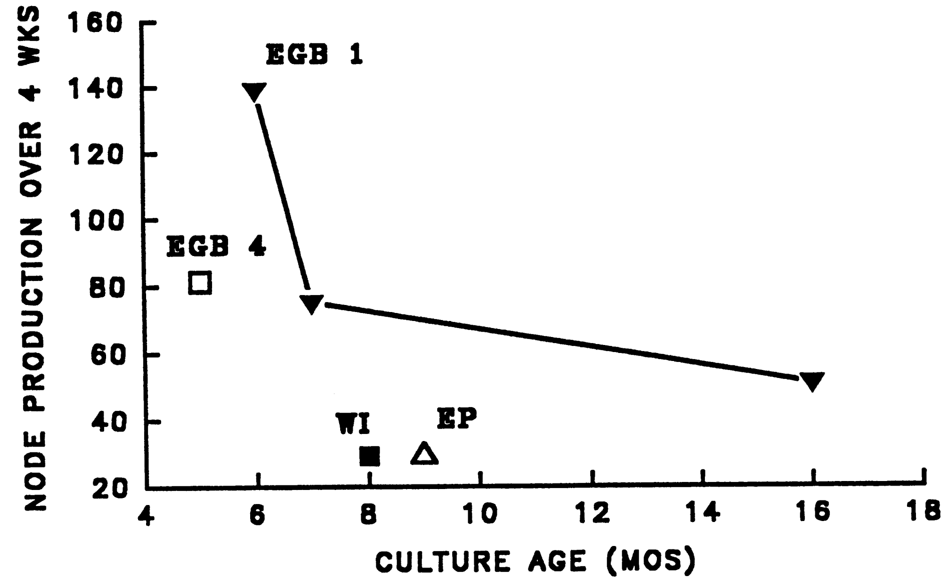

Figure 3. Effect of culture age on the growth of cultured Ruppia explants. (Ruppia populations (donor site): EGBI = East Gandy Bridge, clone #1; EGB4 = East Gandy Bridge, clone #4; WI = Weedon Island; EP = Emerson Point.)

Of special note is the high rate of growth seen in populations that have been in culture less than six months. One stock plant that is less than 6 months old can normally be subdivided into 12 or more explants. As the stock plants become older, production decreases. In the second six months of culture, the same population may produce only five new plants; therefore, more stock plants are needed to produce a desired number of planting units, and many more man-hours are required to aseptically manipulate the additional explants.

Also with regard to growth rates, our experience has been that explants having the higher rate of growth (those in culture less than 6 months) require more frequent medium changes and closer monitoring. New populations in culture appear nutrient stressed within 2.5 to 3 weeks. Once an explant is stressed due to lack of nutrients, at least two months of recovery time is required before it can be returned to stock plant production, or it is lost completely.

Another feature of culture age relates to the "desirable planting unit." We have observed that despite no change in medium protocol (amounts of nutrients, sucrose, or growth regulators), the amount of branching in explants decreases as the culture age of a population increases. Explants produced at a culture age of 5 months often develop more than 100 nodes and 10 branches (each branch producing an apical meristem), whereas culture plants of 10 or 12 months often produce only 2 or 3 branches. When an older plant is taken into the field, it is less likely to survive because it lacks a sufficient number of spreading branches to anchor the initial planting unit. Younger plants have greater numbers of branches to anchor the initial planting unit and in which to store the reserves the plant will need while it is acclimating to being transplanted as well as acclimating to other environmental conditions encountered at the field site (e.g., tidal flow).

Not every population responds to culture conditions in the same way. If a particular population fails to thrive initially, it is our experience that over time (more than six months) the ability of that population's stock plants to produce large numbers of new explants when subdivided declines, and they become a waste of time and expense and are discarded as unproductive. We have found it to be more efficient to introduce new populations to culture conditions more frequently and to work with young-culture-age populations that are growing vigorously and producing robust propagules.

Rooting Time

In planning our overall work schedule for test plantings, we needed to coordinate our field planting efforts with our tissue-culture laboratory plant production schedule in order to have planting units ready by specific dates. This required information regarding the amount of time necessary to root and acclimate the explants before they could be transferred to the field. We have found that there is a relationship between culture age and the rooting time of plants placed in acclimation aquaria. Plants were placed in aquaria and monitored, and the number of days required for rooting was recorded. This procedure was repeated for numerous populations of various culture ages. The results indicate that the longer a population had been in culture, the longer it took the plants to root (Figure 4). In

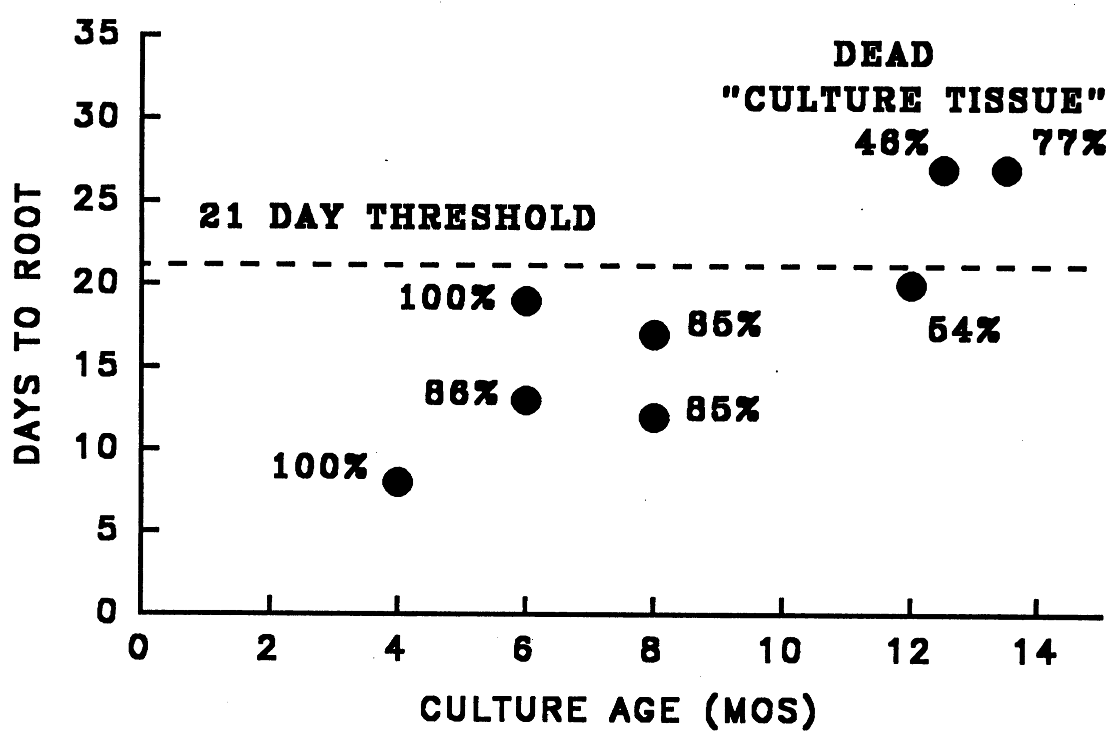

Figure 4. Effect of culture age on the rooting time of cultured Ruppia plants in acclimation aquaria; percentages given at data points represent the percentage of a cohort of plants in an aquarium that rooted in a given number of days. At the 21-day threshold, most of the cultured tissue of the plant is browned and only new growing tips of branches are viable.

addition, as we monitored these plants during their acclimation, we found that plants held in aquaria for 21 days or more turned almost all brown and were no longer "desirable planting units." Therefore, if the explants rooted at about 21 days or later, the size of these planting units were smaller and the tissues were weaker and more easily fragmented when handled than those of explants that rooted in less than 21 days. If a planting unit from an explant that rooted in 21 days or more is taken to the field, it will have much less biomass and fewer reserves for the plant to draw upon as it continues to root and establish itself at the field site than will a planting unit from a faster-rooting explant. There is also the probability that the plant will continue to fragment and lose more tissue before it is established in the sediments. As mentioned previously, in order to survive and thrive, a planting unit needs a robust plant that has more than 100 nodes and abundant branching, has the ability to support continued growth in those branches, and is able to anchor its roots readily.

CONCLUSIONS

It has been our hope to offer helpful suggestions for those micropropagating Ruppia maritima. Its suitability for field restorations has been demonstrated in selected sites, and it is easy to culture. By streamlining our Ruppia micropropagation protocol, we have been able to increase the number of planting units produced and improve their quality. Special attention to culture age of populations, as well as to the growth rate and rooting time of every population, has been especially beneficial in maximizing yields.

ACKNOWLEDGEMENTS

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"ON-SITE, OFF-SITE": THE EMPTY REFRAIN THE EVALUATION OF LOCATION AND FUNCTION FOR MITIGATION

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ABSTRACT

Although the importance of the location of compensatory mitigation is imbedded in the simplification of "on-site" and "off-site", this simplification allows no method for evaluating a particular function in the landscape. Each wetland provides a set of functions to its larger ecosystem, and the process of a proper mitigation plan is to evaluate these lost functions and replace them to the same landscape from which they were lost.

To a landscape, location could be a critical as the magnitude of a given function. As an example, the overall amount of silt-catching action of a wetland might not be as important as where in the watershed this occurs.

In this article, it is argued location is an inseparable part of the relation of a wetland function to the landscape. Hypothetical examples are used to illustrate the importance of disregarding the "on-site, off-site" refrain. In place of this refrain, the concept of a function shed is introduced, and a conceptual equation is presented to help in the consideration of location and function to the landscape in question.

INTRODUCTION

"Replacement is not a single matter of equivalency. It depends upon "context". It is sometimes said about real estate that value depends upon "location, location, and location". Similarly, the importance of wetland functions and values also depend upon "location, location, and location". John Kusler, 1992

The first goal of this discussion is to illustrate that the terms "on-site" and "off-site" provide no insight for evaluating mitigation projects. The second goal is to provide some suggestions for further analysis.

This discussion is built on previous work regarding wetland mitigation, specifically:

1. National Wetlands Assessment Symposium (Kusler ed., 1985).
2. Mitigation of Impacts and Losses (Kusler ed., 1986).

3. Wetland Restoration and Creation: the Status of the Science (Kusler and Kentula eds., 1990).
4. A Guide to Wetland Functional Design (Marble, 1992).
5. Effective Mitigation: Mitigation Banks and Joint Projects in the Context of Wetland Management Plans (Kusler ed., 1992).

DISCUSSION

Verbally applying the distinction "on-site" or "off-site" allows no method for defining, analyzing, or reestablishing mitigated wetland functions.

What is required is a richer analysis of the relation of function and its location. This analysis should begin by addressing the question of "what is a function?"

A function, regardless of its exact type, is inseparable from its location. This is because a function is providing activity to a specific landscape, and it is this activity in a specific landscape that we are concerned about for wetland mitigation.

As an example, it is not just the sheer volume of sediment removed or phosphorus retained by a wetland, but where in the landscape this removal occurs. To a lake, it is important if the wetland is adjacent to the lake, one mile up stream, ten miles up stream, or one mile down stream.

The distinction on-site or off-site fails to provide an adequate method for analysis because the operative word "site", at best, refers to the platted properties held by the owners that will destroy wetland functions. Although this forces some relative proximity of the destroyed and created functions, there is no way to discuss the wetland's relation to the rest of the landscape.

To the larger landscape and ecosystem, there is no meaningful distinction provided by "on-site".

Platted boundaries drawn by county land agents are mostly a matter of recent human history, and the project boundaries are solely a matter of economics; neither of which are relevant to wetland functions.

How could we evaluate the functions this wetland provides its landscape without being very specific about each spatial relation? To answer this, I would like to introduce the concept of a function shed.

FUNCTION SHED

A function shed is a planning region where a specific function can be relocated and continue to provide the same benefit to its original landscape.

Each function has its own function shed. As others have pointed out, using different terminology, some function sheds can be larger than others, depending on the function in question (Kusler, 1992). Figure 1 illustrates three different function sheds. An example of a large function shed would be waterfowl habitat. For

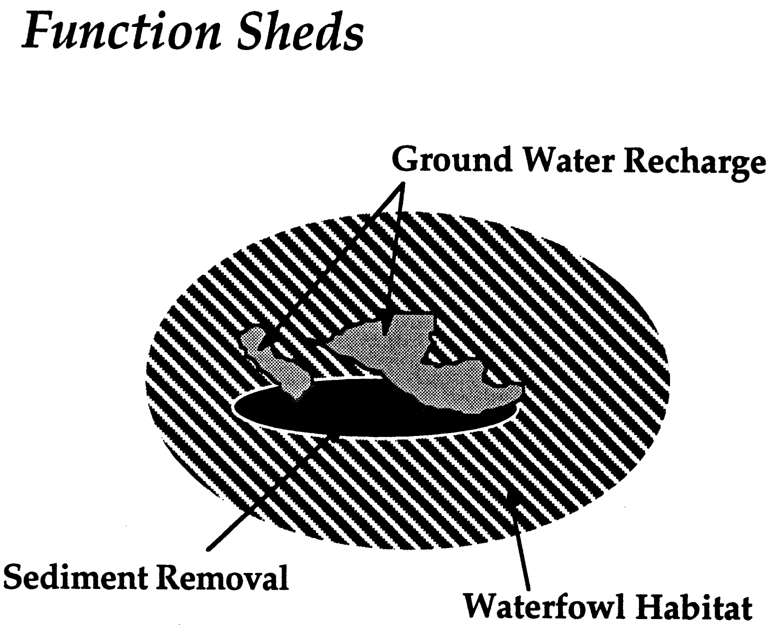

Figure 1. Function Sheds.

migrating waterfowl, the function shed could be ten square miles. An example of a small function shed would be where a wetland is removing sediment from a river before the water enters a lake. A function shed can be discontinuous. For example, in groundwater recharge there might be patches of regions for functional replacement.

In the initial planning phase of a mitigation project, a set of function sheds should be delineated. The boundaries of a function shed would be defined by the local geomorphology, hydrology, plant communities, and other aspects of the landscape. The accuracy and exactness of the boundary would also depend on the function in question.

The concept of a function shed is robust at different stages in the planning process. A course sketch of a function shed could help in initial feasibility studies. At the design stage, a more rigorous analysis would provide insight into design features, such as mitigation ratios.

USEFUL EQUATIONS

In order to quantify the relation of location to function, two distinctions must made by regarding the concept of function. The first is the measured amount of the function, such as the volume of sediment removed or the acre-foot of flood protection. The second is the function provided to a specific landscape.

To formalize these concepts and as a heuristic device, following is the equation:

Eq. 1. $F = f@\underline{l}$	F	= Mitigated function.
	f	= Measured function.
	$@\underline{l}$	= At a specific location.

The term $@\underline{l}$ can be either 1 or 0, depending if the location is within the function shed ($@\underline{l} = 1$), or outside the function shed ($@\underline{l} = 0$).

It is clear if the term $@\underline{l}$ is zero, then the amount of the measured function is irrelevant. The mitigated function should be zero. It is the contention if the restored function is not restored within the function shed, the function has not been replaced.

When the function shed is large, such as waterfowl habitat, then the above equation can be modified (eq. 2). This modification is designed to include the influence of location on the measured function.

Eq. 2. $F = f\underline{xl}$	F	= Mitigated function.
	f	= Measured function.
	$\underline{xl}$	= Location modifier.

The term $@\underline{l}$ can be replaced by the term $\underline{xl}$, where the term $\underline{xl}$ is a modifier of the measured function, depending on the location of the function. As an example, a wetland placed next to a shopping mall might have a $\underline{xl}$ value of .5 in regards to waterfowl habitat to indicate the location has a negative effect on the measured function.

MITIGATION, MITIGATION BANKING AND JOINT PROJECTS

In this framework, the mitigated function should be the currency of planning and law. Trade and economics should be applied to mitigated functions, not just the simple measured functions. The cost to replace a lost function should be the cost to replace the mitigated function.

A mitigation bank has one finite set of mitigated functions. This is the set of functions that the project was designed for, or in the case of post-success credit purchase, the functions that are currently present. With this analysis, a bank could "sell" these mitigated functions individually, or on a per-acre basis.

In the case of joint projects, there should be a delineation as to which mitigated functions would be applied or credited to which project partner. It could be the case that one partner might be involved only for one mitigated function that their own project does not provide.

SUMMARY

- A richer analysis of wetland function is required to replace lost wetland functions.
- Location is inseparable from its function.
- The inclusion of the location/landscape relation is necessary for analysis.
- A function shed is an effective method for consideration of mitigation.
- There is a critical distinction between a measured function and a mitigated function.
- The two equations (eq. 1 F = f@*l* and eq. 2 fx*l*) can help focus and guide analysis.
- Trade and economics should only be applied to "mitigated functions".

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A BIOASSAY APPROACH TO SEAGRASS RESTORATION

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ABSTRACT

A bioassay approach was employed to evaluate whether or not selected sites were suitable for seagrass habitat restoration. We installed small-scale plantings (25-110 units per plot) at eight test sites in the Tampa and Sarasota Bay areas using clonally micropropagated widgeon grass, Ruppia maritima L., and a standardized planting unit system. Measurements of transplant survival and planting-unit areal coverage rates were used as the bioassay system in determining site suitability. Survival ranged from almost 100% to 0%, and many plots showed significant patchiness in both survival and growth. Areal coverage rates of the surviving planting units were generally within the range of values reported for seagrasses in this geographic region. The total lack of transplant survival in several plantings at four of the eight sites suggests that these sites are currently unsuitable for seagrass restoration. Sites at which none of the transplants survived all had highly organic, muddy sediments and restricted tidal exchange. Using pilot plantings of robust, uniform plants of Ruppia provides a standardized, cost-effective method for assessing the suitability of sites in areas being considered for larger-scale seagrass restoration. This information could be critical to the formulation and implementation of management plans involving seagrass ecosystems.

INTRODUCTION

Seagrass communities have experienced significant declines in both area and quality in many coastal areas. For example, 80% of the seagrasses that have historically existed in Tampa Bay have beeen lost (Lewis et al., 1985). Much of this loss has been attributed to excessive nutrient loadings, especially in Hillsborough Bay, where virtually all seagrasses were lost between 1950 and 1984 (Johansson and Lewis, 1992). Reductions in nutrient loading over the past 10 years have contributed to improved water quality in Tampa Bay, and seagrasses have revegetated some shallow areas that had been barren for several decades. However, the lack of recruitment stock may limit the extent to which natural recolonization can occur. Several test plantings of seagrasses in Hillsborough Bay have been successful, and Johansson and Lewis (1992) have suggested that many areas may now be available for seagrass restoration by artificial means (i.e., transplanting).

The goal of this project was to evaluate, using a bioassay approach, whether or not eight selected sites would be suitable for seagrass habitat restoration. To test site suitability, we installed small-scale plantings (≈25-100 planting units) of clonally micropropagated widgeon grass, Ruppia maritima L., and then periodically measured transplant survival and growth. In the EPA's recent state-of-the-art assessment on wetlands creation and restoration, Fonseca (1990) suggested that installing and monitoring small-scale pilot plantings is a cost-effective approach to assessing site suitability. Ruppia has not been widely used in previous studies involving transplanted seagrasses. However, this species may be better suited than any other seagrass for use in the initial testing of site suitability in estuarine systems because it has the broadest physiological tolerances and the most cosmopolitan distribution of any seagrass. In addition, Ruppia is characteristically an early colonizer and frequently colonizes sites previously occupied by climax seagrass species.

MATERIALS AND METHODS

Micropropagation

Ruppia maritima was micropropagated using the protocol described by Koch and Durako (1991), as modified by Durako et al., (1993). The modifications included a vacuum treatment that decreased contamination rates by infiltrating the lacunar spaces of the explants (i.e., field-collected rhizome sections) with sterilizing agents.

Plant material, consisting of rhizome sections with attached short-shoots, was collected from five sites around Tampa and Sarasota bays (Table 1). After sterilization of field-collected material and 10 days of incubation in the sucrose-enriched growth medium (Koch and Durako basal salts [Sigma #K-1254], 1% [wt:vol] sucrose, and 10 mg/L 2iP (N^6-[2-Isopentenyl]adenine) at pH 5.6.), uncontaminated plants were aseptically transferred to sterile 25- x 150-mm culture tubes containing 35-40 ml of fresh growth media. We micropropagated the rapidly growing plants at approximately monthly intervals by subdividing them into 10-node segments (see DeLeon et al., this volume). The planting units installed at the selected sites were produced from this stock material.

One to two weeks prior to being planted in the field, transplant-sized plants (which had approximately 100 nodes) were transferred to aquaria containing synthetic seawater (Instant Ocean®) at 20 ppt so that they would "harden" (i.e., so that they would shift from heterotrophic to autotrophic growth) and begin to produce roots. Each group of plants used in a particular planting consisted of clonally produced plants from the same site of origin and were of the same culture age (see DeLeon et al., this volume); thus, they were essentially 'cohorts.'

Planting Units

The standardized planting units (PU's) used in this project were a modification of the novel, biodegradable cotton-mesh (cheesecloth) bag planting unit developed by Durako et al. (1993) except that the stone ballast was replaced by a U-shaped wire frame (14-gauge electric fence wire) that was sewn into three sides of the 8-cm x 10-cm cotton-mesh bag, with the legs of the U extending ≈10 cm beyond the open mouth of the bags (Figure 1). Individual, transplant-sized Ruppia plants were placed in the wire-framed bags in the aquarium culture room on the day of planting. The open end of each bag was folded over and stapled to secure the plant within the bag. The planting units were then placed in a cooler chest with layers of seawater-moistened paper towels and were transported to the field sites within one to three hours of assembly.

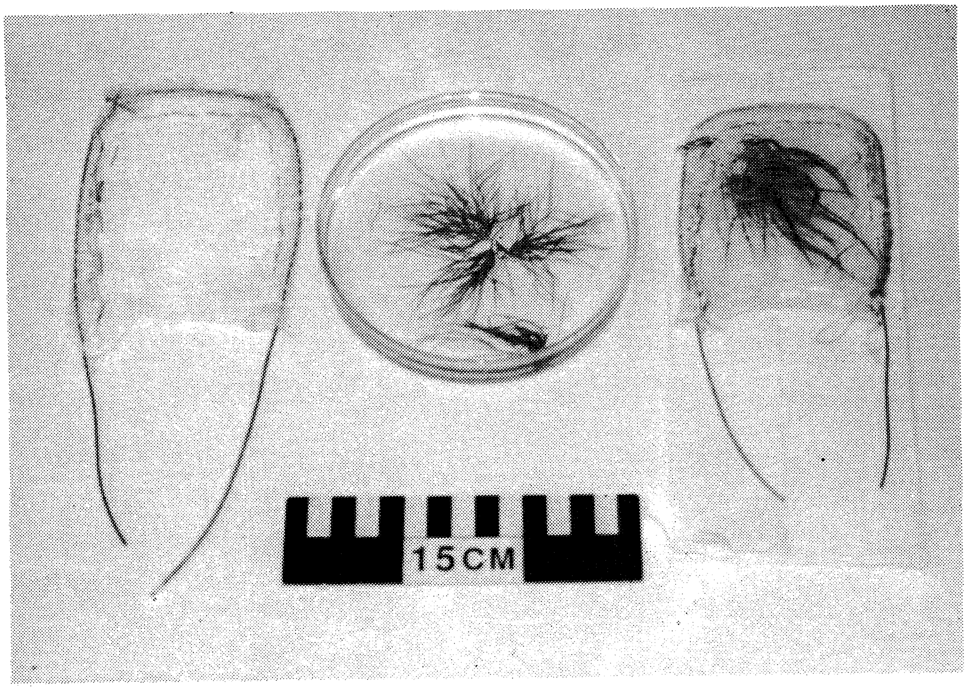

Figure 1. Ruppia maritima planting unit for bioassay tests.

Planting Unit Installation

Clonally micropropagated Ruppia maritima was planted at eight sites in the Tampa Bay and Sarasota Bay areas (Table 1). City Island (CTI) and Leffis Key (LFK) were restoration sites created by the Sarasota Bay National Estuary Program (SBNEP). Mangrove Bay (MNB), Little Bayou (LBU), Simmons Park (SMP), Boca Ciega Park (BCB), and Picnic Island (PCI) were restoration projects created by the Southwest Florida Water Management District Surface Water Improvement (SWIM) program. Hammock Park (DUN) was a City of Dunedin restoration project.

Test plantings were conducted throughout the year (Table 1); at Leffis Key plantings were conducted monthly to determine the effect of season on planting success. Planting units were installed on approximately 50-cm centers in

Table 1. Summary of planting dates, water temperatures and salinities at time of planting, plant sources, and numbers of planting units (Pus) for 1994 bioassay transplanting tests.

PLANTING DATE	SITE	TEMP(°C)	SAL(ppt)	SOURCE	#PU
01/10/94	LFK	18.5	34	LFK	48
01/10/94	LFK	18.5	34	LFK	49
01/10/94	CTI	13.8	34	LFK	25
01/20/94	MNB	15.5	14	LFK	60
01/25/94	SMP	21.1	25	EMP/WDI/LFK	100
02/21/94	LFK	23.6	35	WDI/EMP	77
03/02/94	SMP	19.9	26	WDI	75
03/09/94	LFK	23.0	35	LFK	85
03/31/94	DUN	23.0	28	WDI	41
04/14/94	LFK	23.6	34	EMP	65
04/06/94	BCB	25.2	35	EMP	49
04/06/94	MNB	23.6	28	EMP	60
05/02/94	MNB	29.6	15	LFK	50
05/03/94	DUN	31.8	33	EMP	32
05/04/94	LFK	-	-	EMP	19
05/04/94	LFK	-	-	EMP	80
05/04/94	CTI	34.6	34	EMP	50
05/10/94	SMP	27.1	30	WDI	45
05/25/94	LBU	26.1	29	EMP	80
06/03/94	MNB	-	-	WDI	40
06/06/94	LFK	-	-	LFK/WDI	100
06/10/94	MNB	-	-	WDI	63
06/13/94	DUN	30.3	33	LFK	62
06/27/94	MNB	32.4	28	EGB	80
06/28/94	SMP	38.0	28	EMP	100
07/05/94	DUN	33.7	18	WDI	15
07/06/94	LFK	-	-	EGB	100
08/26/94	LFK	32.0	34	LP	85
09/29/94	LFK	-	-	WDI	100
10/31/94	LFK	29.2	33	EGB	100
11/08/94	LFK	26.3	34	EGB/LFK	110
11/09/94	LBU	31.0	10	EGB	30
11/10/94	BCB	33.9	35	EGB	41
11/18/94	CTI	23.9	37	EGB/LFK	50
11/23/94	DUN	21.9	25	EGB	30
12/02/94	LFK	-	-	EGB	80
12/19/94	LFK	-	32	EGB	100

SITE KEY: CTI=City Island, DUN=Hammock Park, EGB=East Gandy Bridge, EMP=Emerson Point, LBU=Little Bayou, LFK=Leffis Key, LP=Lassing Park, MNB=Mangrove Bay, SMP=Simmons Park, WDI=Weedon Island.

rectangular-grid plots containing 25 - 100 PUs. The number of PU's per plot varied because of planting stock availability and site characteristics. During installation, the legs of the PU wire frame were bent down to form anchoring legs. The PU was then pressed into the sediments until the bag was flush with the sediment surface.

Survival of the PU's was monitored periodically. Areal coverage of surviving PU's was estimated by obtaining a diameter (d, in cm), calculated from the average width of the PU's along two perpendicular axes, and by computing the circular area (A_{PU}) as

$$A_{PU} = [\pi(d/2)^2].$$

Plot area was estimated by multiplying the mean PU area by the number of surviving PUs. The lengths of time during which records were kept for PU and plot areas varied because of the rapid coalescence of the PU's within some plots.

RESULTS AND DISCUSSION

It is somewhat ironic that the site (CTI) where we originally had such good survival and growth and upon which we based our bioassay strategy would, in this study, prove to be unsuitable to support Ruppia maritima. Our original plantings at CTI in the fall and winter of 1992/93 exhibited rapid growth and they even flowered within 4-6 months of planting (Durako et al., 1993). However, by the end of August 1993 all the original plantings had disappeared. We replanted this site 3 times, during the winter, spring, and fall (January, May, and November, Table 1) of the following year, and none of the subsequent plantings survived for more than a month. This change in site suitability at CTI seemed to be due to the growth of transplanted Spartina alterniflora across the mouth of the tidal creek that connects the artificial ponds to Sarasota Bay. These transplanted marsh grasses reduced flushing, so stagnant conditions developed within the ponds.

In contrast to the CTI site, the LFK site has several relatively deep tidal cuts. Thus, the connected ponds at this site have relatively good tidal flushing; in fact, in some areas at this site there is so much tidal current that plants could not be successfully anchored. In addition, the LFK site had only recently been created (< 1 yr old) when we began planting and the relatively new S. alterniflora and mangrove plantings had not coalesced. Figure 2 shows survival rate, PU spread rate, and rate of plot coverage by PUs of six plots planted at LFK from January to April of 1994. The top panel of Figure 2 illustrates that survival varied from near 0% to about 50%. The relatively low survival rate is partly due to the bioassay plot design wherein the plots extended across a depth range. What frequently occurred was that mortality was highest along the shallowest and deepest rows. However, even in plots with low survival, areal growth of surviving units was quite high (note that the PU and plot data have logarithmic Y scales). The rates of planting unit areal growth measured at LFK were generally within the range previously reported (10-70 $cm^2\ day^{-1}$; Fonseca et al., 1987) for vegetatively transplanted shoal grass, Halodule wrightii, in this geographic region. Planting units generally showed exponential areal growth and there was also generally a net increase of plant cover within the plots.

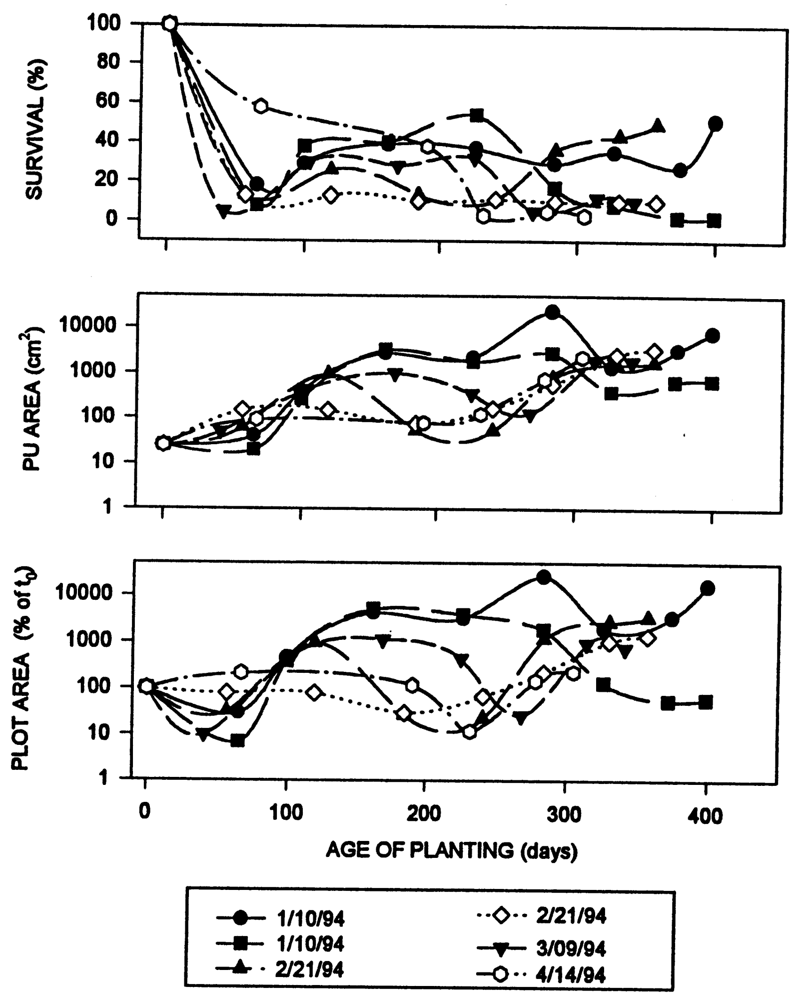

Figure 2. Percent of planting unit (PU) survival, PU areas, and total area of PUs in a plot (plot area) for winter and spring Ruppia maritima plantings at Leffis Key.

The ups and downs in Figure 2 reflect periodic coverage of the planting units by macroalgae, seasonal die-backs (especially during the summer), and some fragmentation of PUs.

One of the important lessons we learned from our monthly plantings at the LFK site was that planting season is an important consideration for R. maritima. Figure 3 shows the survival rate of monthly plantings at Leffis Key at about 100 days post-planting. What is most evident in this figure is the reduced survival of the June to August plantings. Summer was clearly not an appropriate time for transplanting this species, at least at this site. Survival and growth data from the summer plantings also illustrate this poor performance (Figure 4); none of the summer LFK plantings have survived.

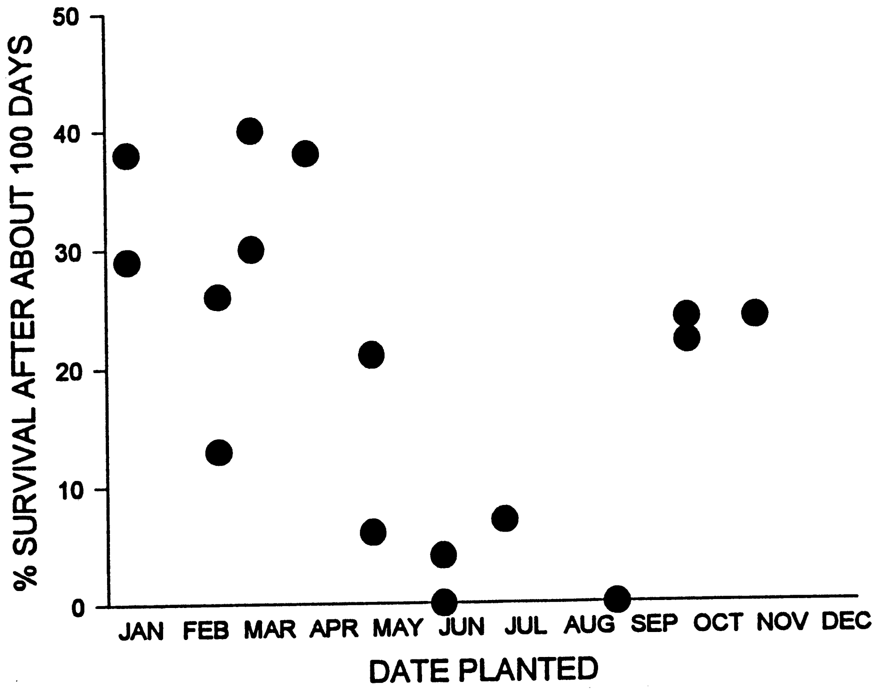

Figure 3. Percentage of Ruppia maritima planting units surviving at Leffis Key at about 100 days post-planting.

The Mangrove Bay (MNB) Restoration Site in Tampa Bay was a relatively mature site (about three years old) when we began our transplanting efforts. The pond where we conducted our test plantings had a well-developed S. alterniflora fringe, but a deep and wide channel connected the pond to Tampa Bay. Six test plantings were installed at this site (Table 1). Survival and areal growth were generally higher at this site than they were at Leffis Key (Figure 5). As with Leffis Key, the plot planted during the warmest season had the lowest growth rate. The shorter period of record for this site (210 vs 400 days) is due to the rapid growth and coalescence of the plots, which prevented continued monitoring of individual PU's.

Unlike the plants at Leffis Key, R.maritima at MNB never exhibited the dramatic die-back during the late summer. Ruppia at the MNB site frequently grew right into the fringing Spartina culms, suggesting that this zone may be a good area to install planting units. The PU's were periodically covered by macroalgae, which accounts for the dips in the survival data in Figure 5. Only plants we could see were recorded as survivors and measured. We did not remove the overlying drift algae because we found that this sometimes uprooted the underlying plants. The PU's under the algal mats did occasionally disappear, but frequently the plants survived and were measured during subsequent monitoring visits.

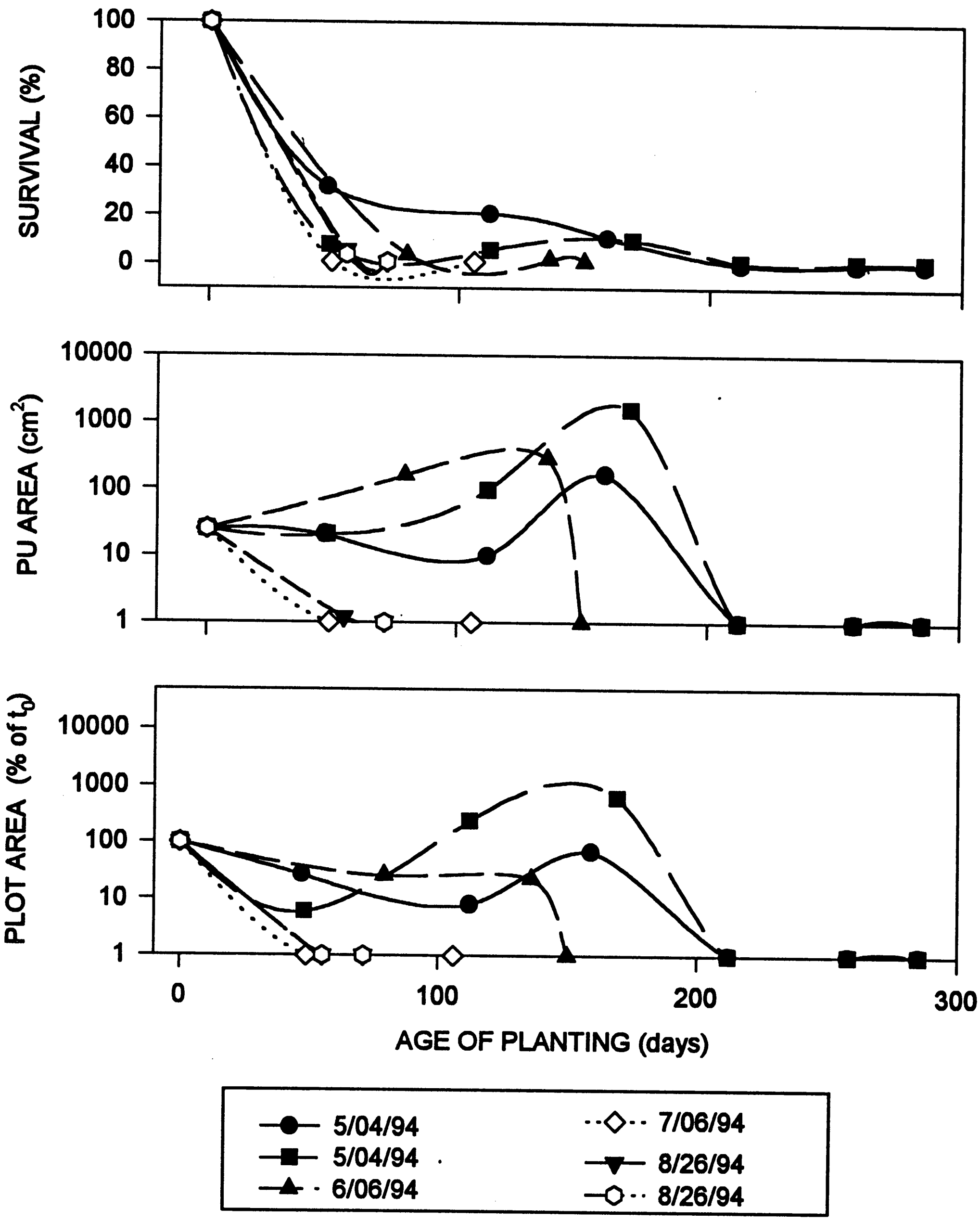

Figure 4. Percent of planting unit (PU) survival, PU areas, and total area of PUs in a plot (plot area) for summer Ruppia maritima plantings at Leffis Key.

The test sites at the SWIM restoration project at Simmons Park in northeast Tampa Bay included a cove and a tidal pond. The cove had a mature mangrove shoreline along the cove sides and a planted S. alterniflora fringe at the upper end. It also had sandy sediments and was open to tidal exchange both at its northern mouth and at a tidal creek at its southern, upper end. The tidal pond at SMP only had restricted tidal exchange through a single shallow channel. The pond also had

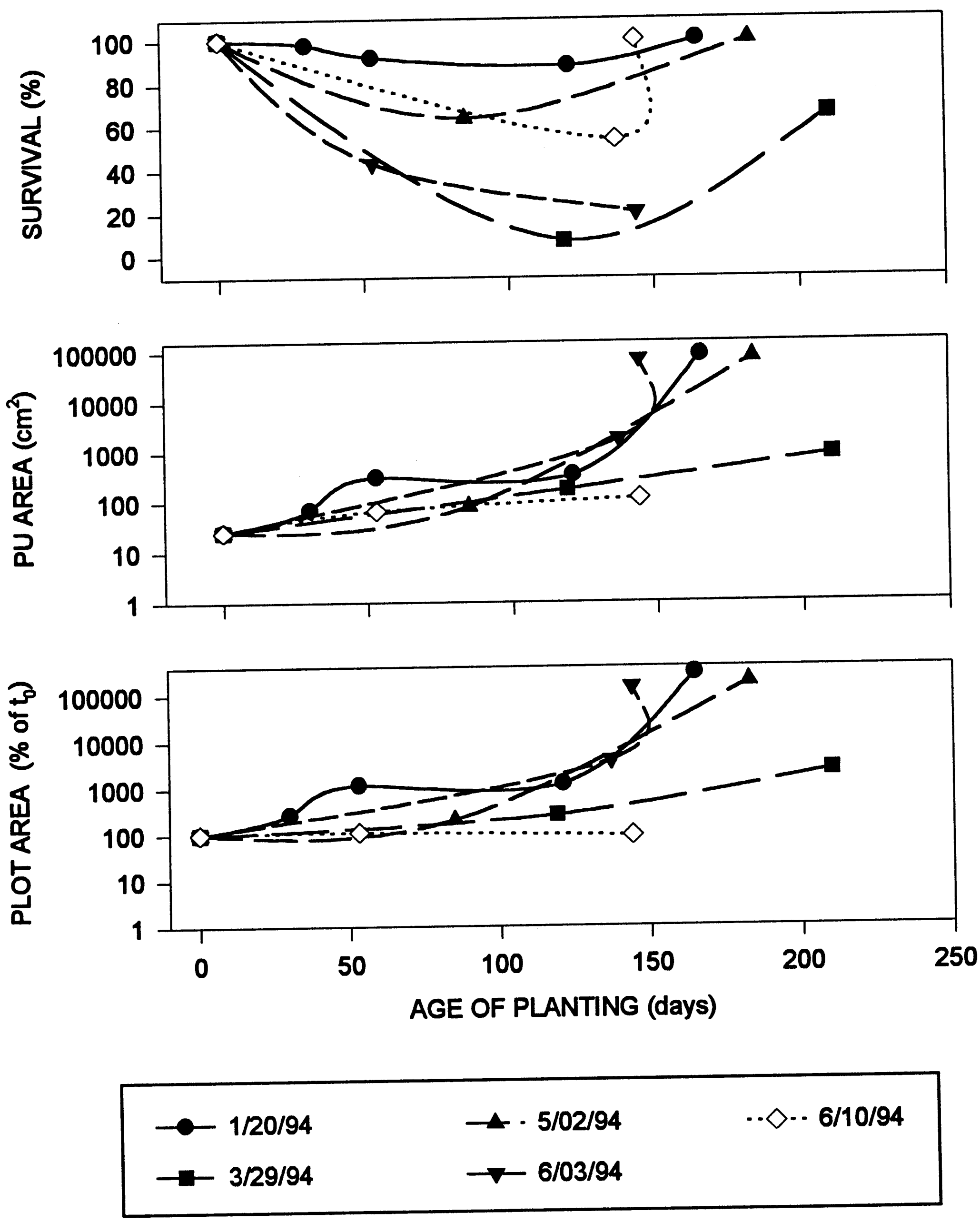

Figure 5. Percent of planting unit (PU) survival, PU areas, and total area of PU's in a plot (plot area) for Ruppia maritima plantings at Mangrove Bay.

muddy, organic-rich sediments. The two plots installed in the cove grew well and increased in aerial cover (Figure 6). The cove planting in May, which was placed at the upper, shallowest end of the cove had no survivors after three months. This planting was installed seaward of the S. alterniflora fringe, but because of the shallow slope of the cove this zone became exposed at extreme low tides (pers. obs.).

The single, summer planting in the tidal pond had low survival, although surviving PUs grew rapidly (Figure 6). The main cause of PU mortality or loss at the cove site seemed to be due to bioturbation by rays (Dasyatis and Rhinoptera spp.). We observed numerous ray blowouts, commonly within the test plots. This activity seemed to be more prevalent during the warmer months. In a report on previous transplanting efforts in Tampa Bay, Fonseca et al. (1994) suggested that bioturbation by rays is a major factor in planting unit loss.

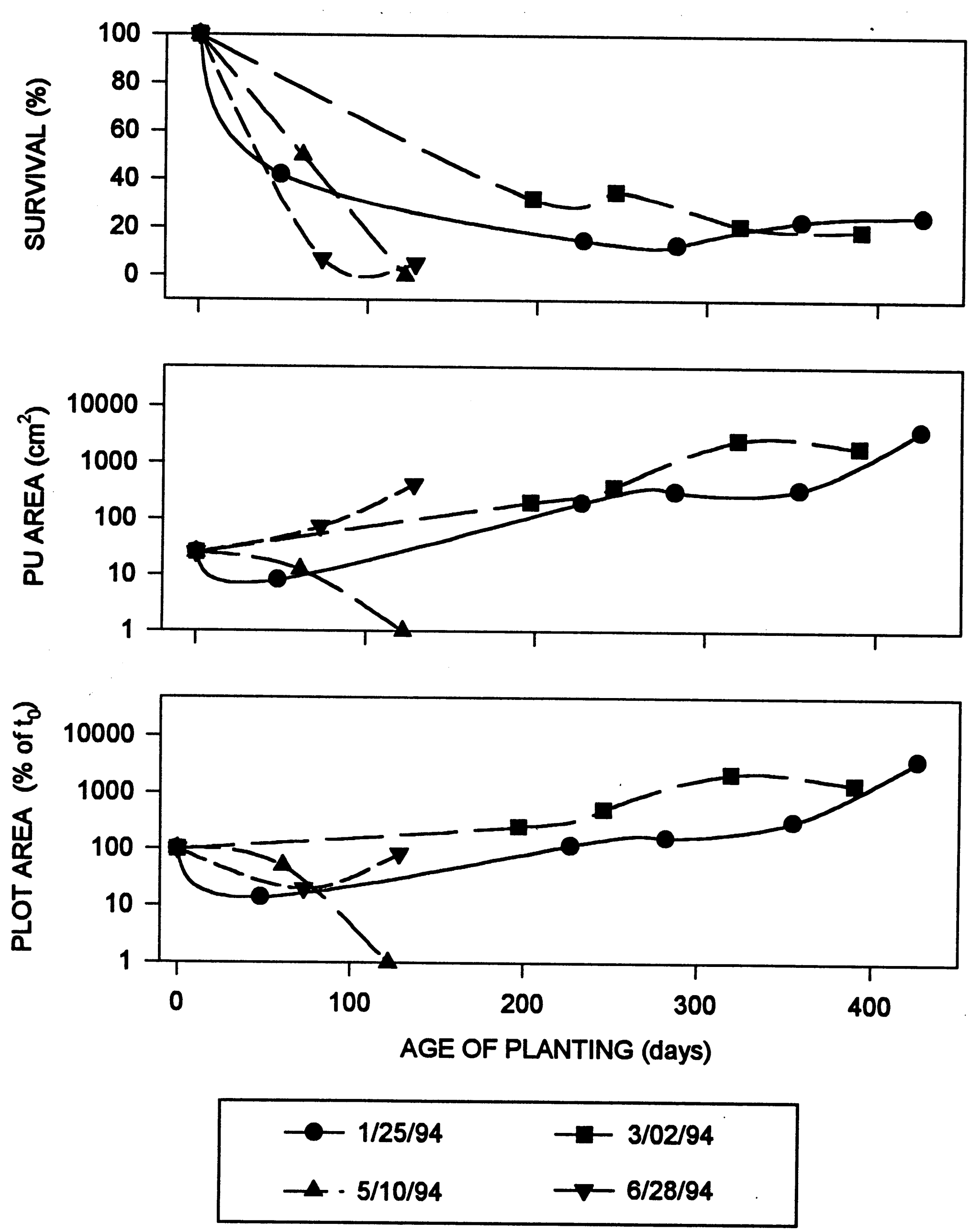

Figure 6. Percent of planting unit (PU) survival, PU areas, and total area of PU's in a plot (plot area) for Ruppia martima plantings at Simmons Park.

Plants installed in two adjacent tidal ponds at the Little Bayou SWIM restoration site in the spring and the fall (Table 1) all died within two weeks. Test plots installed during the same time periods in two similar types of ponds at the SWIM site at Boca Ciega Park had similar results. Both the LBU and BCB ponds had well-developed S. alterniflora fringes; highly organic sediments; single shallow tidal-creek connections; and were apparently flushed only at the highest tides. The S. alterniflora plantings at both sites had begun to grow across the shallow tidal creek mouths. The result was that within the ponds there was an accumulation of soft, highly organic mud. The totally blackened appearance of the wire frames of the PUs, after only one week in the field, indicated that the sediments were highly reduced, and this suggested that the plants may have been killed by high levels of hydrogen sulfide. In contrast, the Leffis Key, Mangrove Bay, and Simmons Park sites had sandy sediments with less organic matter.

The last bioassay test site was the City of Dunedin restoration site at Hammock Park (DUN). Several plantings, installed during the spring, summer, and fall (Table 1) in the central portion of a recently created tidal pond suffered complete mortality, probably for reasons similar to those at Little Bayou and Boca Ciega. Here, the source of the organic material was mulch that was spread around the perimeter slopes and was left unprotected for several weeks before the sod and landscape plantings were installed (Jon Everett, pers. comm.). In addition, tidal exchange with the pond was restricted by a littoral shelf and siltation barrier located under a catwalk across the tidal-creek connection.

CONCLUSIONS

The bioassay approach used in this study allowed us to evaluate what are and are not important considerations when assessing potential Ruppia maritima transplant sites. What doesn't seem to be a factor in transplant survival for this species is salinity (Table 1). City Island, LFK, and BCB salinities were high, frequently near seawater levels, while MNB and LBU had low salinities, as low as 10 parts per thousand. Simmons Park and DUN had intermediate salinities. The most important site factors affecting plant survival seemed to be sediment characteristics and tidal flushing -- and depth. Test sites with total failures all had highly low survival rates were associated with organic, muddy sediments and low tidal exchange because of narrow or shallow tidal cuts. The restoration site designers at the SWIM program are now aware of these problems that resulted from their earlier sit designs and they now incorporate wider and deeper tidal channels into their current restoration projects (Tom Reese, pers. comm.). At test sites where there are surviving planting units, mortality was generally highest in the deepest and shallowest rows, although at the relatively mature MNB site, high survival and growth of PUs were observed right at the S. alterniflora fringe. Time of year when transplanting is conducted is also important for R. maritima; the highest survival occurred in plantings made during the cooler months.

ACKNOWLEDGEMENTS

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RESTORATION OF FRESHWATER MARSHES FOLLOWING CERCLA REMEDIATION

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ABSTRACT

Two freshwater marshes were restored as functional ecosystems following a Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) site remediation in central Florida. The .8 ha (2-acre) and 3.2 ha (8-acre) wetlands had been subjected to clearing, dredging, filling, and hydroperiod perturbation prior to lead contamination of the organic peats and at the time of remediation were dominated by nuisance species, primarily cattail and primrose willow. Evaluation of remediation and mitigation alternatives showed that removal of contaminated peat was a feasible method to restore the wetlands. The Florida Institute of Phosphate Research Hydrology Model (FHM) surface and ground water model predicted post-remediation hydrologic conditions for revegetation. From 0 cm to 45 cm of substrate were excavated from 15 x 15 m grid cells, allowing detailed monitoring of post-remedial elevations and soil conditions. Following remediation, the wetlands were replanted with an assortment of species designed to respond to all probable hydrologic conditions. Factors unique to this type of non-conventional mitigation include the need to design mitigation plans to fit conditions, rather than fit conditions to anticipated plant specifications, and a need to continuously adapt to changes in on-going remediation activities on adjoining lands.

INTRODUCTION

From 1973 to 1986, the site operated as a battery recycling facility where sulfuric acid and lead were removed from discarded automotive batteries. Some empty battery casings were reduced to chips and transported off-site; some were buried on-site. In 1982, the site was listed on the United States Environmental Protection Agency (EPA) National Priorities List (NPL) and a Remedial Investigation was undertaken pursuant to Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) guidelines. Following a consent order in 1986 to effect remediation, a Feasibility Study (FS) and risk assessment were completed in 1988. EPA issued a Record of Decision (ROD) (USEPA, 1990) identifying alternatives from the FS for on-site disposal and treatment of lead-contaminated soils and waters.

The ROD's initially recommended alternative to remediate the two on-site wetlands specified fencing to limit access, coupled with increased inundation to minimize contaminant mobility by maintaining anoxic conditions in the substrate. It was recognized at the time that limitations existed with this approach. In particular, it only inhibited transport of the lead off-site, while leaving contaminated substrates and impaired ecosystem functioning in the wetlands. The flooding would further alter natural hydroperiod and functions. This approach offered the owner of the site no current or future relief from potential Natural Resource Damage Assessments, and may have required perpetual augmentation by groundwater pumping. In addition, the site was surrounded by residential development at elevations that made the development potentially susceptible to flooding induced by maintenance of higher water levels.

A Technical Memorandum was produced because of these concerns (Woodward-Clyde Consultants, 1992), in which the feasibility of maintaining constant wetland inundation was analyzed, and additional options for wetland remediation and mitigation were evaluated. This study determined that the inundation approach was not technically feasible due to difficulties in maintaining continual inundation of contaminants and an unacceptable risk of off-site flooding.

This paper reports on: the hydrologic analysis used to determine the post-remediation conditions; the factors used to evaluate alternatives for remediation (contaminant removal); and mitigation (restoration) of the wetlands, the techniques for remediation and restoration, and the current status of the post-remediation wetlands.

STUDY SITE

The Schuylkill Metals of Plant City, Inc. (SMPCI) site covers approximately 17 acres (6.8 ha) at 402 S. Woodrow Wilson Road in Plant City in Hillsborough County, Florida (Figure 1). The eastern half of the site contains the 10.2 acre (4 ha) East Marsh, while the 2.1 acre (.84 ha) West Marsh lies near the west boundary. The area between the marshes consists of filled land where battery recycling activities occurred. The southern portion of this central area is an excavated pond which was used as for stormwater retention.

The drainage basin of the SMPCI site was composed of three sub-basins. The first 14.6 ha sub-basin drained entirely into, and governed the hydroperiod of, the East Marsh. The largest sub-basin (26.4 ha) comprised the west part of the property and drained into the West Marsh. The remaining sub-basin contained the upland portions of the site between the two marshes and drained away from the marshes.

Prior to use as a recycling facility, numerous other perturbations had occurred on the site. Much of the land between the two marshes had been filled, with several feet of sandy soils atop the original organic substrate. Both wetlands had excavated ditches and areas in which dumping of household trash had occurred. The East Marsh also had been platted for residential lots and streets.

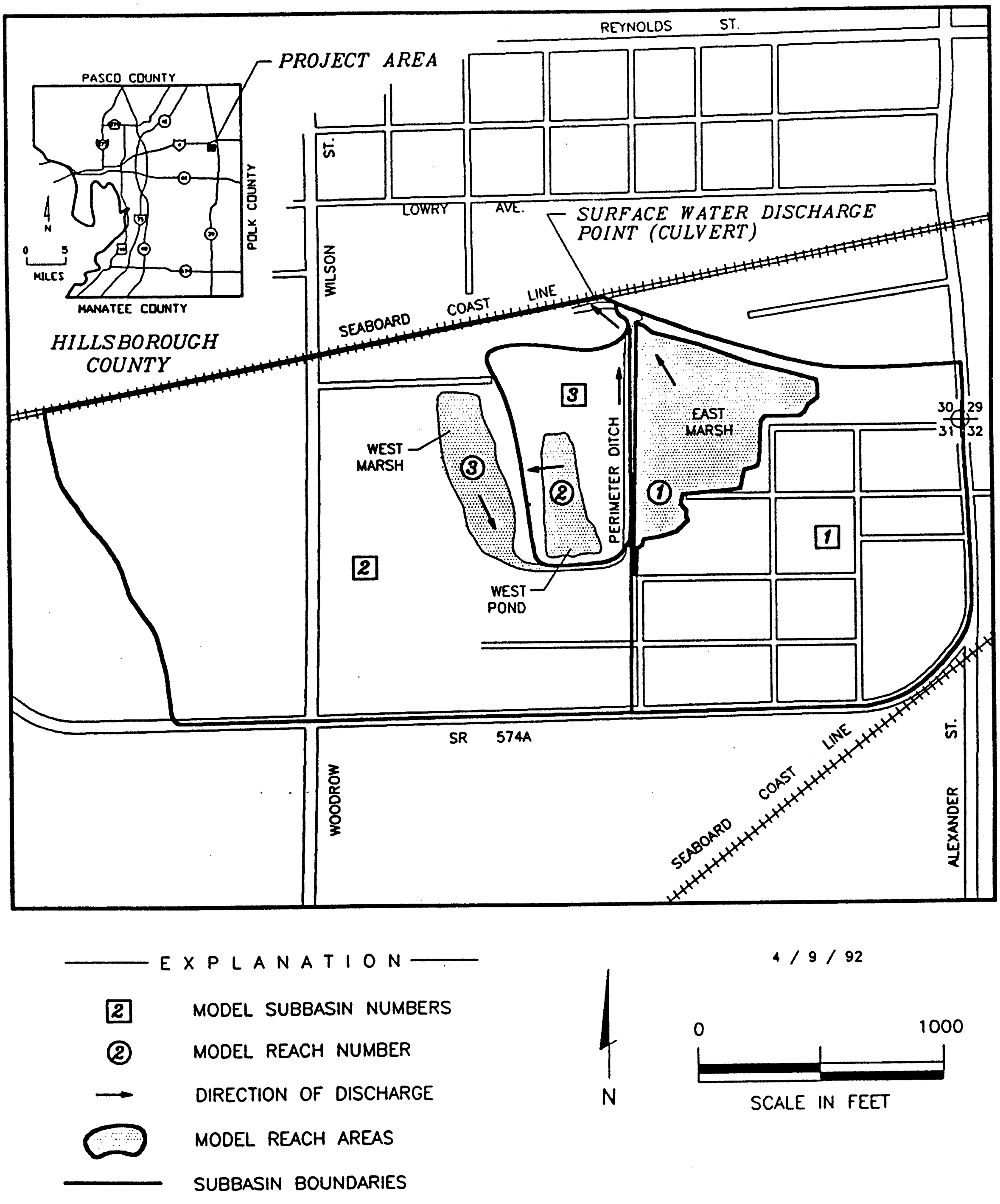

Figure 1.Site location showing marshes and drainage basins.

At the time of remediation, the West Marsh contained 20 vascular plant species, but was dominated by nuisance species, with primrose willow (Ludwigia peruviana) and common cattail (Typha latifolia) most prevalent. Primrose willow comprised 94% of the Importance Value Index for shrubs. Other abundant species were smartweeds (Polygonum hydropiperoides, P. densifolium), spikerush (Eleocharis ascicularis),and lizard's tail (Saururus cernuus).

East Marsh vegetation consisted of several associations, including a Carolina willow (Salix caroliniana) forested zone, a shrub zone of primrose willow/elderberry (Sambucus canadensis), and a deep zone dominated by annual smartweeds and sedges (Cyperus spp.). Much of the East Marsh was upland-transitional with barnyard grass (Echinochloa crus-galli), dog fennel (Eupatorium capillifolium), Caesarweed (Urena lobata), switchgrass (Panicum virgatum), giant foxtail (Setaria magna), balloon vine (Cardiospermum halicacabum), bladderpod (Sesbania vesicaria), and wild taro (Colocasia escalenta). Of the 42 species in the East Marsh, 8 were nuisance or exotic species, and another 11 could be considered upland or invasive species.

MATERIALS AND METHODS

Feasibility and Design Studies

Substrate and Elevation Sampling

For pre-remediation characterization, the site was divided into a 15.2 x 15.2 m grid pattern. The corners of each grid cell and the center point elevation of each grid cell were surveyed. The horizontal and vertical distribution of lead in the wetland substrate were determined in substrate samples taken from the center of each grid, resulting in sample spacing of 15.2 m (50 ft). Separate samples were taken from three depths (0-15 cm, 15-30 cm, 30-45 cm), using a bucket auger. The same bore hole was sampled using separate decontaminated augers and stainless steel sampling equipment to obtain each discrete sample. Samples were sent to Savannah Laboratories in Tampa for lead analysis.

Following remediation, each wetland grid cell unit was sampled for substrate lead concentration, utilizing a composite sample consisting of subsamples from each of the four quadrants of the cell. Post-remediation sampling was done only for the surficial 15 cm, since earlier samples confirmed a trend of decreasing lead concentration with increasing depth. Confirmation of clean conditions in the surface layer therefore indicated clean conditions throughout the substrate profile.

A target clean-up average of 100 mg/kg for lead in wetlands substrates was negotiated with appropriate agencies, with no single sample to exceed 125 mg/kg. A Shapiro-Wilk W-test was used to determine lognormality of the data set (Gilbert, 1987), and the 95th percentile upper confidence limit (UCL) was then determined for the lognormal data (Gilbert, 1987), such that if the UCL of the sample population was below 100 mg/kg, then the average concentration of the entire wetland met the cleanup standard with a 95% confidence limit.

Hydrologic Assessment

Pre-remediation Seasonal High Water (SHW) prior to remediation was estimated at several stations in the marshes based on several factors such as lichen lines and high water marks on trees.

The FIPR (Florida Institute of Phosphate Research) Hydrology Model (FHM) was used to model the hydrology of the wetlands and to predict the hydroperiod of the marshes following remediation, based on anticipated changes in drainage (SDI, 1992). The FHM integrates the HSPF surface water model and the MODFLOW groundwater flow model to predict water levels as a function of both surface and groundwater conditions.

FHM was run using several scenarios and assumptions for both the pre and post-remediation conditions. Among the options considered were three scenarios of groundwater hydraulic conductivity ranging from 1.3 m/day to 5 m/day to evaluate the effects of various rates of recharge to groundwater. The model also evaluated the effect of combining drainage basins by routing all surface water flow through the East Marsh before exiting the site.

The model parameter found to most affect hydroperiod was rainfall. Three cases of rainfall were evaluated - an average year, a "typical" below average year, and a "typical" above average year. To evaluate these effects, actual data, recorded at Tampa International Airport, from "typical" years selected from the past 30 year period were utilized.

Wetland Remediation

Both wetlands were remediated by excavating organic peaty substrates contaminated with lead. After lead levels were determined for each grid cell area, a map was prepared showing the necessary depth of excavation for each cell. Excavation was accomplished with two long-reach (15.1 m) tracked backhoes working from wooden mats. Substrate was loaded into articulated, low surface pressure dump trucks and removed from the wetland. Ditches were cleaned with the backhoes or by a tracked bulldozer moving up the ditch.

Prior to excavation, wetlands were sprayed twice with Rodeo herbicide to kill existing vegetation and cleared for excavation. Large shrubs and trees were cut and removed manually. Excavation began on April 18, 1994 and was completed with a final inspection on June 30, 1994. Natural drying of the marshes was supplemented with portable gasoline powered pumps, discharging to the on-site stormwater retention pond, with no off-site discharge. The heavy equipment moved across the wetland substrate on 4.5 x 4.5 m oak mats, which were laid in position and removed by the backhoes as they moved across the wetlands.

Cells were excavated to depths of 15 cm, 30 cm, or 45 cm, depending on the vertical lead distribution. The accuracy of excavation for each cell was checked with a surveyors rod along the edges of the cell. After excavation, each cell was sampled for lead content. If surficial (<15 cm) lead content was above 125 mg/kg, the cell was re-excavated an additional 15 cm and resampled.

Upon completion of substrate removal, the center-point elevations of each cell were surveyed and a post-remediation topographic map of the wetlands was developed.

Restoration Planning and Planting

Post-remediation elevations of cells were compared to the estimated Seasonal High Water (SHW) and Normal Pool (NP) or Seasonal Low Water (SLW). The hydroperiod or number of days of expected inundation was estimated, as well as the range of water depths expected. Cells were then broken into four groups based on expected hydroperiod and water depths. Groups were defined on the basis of 30 cm ranges of SHW elevations. Thus SHW water depth was expected to range from 0 cm to about 120 cm depending on grid cell elevation.

Wetland restoration following remediation entailed planting selected plant species. The aims of revegetation were to increase perennial species cover, reduce seasonal plant cover fluctuation, and reduce the extent of seasonally bare ground available for colonization by nuisance species. Perennial species are also better suited for long-term survival in the slightly greater water depths and longer hydroperiods resulting from excavation.

For each elevational range group, a minimum of three plant species was selected for planting. Most selected species were capable of reproduction by rhizomes or runners. An average 75 cm distance between plant centers was used for a mean planting density of approximately 17,000 plants/ha (7,000 plants/ac). All plants were bare root. Planting was done manually by Nautilus Environmental Systems personnel, using nursery stock and material transplanted from a Florida Department of Environmental Protection approved donor site in the Tampa area. Much of the material was planted in the largely dormant season (November to February).

Although excavation was essentially completed by May 30, 1994, seasonal high waters and a need to obtain EPA approval of results prior to restoration delayed most planting until November. Initial planting of the West Marsh and planting of approximately 1/3 of the East Marsh was accomplished by December 15, 1994. Above average late season rainfall maintained unseasonable high water levels throughout fall and winter. Supplemental plantings of deeper parts of the West Marsh and remainder of the East Marsh planting were completed between February and May, 1995.

RESULTS

Wetland Remediation

Approximately 1,166 m^3 of organic substrate were excavated from the West

Marsh and 4,950 m^3 from the East Marsh. In the West Marsh 18 grid cells were excavated to 15 cm, 5 grid cells were excavated 30 cm, and 2 grid cells were excavated to 45 cm. Fourteen cells in the West Marsh required no excavation. Within the East marsh, a total of 86 cells were excavated 15 cm, 28 excavated 30 cm, and 4 excavated 45 cm. Eighteen East Marsh cells required no excavation. Only 4 cells required additional excavation to meet criteria after the first sampling indicated that goals had not been met.

All remediation goals were met for the wetlands, with no single cell having a concentration greater than 125 mg/kg. The average lead concentration in the peat substrate was reduced from about 305 mg/kg to 40 mg/kg in the East Marsh and from 375 mg/kg to 42 mg/kg in the West Marsh (Figure 2).

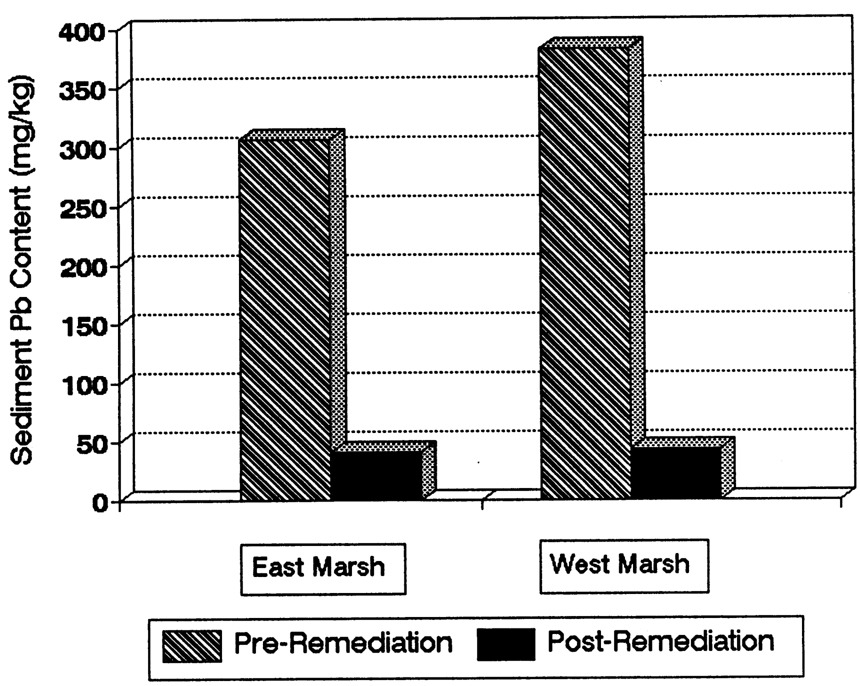

Figure 2. Lead content (mg/kg) in peat substrates before and after remedial excavation.

Hydrologic Regime

Remedial excavation reduced the average elevation of both wetlands by approximately 15 cm. The main effect (Figure 3) was to move the preponderance of cells from the 112.9-113.8 feet NGVD elevation range to the 111.9-112.8 feet NGVD range. The percentage of area below 111.8 feet increased from 3% to 15% of the total wetland.

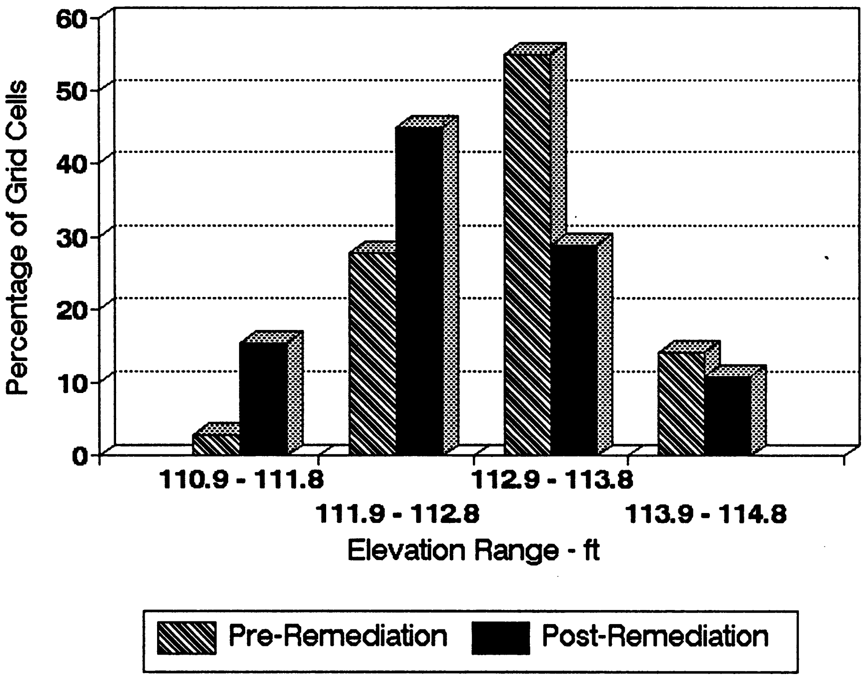

Figure 3. Grid cell class elevation frequency distribution before and after excavation. Data shown is for both marshes combined.

The FHM Modeling results indicated that construction of a control structure at the discharge point of the East Marsh would substantially increase and stabilize water levels in the marshes. However, the increase would not be sufficient to maintain permanent inundation over as much as 50% of the contaminated area if no supplemental excavation occurred.

Figure 4 shows predicted water levels based on a control structure invert elevation of 114.5 feet and an average rainfall year. Figure 5 shows predicted results for a low rainfall year in which almost all of the East Marsh would have been left exposed.

Thus the model indicated that construction of a control structure to maintain a pooled condition would not maintain the degree of permanent inundation necessary to immobilize substrate lead. Additional problems were identified by the modeling. At the 114.5 feet invert level, the 25-year storm event would have exceeded ground surface in several adjoining residential yards. Also, the examples shown in Figures 4 and 5 were based on a one year transient condition. Running the model for subsequent yearly periods indicated that under a long-term quasi-steady-state condition, dry season water levels would be even lower.

A final invert level of 113.7 feet will be used for the East Marsh outfall when completed. This level is expected to be sufficient to maintain fluctuating hydroperiod conditions with water levels ranging from 112.0 to 114.3 feet over the course of an average year. With about 75% of the wetland area between 112 and 114 feet, this is expected to provide inundation levels of 0 to 60 cm over most of the wetland.

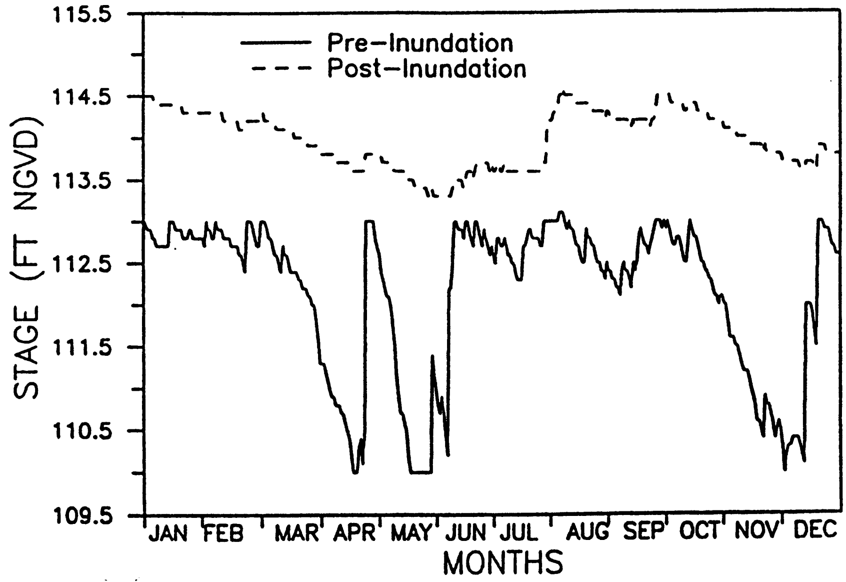

Figure 4. Annual stage height water levels for East Marsh for the pre-inundation existing condition (solid line) and the post-inundation predicted condition (broken Line) based on a new invert level of 114.5 feet. FHM simulations based on average rainfall and hydraulic conductivity scenario.

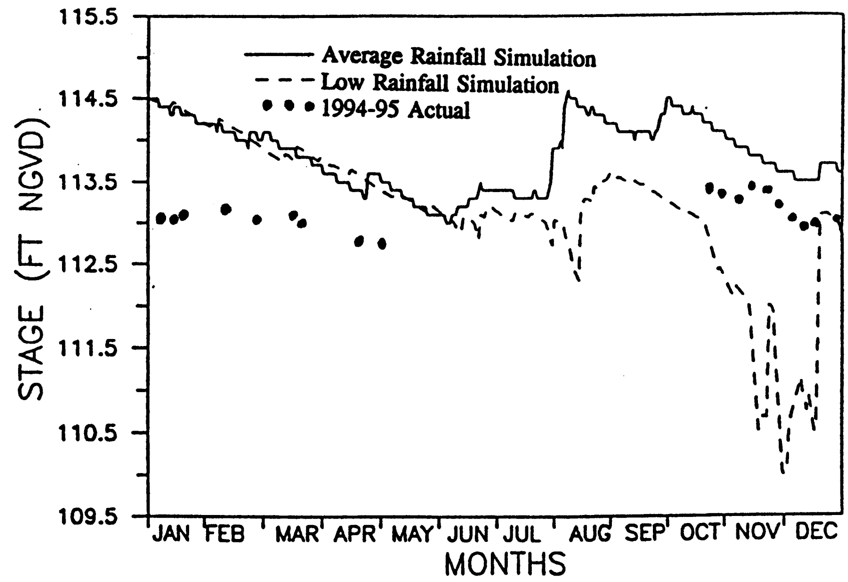

Figure 5. FHM simulated post-inundation annual stage height water levels of East Marsh based on average rainfall (solid line) and low rainfall (broken line) conditions. Circles indicate actual water levels for 1994-95 with 113.7 feet invert level.

Revegetation

Qualitative observations since planting indicate that plant cover rapidly is being reestablished throughout both marshes. When planted in October 1994, cover in the West Marsh was less than 1%. Estimates on February 16, 1995 showed average cover of about 25%, while in late May cover was nearly 80% over the entire marsh, and nearly 100% in all areas except for the ditch and the 3 lowest cells. By April, many planted species had flowered, including softstem bulrush, soft rush, pickerelweed, arrowroot, and flat sedge.

Parts of the West Marsh above 113 feet dried out sufficiently to allow widespread seed germination in the exposed substrate, especially in unexcavated areas where annual species from the pre-existing seedbank, especially the smartweeds, are dominant.

In the East Marsh, plantings made in November, 1994 have had mixed success. Survival and growth rate has generally been equal to the West Marsh in similar elevational zones. However, plants in the highest transitional zones of the East Marsh (sand cordgrass, maidencane, softrush, and blue-flag iris) have had poor survival. Annual grasses have continued to dominate this zone.

High autumn rainfall precluded planting most of the East Marsh until April 1995, but planting was completed by May 20. Post-planting survival of spring plants has been greater than that for the autumn and winter plantings. Maidencane and spikerush had poor survival and growth in winter, but both responded well in the spring planting.

Wildlife Utilization

Bird counts from 16 post-remediation periods between October 1994 and May 1995 have documented an average of 12.0 wading birds/waterfowl and 6.8 other birds per hectare in the East Marsh. Over the same period, average bird utilization of the West Marsh has been 15.5 wading birds/waterfowl and 7.5 other birds per hectare.

Birds have included wading bird and shore bird species typical of freshwater marshes as well as coastal habitats. Among species observed are great blue heron, little blue heron, Louisiana heron, black-crowned night heron, common egret, snowy egret, wood stork, white ibis, anhinga, roseate spoonbill, black-necked stilt, common gallinule, black skimmer, and coot. Other common birds include red-wing blackbirds, common grackles, and boat-tail grackles.

Wading birds have been feeding on fish and invertebrates which survived in ditches and were abundant over the unvegetated flats during the winter months. A large number of insect species, including bees, dragonflies, skimmers, grasshoppers, and water bugs have been found in both wetlands, as well as marsh rabbits, raccoons, and several species of snakes, turtles, frogs, and lizards.

DISCUSSION AND CONCLUSIONS

Remedial Versus Restoration Objectives

This project illustrates the need to coordinate restoration goals with remediation goals. Because this site fell under CERCLA, it was subject to damage assessment by the federal and state governments under the Natural Resource Damage Assessment (NRDA) regulations of CERCLA Section 301c and the Florida Natural Resources Damages Liability Statute. Wetland inundation to immobilize lead was selected to address human health criteria only. It did nothing to relieve the responsible party of long-term NRDA liability for damage to regulated natural resources (water, wetlands, endangered species). By continuing to damage the wetland functions by continued contamination and inundation, that alternative did not close the NRDA liability issue.

In addition to continuing NRDA liability, the responsible party was still liable for mitigation of wetland losses since the existing wetland was no longer functional. The consent order required mitigation for wetland areas left contaminated. Since none of the upland portion of the site was available, this condition would have required off-site mitigation. Available off-site mitigation opportunities were few, other than restoration of wooded riverine systems or construction of wetlands on purchased uplands.

Cost analysis of off-site mitigation alternatives indicated a need for at least 30 additional acres to be restored or created under conditions less conducive than in the on-site wetlands. Remediation/restoration of 10 acres of on-site wetlands was found to be comparable or lower in cost to off-site alternatives and technically easier to accomplish due to existing hydrology and soils. In addition, on-site remediation and restoration relieved future NRDA liabilities due to continued contamination of the site. The on-site restoration approach resulted in substantial future benefits to the client without substantial additional cost.

These benefits were not identified in the alternatives analysis based solely on traditional remediation approaches; the initially selected course of action may have resulted in serious future liability problems. For projects such as CERCLA remediation, therefore, it is important that all regulatory aspects be identified and evaluated. Assessment of NRDA regulations and ecological risk evaluation should be included as a parallel step in planning remedial actions.

Hydrologic Regime

The FHM model had great value in predicting post-remediation conditions and evaluating different options for remediation and restoration. Although it is probably impossible to model all scenarios so that a direct comparison can be made with actual conditions in the period immediately following remediation, a qualitative comparison of predictive results to actual conditions indicates that the model had a high degree of accuracy in predicting actual conditions. The degree of resolution appears to be adequate for use in evaluation and design of restoration projects, particularly where changes in basin size and condition and interaction of groundwater and surface water are important.

The wetlands restored in this project have relatively large variations in water level. This is probably due to two factors. The first is that much of this partly urbanized basin has had increased impervious surfaces and increased runoff due to stormwater drainage ditches. The second is that much of the original storage capacity of the wetland may have been removed due to filling of parts of the wetland prior to SMPCI ownership of the site.

As a result of hydroperiod variation, careful selection of plant species and a mixture of species of differing hydroperiod tolerances was required. For similar projects with changing water depths or a high degree of hydroperiod uncertainty, we recommend that a minimum of 3 species be used in each zone and that the species have sufficient difference in tolerance to allow survival of at least one species in any potential resulting hydroperiod. This approach may result in an low overall survival rate. For this reason, planting densities should be increased to compensate for low survival. Although initial cost may be higher, this will result in greater assurance of meeting cover criteria in the minimum time and will decrease future costs of maintenance, monitoring, and replanting.

Nuisance Species and Weed Control

Primrose willow, wild taro, and cattails were expected to be major problems following remediation. Cattail germination in previously dominated areas of the West Marsh was extensive in March and April, but a control program of pulling seedlings and spraying appears to have controlled this species. Little re-invasion has been noted and levels are well below 1% of total cover. Primrose willow seedlings appeared in the summer and are a potential problem in the East Marsh. Due to the extensive desirable species cover, manual control may be the only feasible option. Alligator weed and water hyacinth were almost absent from the wetlands prior to remediation. Water hyacinths have since appeared in small numbers but have been readily controlled by constant attention, removal and spraying Observations indicate that proliferation of water hyacinth can become uncontrolled if left unattended for more than 6 weeks.

Surprisingly, alligator weed was the greatest control problem. Control was not applied until about 4 weeks after the species was first noted in June 1994, and small patches spread throughout the East Marsh. Rapid spring growth infested many vegetated parts of the wetland. This weed was difficult to control because of its tendency to root at the nodes. After spraying, the plants die back to a node, but the stems break apart before the herbicide can move to the vascular system beyond the node, and are driven by wind across the water surface, rooting when water recedes. This species must be controlled before desired vegetation is established, and it must be monitored at intervals as short as weekly to maintain control.

Annual grasses and weeds like giant foxtail, cogan grass, and barnyard grass out-competed planted species primarily where no excavation occurred. Germination occurred when ground surface was exposed; rapid growth occurred before the cells reflooded in the next wet season. Successful perennial species establishment where annuals are abundant requires proper timing and chemical control prior to planting. Planted vegetation appears to successfully compete when planted before the ground dries. Water levels of 4 to 12 cm appear to be optimal for growth of planted rootstock and are sufficient to inhibit annual seed germination. If rootstock can be planted at least one month prior to substrate drying, desirable species can compete satisfactorily.

The greatest impedance to restoration of this wetland has been control of nuisance species and annual weedy/grass species. We had tried to reduce substrate removal to a minimum; however, the greatest weed problems were encountered in areas where we did no removal. Whether this was solely because the buried seeds were not removed from these areas has not yet been determined. It may also be a function of higher elevation and more rapid substrate drying, allowing earlier germination.

A recent approach to wetland creation has been use of wetlands soils as a mulch to promote germination of buried viable seeds. While this approach is useful where an undisturbed wetland is removed and then replaced (as in mining), it can lead to problems when applied to disturbed area restoration. Total removal of surface soils in disturbed wetlands may reduce nuisance species competition to desirable species, while increasing functionality. Cost savings in reduced maintenance and monitoring may compensate for increased excavation costs.

In any event, the importance of maintenance and weed control can not be over-emphasized. This needs to be carried out intensively at short intervals, as little as 2 weeks, from the time that substrates are exposed to air until desirable plant cover is around 80%. This large initial effort will pay large dividends in shortening time to completion and reducing costly future maintenance and monitoring, and is effort and money well spent. This project has demonstrated that successful functional restoration of contaminated and/or degraded wetlands can be accomplished in a short period if the differences between restoration and creation are recognized and addressed in planning and operation.

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EVALUATION OF NATURAL RECRUITMENT OF WETLAND TREES AND SHRUBS ALONG A NORTH FLORIDA PIPELINE CORRIDOR

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ABSTRACT

Various regulatory agencies have required pipeline rights-of-way to be planted to reestablish forested wetland communities following construction. Such requirements have been made without the benefit of data regarding natural regeneration. This study was conducted to determine if natural regeneration was a viable alternative to replanting linear corridors following construction activities. A natural gas pipeline in north Florida was selected for the study. This pipeline is approximately 86 miles (approximately 140 km) long, crossed 57 wetlands and was completed in May, 1991. Three growing seasons (1992, 1993 and 1994) have occurred since completion of construction activities. Data were collected from 12 wetlands and focused on tree and shrub densities. Data indicate forested wetlands are naturally returning to the corridor. Tree densities are significantly higher than those typically used in replanting efforts while shrub densities are comparable to accepted planting levels. Data collected in this study suggest natural regeneration is a viable alternative to time consuming, expensive and logistically impractical replanting of long narrow linear corridors in general, and particularly in areas with similar physiographic and climatic conditions to north Florida. In other areas, natural regeneration should be considered on a case-by-case basis.

INTRODUCTION

Recent permits issued by the federal and state regulatory agencies for pipeline construction have required that the right-of-way be planted with native wetland vegetation. Alternative methods of achieving the primary objective of revegetation of these rights-of-way, particularly forested systems, are being presented by the permittees and considered by the agencies. The potential for natural recruitment of native wetland species from adjacent wetlands has been identified as a desirable alternative. However, no data on natural recruitment under similar conditions appears to be readily available to support this approach.

A major natural gas pipeline company completed construction of a pipeline in north Florida about 3 years ago (May 1991), offering a unique opportunity to study the effectiveness of natural recruitment in revegetating a pipeline right-of-way. Typical pipeline construction practices were used to install the pipeline and included open cut trenching across wetlands and waterbodies. Following completion of the pipeline, the trench was backfilled and the right-of-way "cleaned up" establishing a "smooth" topography at the preconstruction elevations according to permit conditions and agency guidelines. The final right-of-way was stabilized using grass seed mixtures and allowed to naturally revegetate with native forest, shrub and herbaceous species.

The company is permitted to maintain a 30-foot-wide corridor centered over their pipeline in a non-forested condition. The remaining 20-foot-wide portion of the construction right-of-way, adjacent to the undisturbed wetland community, is to be allowed to naturally revegetate to previous wetland community. The company has performed one maintenance event (during 1993) within the 30-foot-wide maintenance corridor in the three years since pipeline completion. Maintenance consisted mainly of manual removal of large woody vegetation. The company has not mowed within wetland areas.

This study was conducted on December 19-20, 1994 to assess the current condition of natural revegetation of wetland forested community types within the 20-foot-wide temporary construction corridor. The study focused on tree and shrub regeneration. Data were collected to determine the density of trees and shrubs naturally returning to the corridor and the densities of trees within three age classes; one year (those sprouting during 1994), two years (the 1993 growing season), and three years (the 1992 growing season).

By examining these data, prognosis for natural regeneration and reestablishment of former native wetland forested communities along a disturbed corridor should be possible. These data will be useful in evaluating the potential for natural regeneration along future pipeline or linear project facilities, particularly in areas similar to northern and panhandle Florida.

Dames & Moore was designated as the lead consultant for this effort and provided study design, field data collection, and data collection/field management services. The report was also prepared by Dames & Moore. Dr. G. Ronnie Best and students from the University of Florida's Center for Wetlands and Water Resources assisted in the study.

STUDY SITE

The pipeline was built from Suwannee County (near the Suwannee River) to a point just west of Jacksonville, Florida in Nassau County. (See Figure 1.) The pipeline is approximately 86 miles in length and affected 57 wetlands, most of which were forested community types. Twelve wetlands were selected (based on their community types) to provide a sample set of the major community types affected by this pipeline.

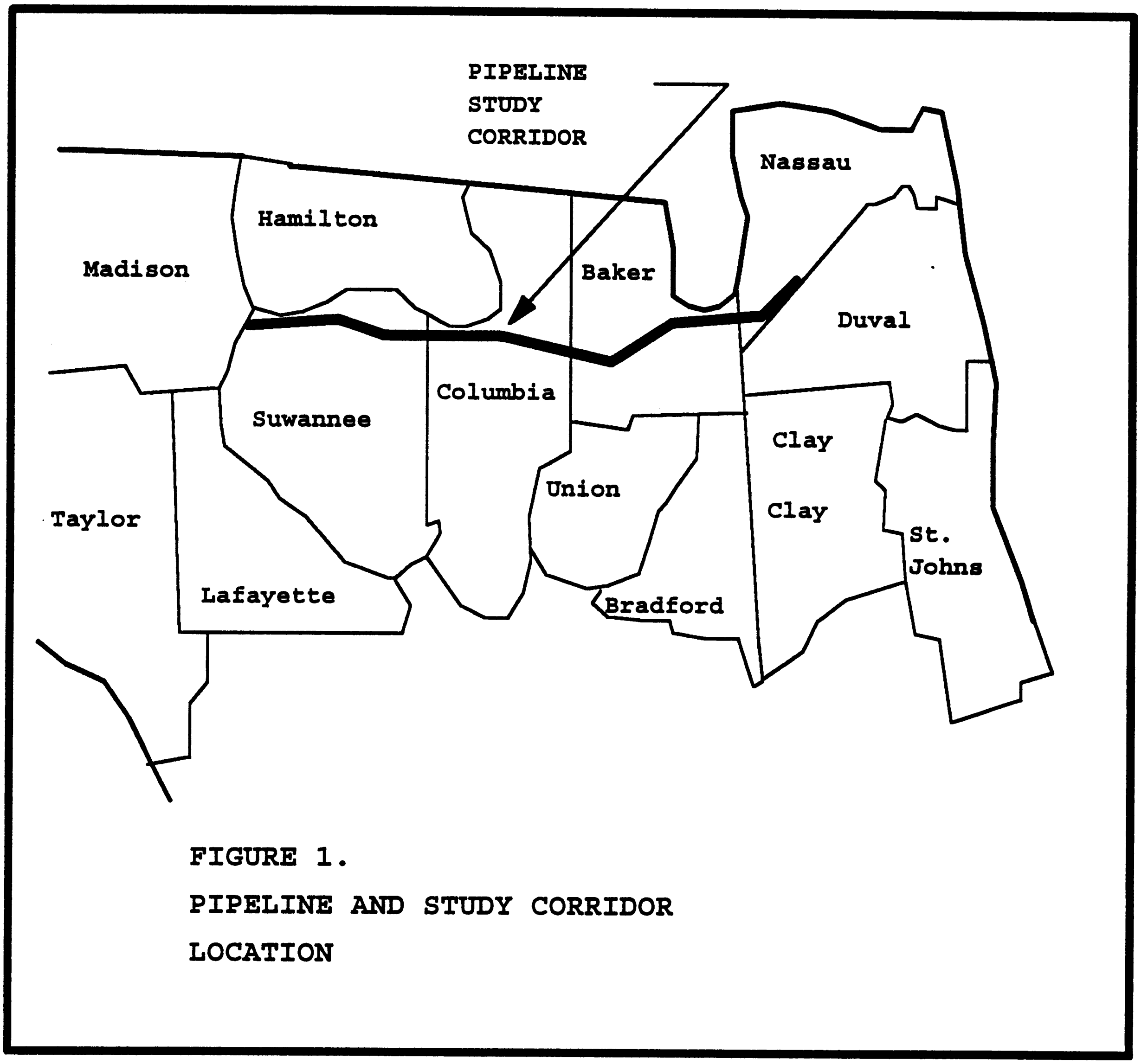

Figure 1. Pipeline and Study Corridor Location

MATERIALS AND METHODS

Dames & Moore served as consultant to the pipeline company during permitting and construction of the pipeline used in this study. Based on this experience, Dames & Moore identified five (5) major and several minor wetland community types found along the corridor. Each of the 57 wetlands along the 86 mile pipeline were categorized into one of these community types. Five major wetland community types, based on tree species composition and hydrology, were selected for this study:

- Red Maple Swamp type; dominated by red maple (Acer rubrum), but may have various other species as associate canopy components. Hydroperiods are typically seasonal.

- Deep Swamp; a community with prolonged inundation typically dominated by cypress (Taxodium spp.) and/or blackgum (Nyssa biflora).

- Mixed Hardwood Floodplain/Seepage Slope; having a gentle slope from upland to stream with seepage evident. Dominants include: red maple, sweetgum (Liquidambar styraciflua), oaks (Quercus spp.), sweet bay (Magnolia virginiana) and pines (Pinus spp.).

- Mixed Hardwood Swamp; dominated by a mixture of typical hardwoods including red maple, sweetgum, oaks, blackgum, and cypress. Inundation is generally from one to three months duration.

- Mixed Hardwood Floodplain; typical floodplain containing a moderate to large stream. Topography and species composition variable with dominants including; oaks, red maple, sweetgum, cypress, pines, blackgum and various associates such as ironwood (Carpinus caroliniana), river birch (Betula nigra) and some non-wetland species such as American holly (Ilex americana). Hydroperiods correspond to seasonal rainfall, fluctuating, inundation from one to two months.

Three wetlands from each of the major community types (or all wetlands if there were less than three affected) were selected for study resulting in twelve wetlands selected.

Quantitative sampling was conducted in plots along a single linear transect established three (3) meters from the undisturbed wetland edge within the 20-foot-wide temporary construction corridor. The total transect distance was measured between permanent wetland signs erected by the pipeline company just landward of the wetland edge. If the wetland sign had been placed significantly landward or waterward of the jurisdictional wetland edge, the length of the transect was adjusted accordingly and noted on the field data sheets. Each transect paralleled the undisturbed wetland edge and the pipeline across the length of the entire wetland.

Ten (10) 2.5 by 2 m plots were established equidistantly along each transect. The beginning of the first and the end of the last plot were placed at a distance of 5 meters from the wetland edge. (See Figure 2.) If distinct zones of vegetation occurred within the wetland and plots were not established within these zones when evenly spaced, plots were moved so that zones were represented and uneven spacing was noted on the data sheets.

Within each plot the number of tree and shrub seedlings was recorded by species (wetland and non-wetland). Trees were recorded within the following height classes: 6", 6-12", 12-24", and 24". Trees 24" were measured and actual height was recorded. Additionally, tree species' ages; one, two, and three year classes, were estimated. These data were utilized for seedling/sapling density estimations. Only wetland species were used in calculations, however.

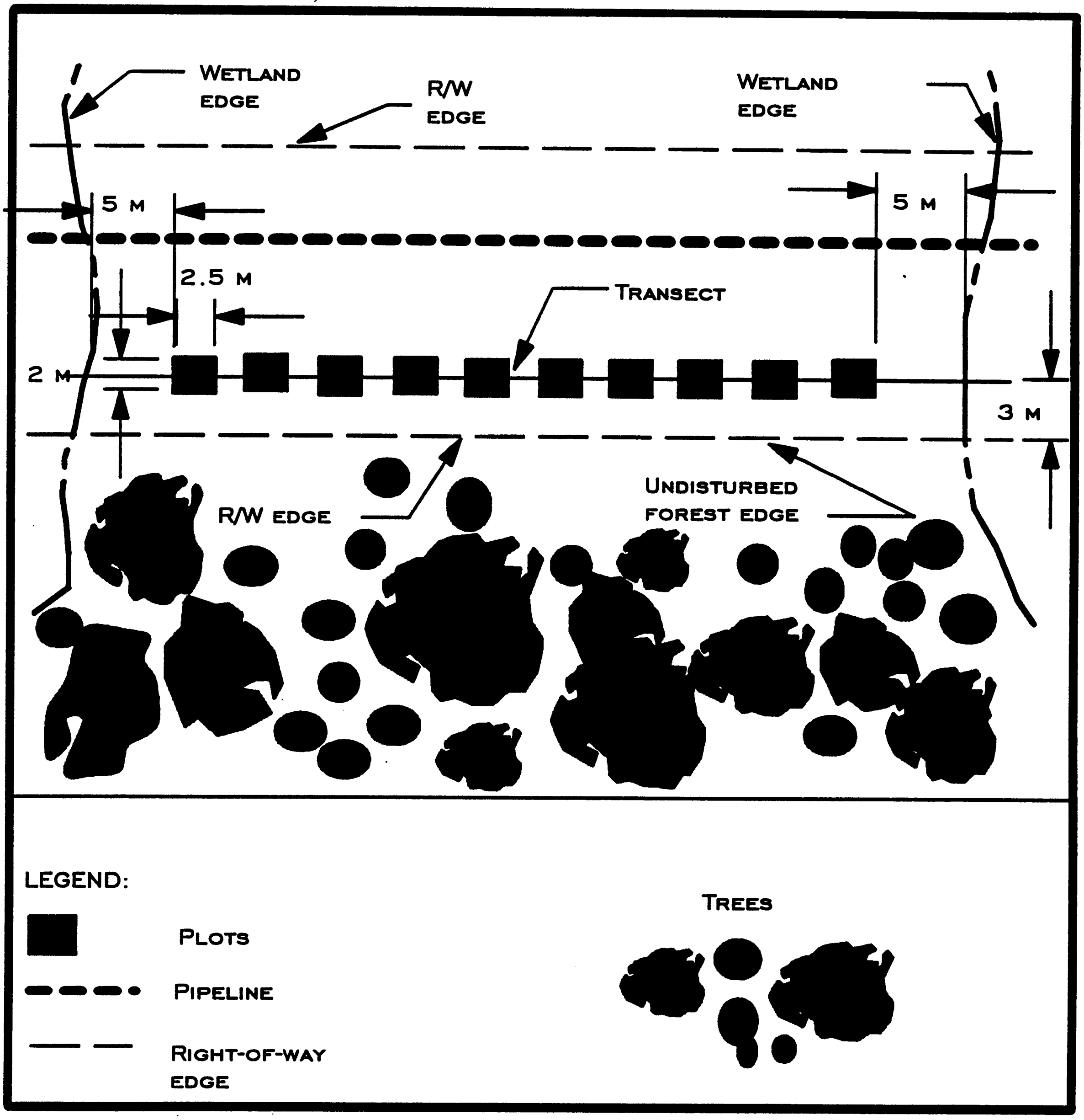

Figure 2. Plot Arrangement

Data analysis for this report focused on the community level. Plot data were averaged within each wetland. Wetland data were then averaged within each community type to achieve community level results. Finally, plot data for the entire study were averaged or summed (as appropriate) to obtain overall study results.

RESULTS

The purpose of the study was to assess post-construction natural tree and shrub recruitment in a linear corridor by observing the natural regrowth occurring after completion of this pipeline. Tree seedling/sapling density and age were utilized for revegetation densities and species composition of the regenerating "canopy" component within disturbed wetland corridors. Shrub species densities and composition were also examined to evaluate the potential for natural regeneration of this wetland community component.

Tree and Shrub Species Densities

To determine the number of trees and shrubs and species mixes returning to the disturbed corridor, tree and shrub species were identified and counts made by species within sample plots. Tree data are presented for average density (D/ha) of one, two and three year age classes and overall average density within each of the five community types (Table 1). Shrub densities for each community type and overall average values are presented (Table 2). Tree and shrub composition are discussed. Also presented are average densities by stem height classes for tree species within each community type (Table 3).

TABLE 1
TREE SEEDLING DENSITIES BY AGE CLASS

COMMUNITY TYPE	AV. D/ha BY AGE			OVERALL MEAN ($\bar{x}$) D/ha
	1 YR	2 YR	3 YR	
Red maple swamp	28412	25698	3459	**57574**
Deep swamp	2471	32864	18038	**17791**
Mixed hardwood flood-plain/seepage slope	9390	741	0	**5066**
Mixed hardwood swamp	10131	5930	3954	**6672**
Mixed hardwood floodplain	9884	20262	8649	**12931**
Overall $\bar{x}$ Density/ha.	**5024**	**7124**	**2842**	**14992**

TABLE 2 SHRUB DENSITIES BY COMMUNITY TYPE	
COMMUNITY TYPE	**MEAN ($\bar{x}$) DENSITY (SHRUBS/HA)**
Red Maple Swamp	**9143**
Deep Swamp	**8278**
Mixed Hardwood Floodplain/Seepage Slope	**6425**
Mixed Hardwood Swamp	**11367**
Mixed Hardwood Floodplain	**2637**
OVERALL AVERAGE (SHRUBS/HA)	**6425**

TABLE 3 WETLAND TREE DENSITY BY HEIGHT CLASS					
COMMUNITY TYPE & WETLAND NO.	**AVERAGE DENSITY BY HEIGHT CLASS (Stems/ha)**				
	<6 IN.	**6-12 IN.**	**12-24 IN.**	**>24 IN.**	**AV. D/ha**
Red Maple Swamp	36818	19768	988	0	**57574**
Deep swamp	4448	26193	20509	2224	**17791**
Mixed hardwood floodplain/seepage slope	8401	1483	247	0	**5066**
Mixed hardwood swamp	7413	5683	4695	2224	**6672**
Mixed hardwood floodplain	15567	18285	3954	988	**12931**
Overall Average Density (Stems/ha)	**6054**	**5950**	**2533**	**452**	**14992**

Average tree densities (Table 1) for the study range from a low of 5,066 trees/ha. (Mixed Hardwood Floodplain/Seepage Slope) to a high of 57,574 trees/ha. (Red Maple Swamp). Overall average density for the study was 14,992 trees/ha. Highest average density for the study was in the two year age class with 7,124 trees/ha. Densities of the three year age class ranged from 0 trees/ha. in the Mixed Hardwood Floodplain/Seepage Slope community to 18,038 trees/ha. in the Deep Swamp. The overall average density of the three year saplings was 2,842 trees/ha. Seedling and sapling distribution was patchy as opposed to regular spacing typically found in a planted area. By averaging data, areas with no (or sparsely distributed) individuals tend to adjust densities downward from areas with more clustered individuals, resulting in values more closely approximating conditions of planting where plants are spread out over the entire area.

Shrubs in these wetlands were noted in the overall species composition, but no counts are available. Accordingly, shrub densities reported do not include these four (4) wetlands and are based on eight (8) wetlands and 80 plots.

Overall average shrub densities for each community type ranged from a low of 2,637 shrubs/ha in the Mixed Hardwood Floodplain to a high of 11,367 shrubs/ha in the Mixed Hardwood Swamp. The overall average shrub density for the study was 6,425 shrubs per hectare.

Red Maple Swamp

Only one wetland of this type was available for this study. Accordingly, one wetland (one transect, ten plots) was sampled in this community type. As expected, the primary species sampled in this transect was red maple. High D/ha values occurred in the one and two year age classes (Table 1) with the highest in the two year group. The mean D/ha is 57,574 trees/ha. Non-wetland species (pines) were present, but were not included in the density calculations.

Highest densities for the study occurred in this wetland. One hundred fifteen (115) one year seedlings and 104 two year seedlings were present in the ten study plots for densities of 28,412 and 25,698 trees/ha, respectively. Three year old seedlings totalled fourteen (14) for a density of 3,459 trees/ha.

The mean D/ha for shrubs was 9,143. Two species comprised the shrub component; Virginia willow (Itea virginica) and wax myrtle (Myrica cerifera). One plot (number 3) was shifted 18 m to the east to allow sampling within vegetational zones, an area of red maple seedling distribution.

Deep Swamp

Three wetlands (three transects, thirty plots) were sampled in this community type. Primary species encountered included cypress, blackgum, pine and red maple. Highest D/ha values occurred in the two year age class with 32,864 trees/ha. Thirty-seven percent of the plots in this community type were comprised of cypress, blackgum or mixtures of these two species. Pines were not included in calculations, mean D/ha was 17,791.

Two of these wetlands exhibited some microtopography in the form of stumps remaining in the sampling area. These stumps appear to provide potentially drier germination sites for seeds of the typical Deep Swamp species in areas of otherwise long term hydroperiods. Seedlings and stump sprouts appeared to be most commonly associated with, or clustered around, these slightly raised areas.

The Deep Swamp mean D/ha for shrubs was 8,278. This density was based on results from only two of the three wetlands. No shrub count data are available for Wetland 16. Wetland shrub species included; myrtle-leaved holly (Ilex myrtifolia), Virginia willow and fetterbush (Lyonia lucida).

Mixed Hardwood Floodplain/Seepage Slope

Two wetlands (two transects, twenty plots) were sampled in this community type. Species consistently found included red maple and sweetgum (Liquidambar styraciflua). Highest D/ha values (Table 1) occurred in the one year age class.

The mean D/ac for this community type is 9,390. Pine trees were present in the study corridors of this community type although not included in calculations.

The shrub mean D/ha for these two wetlands was 6,425. Non-wetland shrub species including, American beautyberry (Callicarpa americana), groundsel tree (Baccharis halimifolia) and sumac (Rhus copallina), were encountered in this community type. These species were not included in the calculations. Wetland shrub species occurring in higher densities (988 to 2224 shrubs/ha) included wax myrtle, buttonbush (Cephalanthus occidentalis) and Virginia willow.

Mixed Hardwood Swamp

Three wetlands (three transects, thirty plots) were sampled in this community type. Recurring species included red maple and loblolly pine (Pinus taeda). Loblolly bay (Gordonia lasianthus) occurred consistently in one wetland. Highest D/ha values occurred in the 1 year age class (Table 1). Red maple and blackgum accounted for one third of the individuals in this category. The mean D/ha for the Mixed Hardwood Swamp is 6,672 with the exclusion of pine from the data set. One plot was shifted to avoid the open water area of a creek in one wetland and to ensure sampling of the shallow creek bottom area in another.

Pines were also present in significant numbers within two wetlands of this community type, but not included in calculations. One of these wetlands (Wetland 25) is an anomaly (and perhaps should not have been included in the study) in that the western portions are adjacent to and part of a more mesic pine and oak woods with more saturated conditions.

Shrub data were only available from Wetland 21 within this community type. Accordingly, density data are for this single transect with mean D/ha of 11,367. The wetland shrub component was dominated by fetterbush and wax myrtle.

Mixed Hardwood Floodplain

Three wetlands (three transects, thirty plots) were sampled in this community type. The species consistently found in these transects was red maple. Highest D/ha values occurred in the 2 year age class (Table 1). The mean D/ha for the community type is 12,931. Two wetlands had distinct zones of vegetation requiring shifting of one plot in each wetland.

Willows dominated the shrub component of Wetland 9. Additionally, buttonbush and groundsel tree were noted as present on the data sheets, but were not counted. Utilizing data available, the mean D/ha for shrubs in this community type was 2,637. Wetland 39 also contained monotypic stands of buttonbush.

Tree Height

Tree height data were collected within each of the plots. These data were not collected in conjunction with age so no direct correlation with, or inference on, growth rates can be made. Such analysis is beyond the scope of this study.

Densities of trees within each height class were calculated from the data collected (Table 3). For the study, the overall average density of trees one foot or greater in height was 2,985 stems/hectare. Overall average density of stems greater than two feet in height was 452 stems/ha.

Data show (Table 3) that 20% of wetland tree stems counted were at least one foot tall. Highest percentage of stems over one foot tall occurred in the Deep Swamp community type with 63% of all stems over one foot in height occurring here. The Mixed Hardwood Floodplain/Seepage Slope had the lowest percentage of stems over one foot at only one (1) percent.

DISCUSSION

Planting densities of between 988 and 13,343 trees/ha (400 and 540 trees/acre) have been employed on past wetland restoration and creation projects to meet state and federal permit requirements. Typical requirements specify a final density of 988 trees/ha (400 trees/ac). Shrub densities of 2,990 shrubs/ha (1,210 shrubs/ac) were chosen as typical for similar reasons. These values were used as comparison values in this study to evaluate natural regeneration as an alternative to planting.

Tree and Shrub Species Density and Composition

During the study, both wetland and upland species were counted. This approach was taken to obtain an accurate picture of species regeneration within each wetland and community type. Densities were calculated, however, excluding all non-wetland species counted and are presented as such in the tables. In all five community types, wetlands contained non-wetland trees within the study corridor.

Tree species densities were highest (57,574 trees/ha) in the Red Maple Swamp community. With high seed production and high germination rates the large number of seedlings is not surprising. Highest values for D/ha by age class occurred in the two year group, being 42% higher than the one year class and 151% higher than the three year class. The three year age class comprised 19% of the overall average density for the study.

As stated earlier, a total of five plots in five different wetlands (Wetlands 8a, 17, 21, 25 and 39) were adjusted to ensure sampling of zones. One such shift in Wetland 8a placed the plot in an area of high seedling density. Sampling this edge may have skewed the results to a higher density as this plot had the highest density of the entire study (338,527 seedlings/ha). If this plot and plot number ten (also having high densities, 269,339 seedlings/ha) are removed from the data set, D/ha for this community type is 17,297 trees/ha. Densities within the other plots which were shifted were all within the range of the rest of the data set and appear not to have had an effect on the data.

The Red Maple Swamp (Wetland 8a) and Wetland 4 of the Mixed Hardwood Swamp type have experienced prolonged inundation of their central portions since construction. Seedling establishment within these deeper centers has been very low to nonexistent. It appears that water levels have fluctuated little in the three years and seedling establishment has occurred mainly in more landward plots. Contours within the disturbed corridor appear to be similar to those of the adjacent community suggesting the hydroperiod has not been significantly altered due to construction activities. As hydrologic cycles fluctuate over time, conditions favorable for seedling establishment should coincide with seed fall and germination periods allowing seedlings to establish within these central zones. Existence of mature forested communities directly adjacent to the disturbed corridor suggests that at some time, water levels were such that seedlings established and survived to maturity.

Study results show that tree seedlings of appropriate wetland species are recruiting into the corridor every year. With new recruitment every year, natural regeneration is providing continually for replacement material to ensure success.

Tree species typical of each wetland community type compared to those found in the adjacent undisturbed wetland are returning to the corridor. No attempt was made in this study to assess community similarity or diversity, however.

Through monitoring, areas having patchy distribution during early regeneration and species which may not readily return to the area can be identified. As time progresses, monitoring data can be used to assess the need for supplemental planting efforts to adjust for such occurrences and ensure the return of a natural forested system.

Overall density of wetland shrubs was 6,425 shrubs/ha, two times the generally accepted planting densities of 2,990. Wetland shrubs encountered frequently include; Virginia willow, buttonbush, fetterbush and sweet pepperbush. As with trees, a logical assumption is that a typical wetland shrub component will develop as appropriate for each community type within the right-of-way.

Tree Height

As stated previously, no attempt was made in this study to evaluate tree growth rates. However, tree seedling and sapling heights were assessed to give a picture of stem heights following three growing seasons in conjunction with natural revegetation. Highest densities were recorded for wetland tree stems less than one foot tall at 583 (or 80%). Stems over one foot in height accounted for 20% (145) of the total wetland tree stems in the study. Three (3) percent (22 trees) were greater than two feet in height.

When densities are examined by height class, the data show that there were 2,985 trees/ha greater than 24 inches in height over the entire study. This value is three times the typically required density. Trees greater than 24 inches only constitute 46% of accepted planting densities, however.

CONCLUSIONS

Data collected during this study support natural revegetation of linear corridor facilities, particularly in areas similar to northern and panhandle portions of Florida. This study shows that native wetland tree species are establishing within the construction corridor in densities which exceed the 988 to 1,334 trees/ha typically employed in planting efforts. Based on the results of data collection, average densities of over 5,000 trees/ha (and up to 57,574 trees/ha) overall average densities exceeding 2,800 trees/ha are found to naturally occur following construction. The low numbers of trees greater than 24 inches should not be of concern. Herbivore pressure and similar height reduction is also often seen in planting efforts as animals select the young shoots. The occurrence of 46% of the accepted tree densities naturally achieving this height (and the existence of mature forests) suggests a mature forest component will be achieved over time. Replanting may reduce the time required to reach maturity, but even that is not guaranteed. Furthermore, based on this study, the reduction in time may not be significant over the 30 to 50 years required to grow a mature forest.

Similar results are expected along other linear corridors, especially within northern and panhandle areas of Florida. Each project should be examined for physiographic conditions which may be unfavorable for natural regeneration, or where physical conditions are such that seed sources are not immediately adjacent to the construction corridor. Where such unfavorable conditions do not exist, replanting should not be necessary to reestablish native forested and shrub communities.

It seems natural regeneration should be utilized whenever possible for several reasons. First, as shown by this study, seedlings will germinate in densities which should be sufficient to support a successful forested community over time. Moreover, under natural regeneration, the trees which survive will be typical of the natural community existing at the site and should mimic the densities, spacing and composition of the adjacent undisturbed wetland and, therefore, the preconstruction wetland. Trees resulting from natural regeneration will be of the same genetic stock and therefore should be better able to persist with naturally fluctuating environmental conditions in that location. Finally, additional disturbance to the wetland and surrounding area will be minimized if replanting is not required.

WEEDON ISLAND PRESERVE DEVELOPMENT FOR ECOSYSTEM MANAGEMENT AND PUBLIC USE

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ABSTRACT

Weedon Island Preserve, in St. Petersburg, Florida, has 11 different ecosystems and is home to as many as 37 endangered, threatened, and special-concern species. Parsons Engineering Science has been contracted by Pinellas County to prepare a Master Plan for design of proposed improvements to the preserve. Parsons ES also investigated the existing ecological and cultural resources in the areas proposed for improvement, assessed impacts to these resources, and made recommendations for ecosystem management and improvement. These recommendations focused on placement of facilities in previously-disturbed areas, native plantings, exotic species control, and mitigation for gopher tortoise relocation, and wetland mitigation.

INTRODUCTION

Weedon Island Preserve represents an excellent example of a community's efforts at ecosystem restoration, preservation, and management. The preserve was established in 1974 as the first purchase under Florida's Environmentally Endangered Lands Act. Pinellas County, the City of St. Petersburg, and the Southwest Florida Water Management District are currently funding the development of the preserve to manage, restore, and enhance its ecosystems, while providing visitors with aesthetic, educational, and recreational enjoyment with emphasis on interpretation of the natural and cultural attributes of the preserve.

Pinellas County has proposed improvements to the Weedon Island Preserve including upgrading the main access road, constructing a maintenance building and new residence for the preserve superintendent, replacing the fishing pier, constructing infrastructure to support an interpretive center, and constructing boardwalks and observation towers. Many of the areas proposed for improvement will not be significantly altered from their existing state, with the exception of the interpretive center complex.

The County contracted with Parsons Engineering Science (Parsons ES) to develop a Master Plan to implement for the Unit Management Plan (Pinellas County Park Department, 1993). The stated goal of the management plan is to develop compatible facilities that will provide for a variety of outdoor recreational experiences and serve to protect examples of native Florida for the enjoyment of the public. In other words, the County intends to provide the public the ability to experience the unique natural and man-made habitats on the island while protecting and enhancing the existing ecosystem.

Along with developing the Master Plan for the preserve improvements, Parsons ES investigated the existing ecological and cultural resources at the site, assessed impacts to the areas planned for development, and made recommendations for ecosystem management and enhancement.

STUDY SITE

Weedon Island Preserve is located in St. Petersburg, on the west shore of Old Tampa Bay (Figure 1). It covers 250 hectares (627 acres) of upland and submerged land, and consists of a group of low-lying islands of which Weedon Island is the largest (Figure 2). The smaller islands, Mud Hole Island, Benjamin Island, Snake Island, Christmas Island, Googe Island, and Ross Island, are all located to the east and south of the main island. Tampa Bay is the main body of water to the east and south of Weedon Island, while Riviera Bay and Grande Bayou form the western border. A spit of land, located between Masters Bayou to the north and Riviera Bayou to the south, connects Weedon Island to the St. Petersburg side of the mainland. Florida Power's Bartow oil-fired power plant lies to the north of the preserve.

The preserve encompasses 11 ecological communities, from upland to wetland to aquatic (Figure 3). Thirty-seven endangered, threatened, or special-concern species have been identified within the preserve boundaries. Weedon Island has been designated an aquatic preserve and all permanent water bodies are designated Outstanding Florida Waters.

In addition to its outstanding ecological attributes, Weedon Island also has a rich history, with cultural resources representing human presence dating back as far as 400 AD. Dr. Leslie Weedon, an early owner of the island, found three graves of the Spanish Colonial period, containing iron armor and other artifacts. Other archaeological explorations of the island's burial mounds and shell mounds conducted earlier in this century found Indian skeletons, ceramic vessels, and shells. The Indian burial mound at the north end of the island is listed on the National Register of Historic Places.

Weedon Island also has ties to the early and recent history of Pinellas County. A speakeasy and dance hall provided entertainment during the 1920's; the headquarters of Eastern Air Transport were located at Grand Central Airport in the uplands of the island; and the Sun Haven Movie Studio made movies on a soundstage on the island (Thompson, 1992).

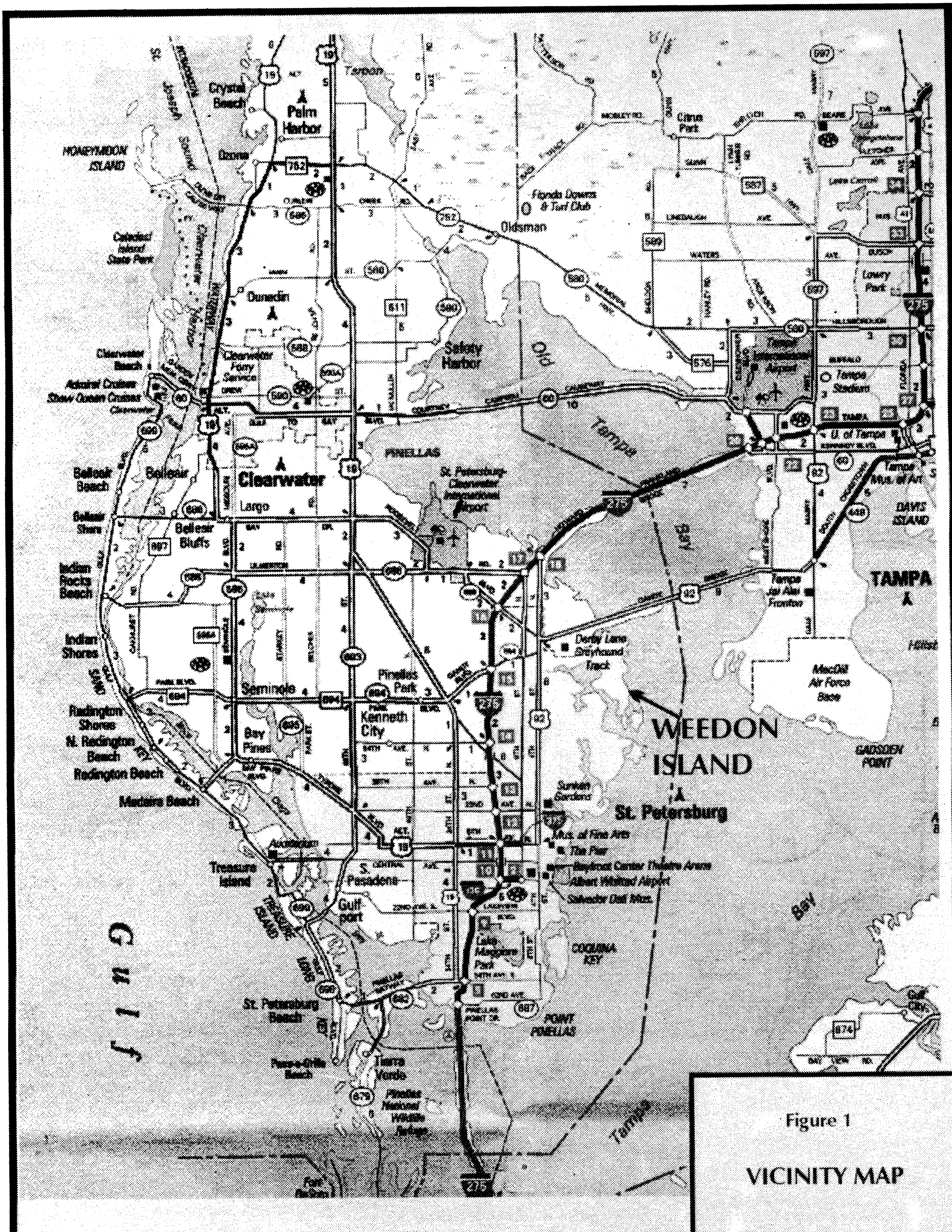

Figure 1

VICINITY MAP

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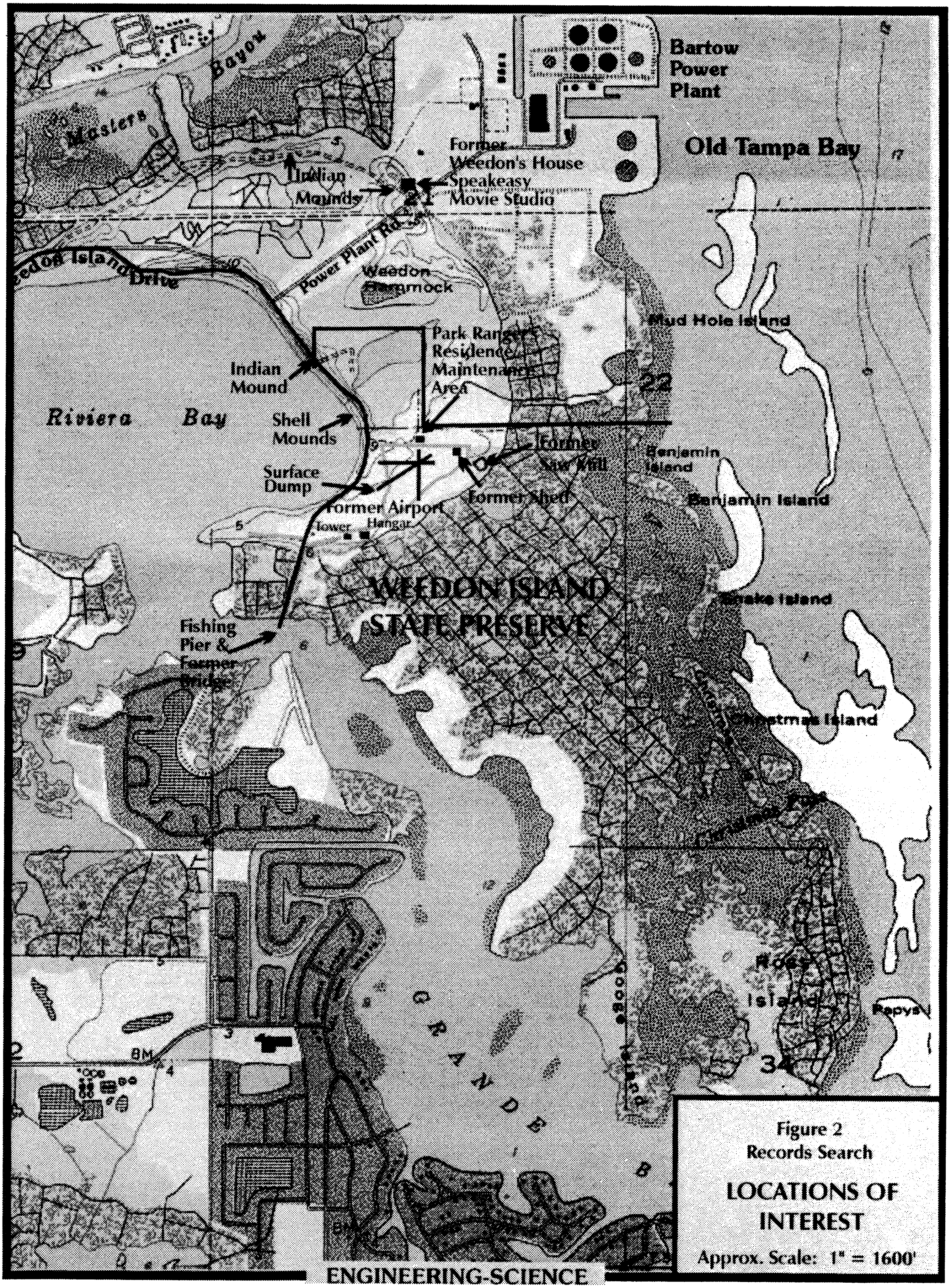
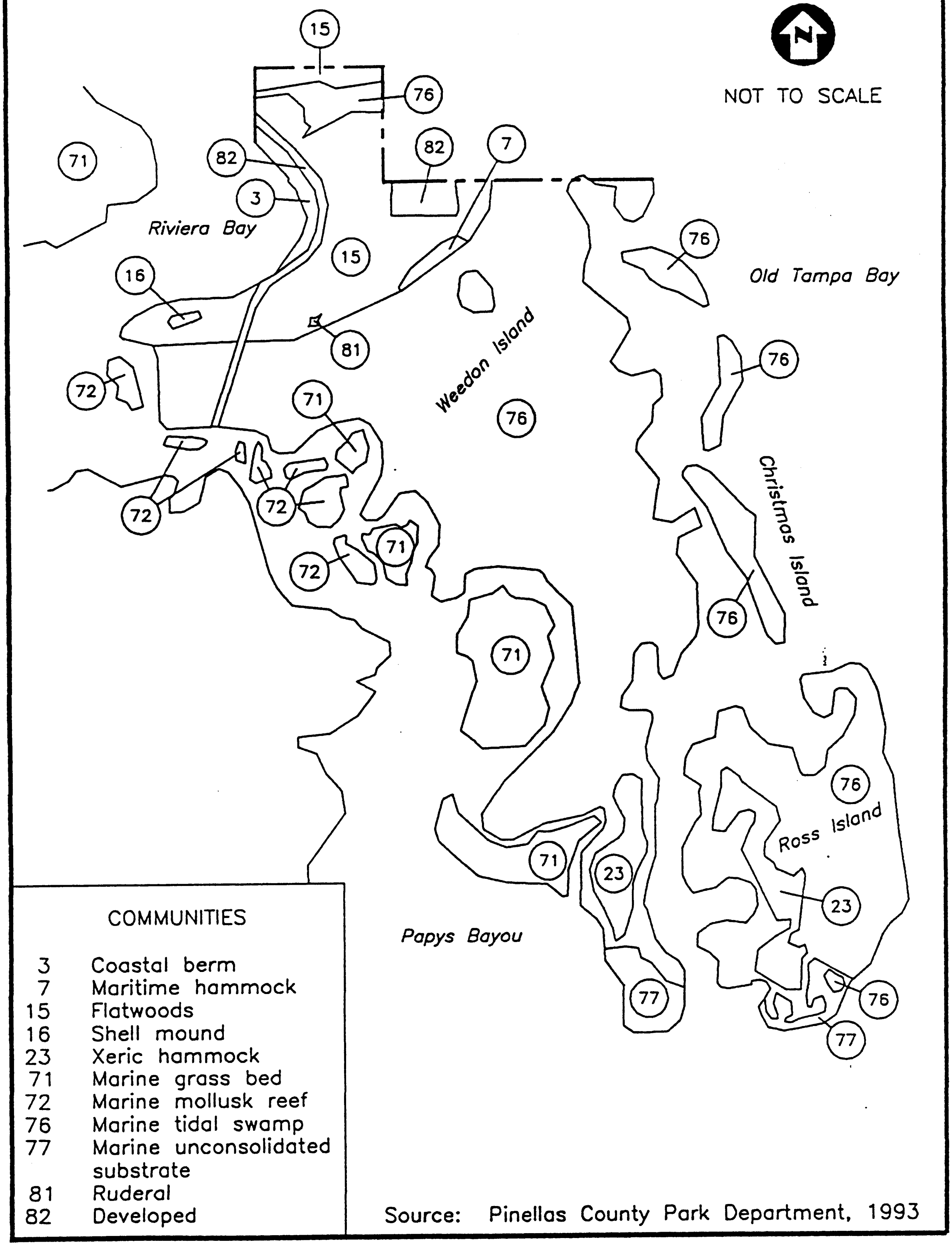


Bartow Power Plant
Old Tampa Bay
Former Weedon's House Speakeasy Movie Studio
Indian Mounds
Power Plant Rd
Weedon Island Drive
Weedon Hammock
Mud Hole Island
Park Rangers Residence Maintenance Area
Indian Mound
Riviera Bay
Shell Mounds
Former Saw Mill
Benjamin Island
Surface Dump
Former Shed
Former Airport
Tower
Hangar
WEEDON ISLAND STATE PRESERVE
Snake Island
Fishing Pier & Former Bridge
Christmas Island
GRANDE
Papys
Figure 2
Records Search
LOCATIONS OF INTEREST
Approx. Scale: 1" = 1600'
ENGINEERING-SCIENCE

Figure 3. Ecological communities at Weedon Island Preserve



Source: Pinellas County Park Department, 1993

METHODS AND MATERIALS

As the Unit Management Plan states, "preservation and enhancement of natural conditions is most important" in management of the Weedon Island Preserve. To accomplish these preservation and enhancement goals, the County contracted with Parsons ES to survey the existing state of the ecosystems, including plants and wildlife, vegetation communities, protected species, wetlands, and exotic plant control practices.

Parsons ES conducted ecological surveys during October 1994 to identify and define the natural communities within the areas proposed for improvement, as shown in Figure 2. Comprehensive upland vegetation and wildlife surveys were conducted for areas surrounding the proposed location of the interpretive center complex, which lies in the largest contiguous area of native vegetation. General community evaluations were conducted for the five other management areas. As part of the surveys, biologists identified all plant communities, noted species composition, made wildlife observations, and identified protected flora and fauna that use or potentially use the area. Recommendations were made on ways of limiting the impact of the improvements and maintaining and improving the existing ecosystems.

At the proposed interpretive center location, upland vegetation was surveyed along defined transects at 60-foot intervals along a northwest to southeast baseline. Percent aerial coverage of the canopy, shrub/scrub, and herb layers was visually estimated for the dominant plant species between the transects. Qualitative natural community evaluations were conducted for the remaining improvement locations. Natural community descriptions as defined in the Guide to Natural Communities of Florida (Florida Natural Areas Inventory [FNAI], 1990) were used to classify all of the natural communities within or adjacent to the five major development areas.

Surveys for federal and state-listed endangered, threatened, and special-concern species were incorporated into the detailed vegetation survey. All observed gopher tortoise (Gopherus polyphemus) burrows were mapped and classified as active, inactive, or old, using the criteria of Auffenburg and Franz (1982). A conversion factor of 0.614, developed by Auffenburg and Franz to equate the number of active and inactive burrows counted to the actual number of tortoises residing on the site, was used. Use of the standard 0.614 conversion factor is recommended by the Wildlife Methodology Guidelines (Allen, 1988) and is widely used in Florida as no other consensus conversion factor is available for differing regions of the state or habitat types.

Wetlands are the largest ecological resource on the Weedon Island Preserve, totaling approximately 160 hectares (400 acres). Parsons ES identified wetlands in the improvement areas and located jurisdictional boundaries. Wetland involvement and permitting issues were evaluated for the improvement areas.

Another aspect of ecosystem management being implemented at the preserve is exotic plant control. Historical development activities on Weedon Island have led to the invasion and replacement of native plant communities by aggressive, non-native (exotic) plant species that have an adverse effect on the native plant and animal communities. One resource management objective that Pinellas County is pursuing for the preserve is restoration and preservation of native plant communities. This objective includes management, and elimination where possible, of exotic and nuisance plant species. Parsons ES reviewed and made recommendations on the County's Exotic Plant Control Management Plan to determine if any changes need to be made to the plan, and to provide recommendations for revisions and additions to the plan where necessary. Aspects of the current control plan that Parsons ES reviewed include locations where controls are used, plant species that are controlled, types and methods of physical and chemical controls, control schedules, and disposition of plant materials, as well as the effects of proposed preserve improvements on the exotics control plan.

RESULTS

Vegetative Community Survey Results

Plant species observed during the Parsons ES survey are listed by vegetative community type in Table 1. Because the survey covered only those areas proposed for improvement, this list is not comprehensive for the entire preserve.

In general, the vegetation survey identified flatwoods as the dominant plant community at the proposed location of the interpretive center. As part of this community type, several distinct vegetative phases were observed, including mesic pine flatwoods, dry prairies, and scrubby flatwoods. In addition to the flatwoods associations, several areas of disturbed land were identified that have not been completely recolonized by native vegetation.

The mesic pine flatwoods community is characterized by an open canopy forest of widely spaced pine trees (Pinus eliotii) and cabbage palms (Sabal palmetto) with a dense ground cover of saw palmetto (Serenoa repens) and various herbaceous species. Mesic pine flatwoods occur on relatively flat, moderately to poorly drained soils.

The dry prairie community is characterized as a nearly treeless plain with a dense ground cover of saw palmetto, wiregrass (Aristida stricta) and other grasses, forbs, and low shrubs. The dry prairie community is similar to the mesic pine flatwoods area but lacks pine and palm trees or their density is below 2.5 trees per hectare (1 tree per acre).

The scrubby flatwoods community is drier than the mesic flatwoods or dry prairies. Dominant scrubby flatwoods vegetation includes live oak (Quercus virginiana, Q. geminata), Chapman oak (Q. chapmanii), slash pine, cabbage palm, saw palmetto, and rusty lyonia (Lyonia ferruginea). Scrubby flatwoods are found at slightly higher elevation than mesic flatwoods or dry prairies. Soils consist of well-drained, white, sandy soils.

Table 1. Weedon Island Preserve - Plant Species List
Weedon Island Preserve - November 1994

Scientific Name	Common Name	Observed					
		Mesic Pine Flatwoods	Dry Prairie	Scrubby Flatwoods	Estuarine Tidal Swamp	Estuarine Brass Bed	Ruderal
Ambrosia artemisiifolia	Ragweed	x	x	x			x
Ampelopsis arborea	Pepper vine						x
Andropogon floridanus	Florida bluestem	x		x			
Andropogon virginicus	Broomsedge	x	x	x			
Aristida stricta	Wire grass	x	x	x			x
Asimina reticulata	Pawpaw	x		x			
Avicennia germinans	Black mangrove				x		x
Baccharis halimifolia	Salt myrtle	x			x		
Bambusa sp.	Bamboo						x
Blechnum serrulatum	Swamp fern	x	x		x		x
Callicarpa americana	Beautybush	x	x	x			x
Canavalia maritima	Seaside bean			x			
Casuarina equisetifolia	Australian pine				x		
Cenchrus incertus	Sandspur	x	x	x			x
Chamaecrista sp.	Sensitive plant	x	x				x
Chrysopsis sp.	Goldenaster			x			x
Conocarpus erecta	Buttonwood				x		
Cuscuta gronovii	Dodder	x	x	x			x
Cyperus ligularis	Alabama flat sedge				x		
Dactyloctenium aegyptium	Crow foot grass	x	x				
Dioscorea bulbifera	Air potato						x
Distichlis spicata	Seashore saltgrass				x		x
Eragrostis sp.	Love grass	x	x	x			x
Erechtites hieracifolia	Firewood		x				x
Eupatorium capillifolium	Dog fennel	x	x	x			x
Galactia elliottii		x	x				x
Halodule wrightii	Shoalgrass					x	
Hypericum gentianoides	Pineweed	x	x				
Hypericum tetrapetalum	St. Peters wort	x	x				
Ilex glabra	Gallberry	x					
Iva frutescens	Marsh elder				x		
Laguncularia racemosa	White mangrove				x		
Lantana camara	Lantana	x	x	x			
Liatris sp.	Blazing star			x			
Lippia nodiflora	Frog-fruit		x				x

Table 1. Weedon Island Preserve - Plant Species List (continued)

Scientific Name	Common Name	Observed					
		Mesic Pine Flatwoods	Dry Prairie	Scrubby Flatwoods	Estuarine Tidal Swamp	Estuarine Brass Bed	Ruderal
Ludwigia decurrens	Primrose willow				x		x
Lyonia ferruginea	Rusty lyonia	x	x	x			
Lyonia lucida	Shiny lyonia	x		x			
Magnolia virginiana	Sweet bay magnolia	x					
Malavaviscus arboreus	Turk's-cap mallow	x					x
Melaleuca quinquenervic	Melaleuca						x
Melia azedarach	Chinaberry						x
Mikania scandens	Hemp vine	x	x				x
Momordica charantia	Wild balsam apple	x					x
Monarda punctata	Horsemint			x			x
Myrica cerifera	Wax myrtle	x					
Opuntia stricta	Prickly-pear cactus	x	x	x			x
Physalis walteri	Ground cherry						x
Phytolacca americana	Pokeweed	x					
Pinus elliottii	Slash pine	x		x			x
Poinsettia cyanthophora	Painted-leaf	x	x				x
Polygala nana	Batchelor button		x				
Pteridium aquilinum	Bracken fern	x	x	x			x
Pterocaulon virgatum	Blackroot	x	x	x			x
Quercus chapmanii	Chapman's oak			x			
Quercus geminata	Scrub live oak			x			
Quercus virginiana	Virginia live oak			x			
Rhizophora mangle	Red mangrove				x		
Rhus copallina	Winged sumac	x		x			x
Ruppia maritima	Widgeon-grass					x	
Sabal palmetto	Cabbage palm	x		x			x
Sabatia brevifolia	Narrow leaved sabatia	x	x				
Salicornia virginica	Glasswort				x		
Schinus terebinthifolius	Brazilian pepper	x		x	x		x
Scoparia dulcis	Sweet broom	x	x				x
Serenoa repens	Saw palmetto	x	x	x			x
Sesuvium portulacastrum	Sea purslane				x		
Setaria sp.	Small foxtail		x				
Smilax auriculata	Greenbriar	x	x	x			x
Smilax laurifolia	Catbriar	x	x	x			x

Table 1. Weedon Island Preserve - Plant Species List (continued)

Scientific Name	Common Name	Observed					
		Mesic Pine Flatwoods	Dry Prairie	Scrubby Flatwoods	Estuarine Tidal Swamp	Estuarine Brass Bed	Ruderal
Strophostyles helvola	Sand bean				x		
Thalassia testudinum	Turtle grass					x	
Toxicodendron radicans	Poison ivy	x		x			x
Urena lobata	Caesar-weed	x					x
Vaccinium myrsinites	Shiny blueberry		x	x			
Vitis rotundifolia	Muscadine	x					x
Yucca aloifolia	Spanish bayonet			x			

These flatwoods communities are classified by the Florida Game and Freshwater Fish Commission as potential gopher tortoise habitat and 72 gopher tortoise burrows were observed during the Parsons ES site survey.

A building with paved access and parking areas, landscaping, and a storm water retention facility is planned for the interpretive center complex. Parsons ES recommended that these be located within previously disturbed portions of the site to minimize impacts to the native flatwoods community. New plantings will be material native to the preserve or to the types of communities found in the preserve. This will reduce the need for irrigation while preserving and enhancing the native ecosystems.

The other areas proposed for improvements are located in a variety of communities.

Natural communities located adjacent to the main access road vary from scrubby flatwoods to estuarine tidal swamp. The latter is dominated by red mangrove (Rhizophora mangle), black mangrove (Avicennia germinans), white mangrove (Laguncularia racemosa), and buttonwood (Conocarpus erecta) in the canopy, with glasswort (Salicornia virginica) and sea purslane (Sesuvium portulacastrum) as ground cover. This study recommended that improvements to the access road use the existing roadway surface, to avoid impacts to natural communities and to archaeological resources.

The maintenance and residence area improvements are proposed for a location that currently supports similar uses. This location is surrounded by mesic flatwoods, and no expansion of the existing developed areas is proposed.

The fishing pier is proposed to be removed and replaced in the same footprint as the existing pier. Natural communities associated with this structure include estuarine grass beds, tidal swamp along the shoreline, and developed land along the access road to the fishing pier. Estuarine grass beds are extensive on all sides of Weedon Island and are vegetated with the marine grasses Thalassia testudinum, Halodule wrightii, Ruppia maritima, and various algae. Because the footprint of the pier will not be changed, no permanent impact is expected to the natural communities.

The boardwalk alignments are located in the tidal swamp community. In most areas, the boardwalk is proposed to be located on top of existing spoil berms, which were created during dredging of mosquito control ditches. Vegetation along the spoil mounds is dominated by the exotics Australian pine (Casuarina equisetifolia) and Brazilian pepper (Schinus terebinthifolius). Placement of the boardwalk along the spoil mounds will have the benefit of requiring removal of exotic vegetation and will minimize impacts to the tidal swamp natural community.

Protected Species Review

The detailed survey of protected species focused on the gopher tortoise, federally-classified as a Category II species, and state-listed as a species of special concern. Parsons ES observed 72 gopher tortoise burrows, 22 active, 34 inactive, and 16 old, in the area proposed for the interpretive center complex. Approximately 100 percent of the site was surveyed, since all of the area is potential gopher tortoise habitat. Only active and inactive burrow locations are considered in the population estimate, per the method of Auffenburg and Franz (1982).

In the flatwoods community, 56 burrows (22 active and 34 inactive) times the 0.614 conversion factor equates to 34 tortoises that use the proposed improvement area. This indicates a density of approximately 1 tortoise per hectare.

To allow development of the interpretive center complex, Parsons ES recommended relocation of tortoises through consultation with Florida Game and Fresh Water Fish Commission. Procedures necessary for this alternative include identification and survey of a recipient site capable of supporting an increase in tortoise density, securing relocation permits, trapping and moving the tortoises, and development of a relocation plan.

Construction of other proposed improvements is not expected to directly affect protected species. Parsons ES recommended that the County coordinate with the U.S. Fish and Wildlife Service and the National Marine Fisheries Service to identify precautions required in potential manatee (Trichecus manatus) areas during construction of the replacement fishing pier.

Wetland Evaluation

With the exception of the boardwalk entrance, no wetlands were identified in the proposed interpretive center area, nor in the area proposed for maintenance building/residence improvements.

Construction of the boardwalk entrance is proposed to be located in a wetland located south of the interpretive center complex. Vegetation observed along the upland edge included beautybush (Callicarpa americana), Turk's-cap mallow (Malavaviscus arboreus), saw palmetto, and horsemint (Monarda punctata). The wetland has 7 to 15 meters of predominantly freshwater wetland vegetation along the upland/wetland fringe. A predominance of Alabama flatsedge (Cyperus ligularis), swamp fern (Blechnum serrulatum), and hydric soils define the wetland boundary. Tidal swamp vegetation was observed in the main portion of the wetland. Red mangrove, black mangrove, white mangrove, and buttonwood are the dominant species. Australian pine and Brazilian pepper vegetate the upland spoil berms within the wetland.

Construction of approximately 3,500 linear meters of boardwalks with tiered observation towers will impact these wetlands. Parsons ES recommended that, whenever possible, the boardwalks be located on top of upland spoil berms created by historical mosquito ditching. Minimal impacts from crossing mangrove-lined ditches are expected. Construction of observation towers in the wetlands will require temporary construction of access roadways. These access roadways will be located on top of upland spoil berms or routed through non-forested wetlands. Where wetland impacts are minimal, the roadways will be considered to have temporary impact. Disturbed areas shall be restored to the pre-construction condition. Excessive or permanent impacts to wetlands will require mitigation.

In order to determine mitigation and/or compensation requirements for the boardwalk impacts, cross sections of representative wetland ditch crossings will be prepared and an average wetland impact will be determined. The total wetland impact will be calculated by multiplying the number of crossings by the average acreage of wetlands per crossing. Mitigation for impacts to tidal swamp wetlands is proposed at one-to-one, type-for-type. Any additional mitigation may be achieved through enhancement activities, exotic plant management plans, etc.

Improvements to the main access road are expected to be contained in the existing cleared right-of-way with the exception of constructing additional parking and improving rest room facilities near the fishing pier and possible improvements to a culvert crossing. A portion of the existing access road has jurisdictional wetlands along one or both sides. These tidal swamp wetlands are dominated by mangroves.

The improvements to the main access road turn-around and rest room area will impact historically disturbed wetlands. These impacts will not require wetland mitigation. However, impacts that may occur to the adjacent tidal swamp wetlands, including those at the culvert crossing, will require wetland mitigation.

The fishing pier is located in the tidal waters of Riviera Bay. Wetlands associated with this area include estuarine grass beds located in the general vicinity and tidal swamp along the shoreline. Replacement of the fishing pier will be in the same footprint as the existing structure; therefore, no significant impacts to wetlands are expected.

Exotic Plant Control

Aerial photos dating as far back as 1926 show vegetation disturbance from dredge spoil piles and clearing activities associated with cultivation construction of dwellings. Most of the upland areas within the preserve were altered during development associated with the dance hall and the airfield.

Predominant exotic tree species present in disturbed upland and lowland areas of the preserve include Brazilian pepper, Australian pine, ear-tree (Enterolobium contortisiliquum), chinaberry (Melia azedarach), and melaleuca (Melaleuca quinquenervia). Additional exotic species include bamboo (Bambusa sp.), Turk's cap mallow, and air potato (Dioscorea bulbifera). In addition, most mangrove areas of the preserve were draglined for mosquito control in the 1960s. Spoil mounds from these dredging activities have been heavily colonized by Australian pine and Brazilian pepper.

The state initially implemented an exotic plant control program to eradicate exotic and nuisance plant species from the preserve, as detailed in the Exotic Plant Control Management Plan section of the Unit Management Plan. This program has continued under Pinellas County management of the preserve. The exotic plant control activities to date have successfully eliminated undesirable species in many accessible areas of the preserve; most of these areas are currently in the maintenance phase. Due to a lack of manpower and funding, the exotics control management plan has not been implemented in most areas that are less accessible.

Current exotic plant control techniques used at the preserve include manual removal, herbicide application with Garlon 4/diesel solution, and burning. Mechanical removal is not effective for control of Australian pine, Brazilian pepper, or melaleuca when used alone because the soil disturbance creates conditions for regrowth from seeds and root fragments, and allows further invasion by pioneering exotic plants. Intense follow-up with other control methods is also required. In addition, this technique is not recommended for natural areas due to disturbance of soil and potential damage to non-target vegetation. Burning is currently used for maintenance of flatwoods areas, but is not applicable to remaining areas requiring exotics control (mangroves, xeric hammocks). Water level manipulation is not readily applicable due to constraints and tolerance of exotic species present. Effective biological controls are not currently available for exotic species present at the preserve.

Activities associated with the proposed management plan will provide an opportunity for exotics removal and maintenance in some areas of the preserve by providing improved access. For example, the approximately 3.5 kilometer boardwalk will improve access for exotics removal, disposal of plant materials, and maintenance in mangrove areas. Design of the boardwalk to allow small carts and chippers/shredders will greatly facilitate removal of exotics along, and adjacent to, the proposed boardwalk.

Parsons ES recommended that precautions be taken during all grading and filling activities associated with the preserve improvements, as well as the removal of large-stem exotics, to minimize the enhancement of seedbed conditions. Damage to non-target vegetation and potential water quality impacts are potential concerns for exotics control in mangrove areas. Herbicides effective for Australian pine and Brazilian pepper control are not selective and mangroves are highly sensitive to these herbicides. Overspray and heavy rainfall following application can wash the herbicide off the trunk and damage non-target vegetation and/or enter surface water. For these reasons, use of injection techniques instead of basal bark or girdle application is preferred for exotics control in mangrove areas.

CONCLUSIONS

Much of the uplands within the preserve, including the location of the proposed interpretive center complex, was altered during development earlier in this century. The mangrove areas were draglined for mosquito control during the 1960S. These activities have resulted in disturbance of native communities and invasion of exotics in many parts of the preserve. Today, the County is working toward restoration of these areas through the exotic plant control activities and prescribed burn programs. The proposed improvements, while increasing the public use value of the preserve, also offer opportunities for ecosystem improvement as well. Recommendations for siting have the goals of minimizing impact to natural communities and incorporating restoration and preservation to the maximum extent possible.

ACKNOWLEDGEMENTS

The authors wish to acknowledge the special assistance from Mr. Keith Thompson, Pinellas County Park Department Weedon Island Preserve Supervisor. Thanks are also extended to Ms. Kim Marler and Mr. Robert McConnell of Parsons ES for their work in analysis of biological resources and to Dr. Petar Glumac, Parsons ES, for his work on historical and archaeological resources.

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HYDROLOGIC RESTORATION OF THE DuPUIS RESERVE

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ABSTRACT

The 8,800 hectare (22,000 ac.) Whitebelt Ranch (DuPuis Reserve), which straddles Martin and Palm Beach Counties, was acquired in 1986 by the South Florida Water Management District through the Save Our Rivers program. DuPuis Reserve has 11 km of common boundary with the 22,800 ha. (57,000 ac.) J.W. Corbett Wildlife Management Area, which is owned and managed by the Florida Game and Fresh Water Fish Commission. District management of DuPuis began in late 1987. Inspections of the ranch and comparisons with 1940 aerial photography revealed that the property had been overdrained.

An environmental assessment completed in 1991 revealed that the most significant impacts had affected wet flatwoods, basin marsh, and wet prairie. A three phase hydrologic restoration plan was developed:

Phase I Installation of earthen plugs in the ditches and swales

Phase II Reestablishment of sheetflow between Corbett WMA and DuPuis

Phase III Construction of a 13 km long earthen levee with water control structures to reflood 1,000 ha. of drained Everglades marsh

INTRODUCTION

The Florida Legislature enacted a bill in 1991 to provide funding for the acquisition and management of water resource and environmentally sensitive lands. Formally known as the Florida Resource Rivers Act, this legislation is commonly called the Save Our Rivers (SOR) program, and is operated by the state's five water management districts.

The South Florida Water Management District (SFWMD) acquired the 8,800 ha. Whitebelt Ranch in 1986 with Save Our Rivers monies. After purchase the name was changed to the DuPuis Reserve. The property is located in western Martin and Palm Beach Counties and is the westernmost link of a proposed 44,000 ha. natural area corridor that extends from Lake Okeechobee to the Atlantic Ocean (Figure 1). DuPuis lies three miles east of Lake Okeechobee, and is bordered on the east by the 22,800 ha. J.W. Corbett Wildlife Management Area, which is owned and managed by the Florida Game and Fresh Water Fish Commission. To the

Major Natural Areas of Palm Beach and Martin Counties

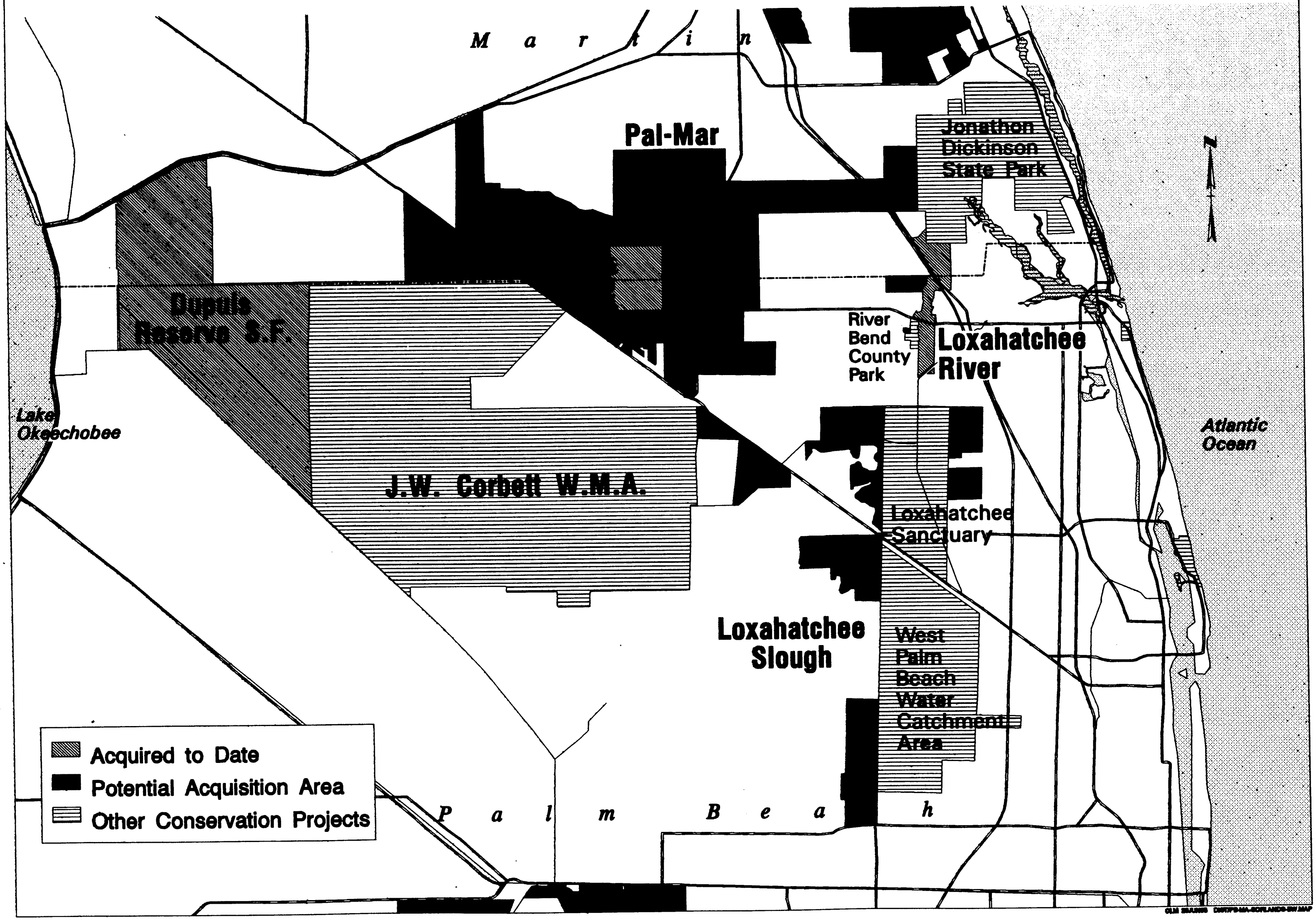

Figure 1

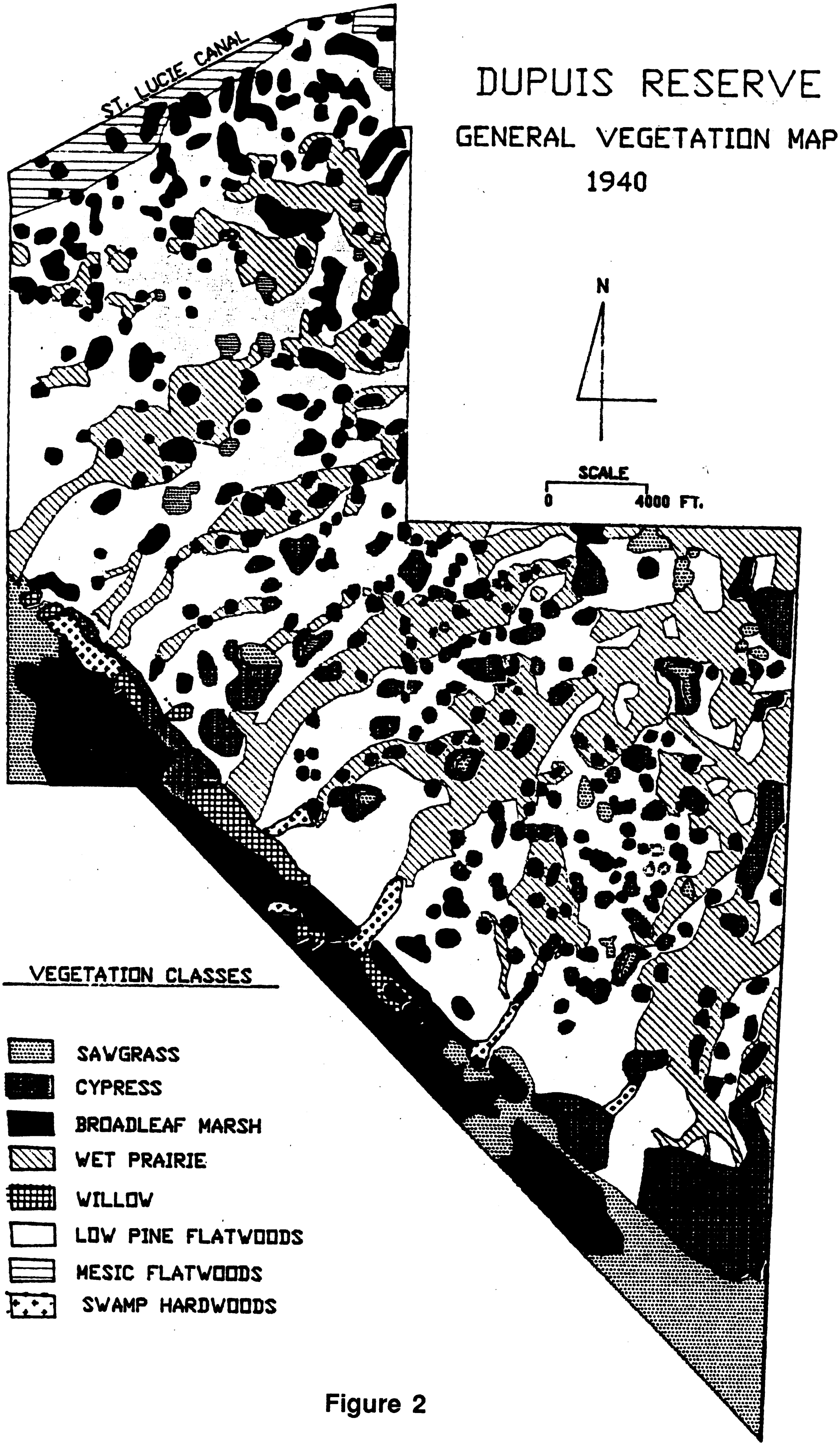
DUPUIS RESERVE
GENERAL VEGETATION MAP
1940
ST. LUCIE CANAL
N
SCALE
0
4000 FT.
VEGETATION CLASSES
SAWGRASS
CYPRESS
BROADLEAF MARSH
WET PRAIRIE
WILLOW
LOW PINE FLATWOODS
MESIC FLATWOODS
SWAMP HARDWOODS

Figure 2

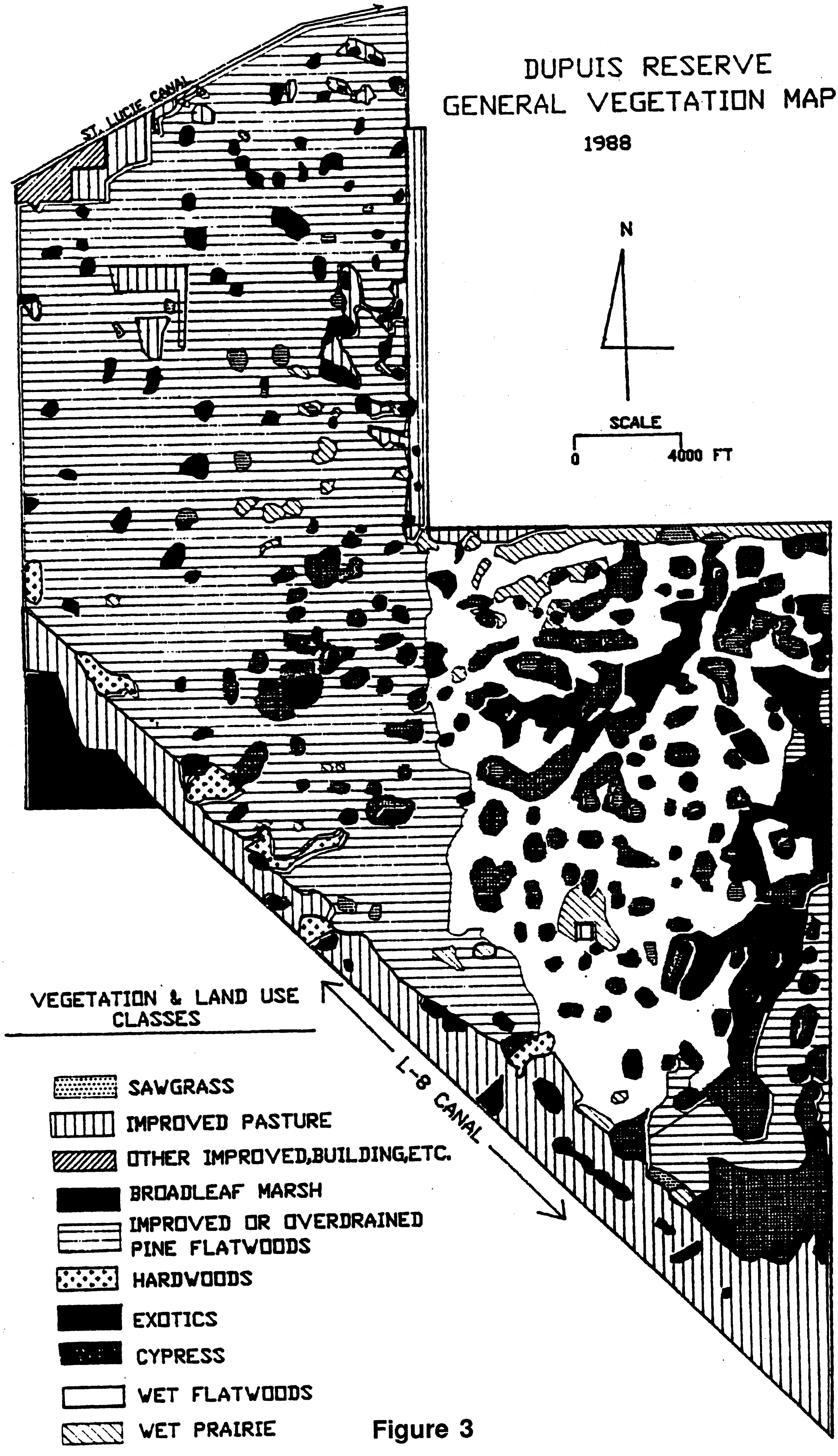
DUPUIS RESERVE
GENERAL VEGETATION MAP
1988
N
SCALE
0
4000 FT
ST. LUCIE CANAL
L-8 CANAL
VEGETATION & LAND USE CLASSES
SAWGRASS
IMPROVED PASTURE
OTHER IMPROVED, BUILDING, ETC.
BROADLEAF MARSH
IMPROVED OR OVERDRAINED PINE FLATWOODS
HARDWOODS
EXOTICS
CYPRESS
WET FLATWOODS
WET PRAIRIE
SCRUB CYPRESS

Figure 3

northeast lies Pal-Mar, a former land sales development, consisting of 10,000 ha. (25,000 ac.) of wet flatwoods, interspersed with extensive wet prairie and basin marsh systems. In 1994, SFWMD and Martin County took the first step in protecting Pal-Mar with a 720 ha. purchase. Pal-Mar connects with Jonathan Dickinson State Park, a 4,000 ha. mosaic of scrub, flatwoods, and fresh and saltwater wetlands which extends to the coast.

Actual management of DuPuis Reserve by the District began in late 1987. Inspections of the ranch and comparisons of existing conditions with aerial photography from 1940 revealed that the property had been over-drained. In 1991, an environmental assessment was completed (David, 1991) and included general vegetation maps which were prepared from 1940 and 1988 aerial photography (Figures 2 and 3) (NOTE: Original figures were prepared in color; color copies may be obtained by contacting author). Photointerpretation of these maps revealed several significant changes in wetland communities between 1940 and 1989. The most noticeable changes were decreases in wet flatwoods, basin marsh, and wet prairie, and a dramatic increase in area covered by mesic flatwoods.

Community Type	1940	1989	Change
Wet Flatwoods	3,160 ha	120 ha	- 96
Basin Marsh	1,800 ha	560 ha	- 69
Wet Prairie	2,120 ha	200 ha	- 91
Mesic Flatwoods	160 ha	1,520 ha	

Drainage of the property had been undertaken over a number of years to increase and improve the forage for cattle and sheep grazing. In 1940, broad wet prairie sloughs extended across the property, as the land drained to the southwest. The northern portion was comprised of isolated depression marshes interspersed among wet flatwoods. By 1988, most of the wet prairies had been converted to improved pasture. At its peak as an operating ranch, Whitebelt supported more than 2,500 head of cattle, 2,000 sheep, and 1,000 goats. Basin marshes, wet prairies, and dome swamps were connected by series of shallow swales and ditches which facilitated drainage.

METHODS

In 1989, the SFWMD (David, 1991) developed a three-phase hydrologic restoration plan (Figure 4):

Phase 1

Installation of earthen plugs in ditches and swales for the purpose of re-inundating isolated interior wetlands. In 1990-1991, 41 ditch plugs were Installation of 41 earthen ditch pugs in 1990-1991 have reflooded more than 1600 ha. of wet prairies and wet flatwoods.

In 1990 and 1991, 41 earthen ditch plugs were installed to stop the drainage of isolated wetlands. This was accomplished using field personnel and equipment from the SFWMD West Palm Beach Field Station and Land Stewardship Division.

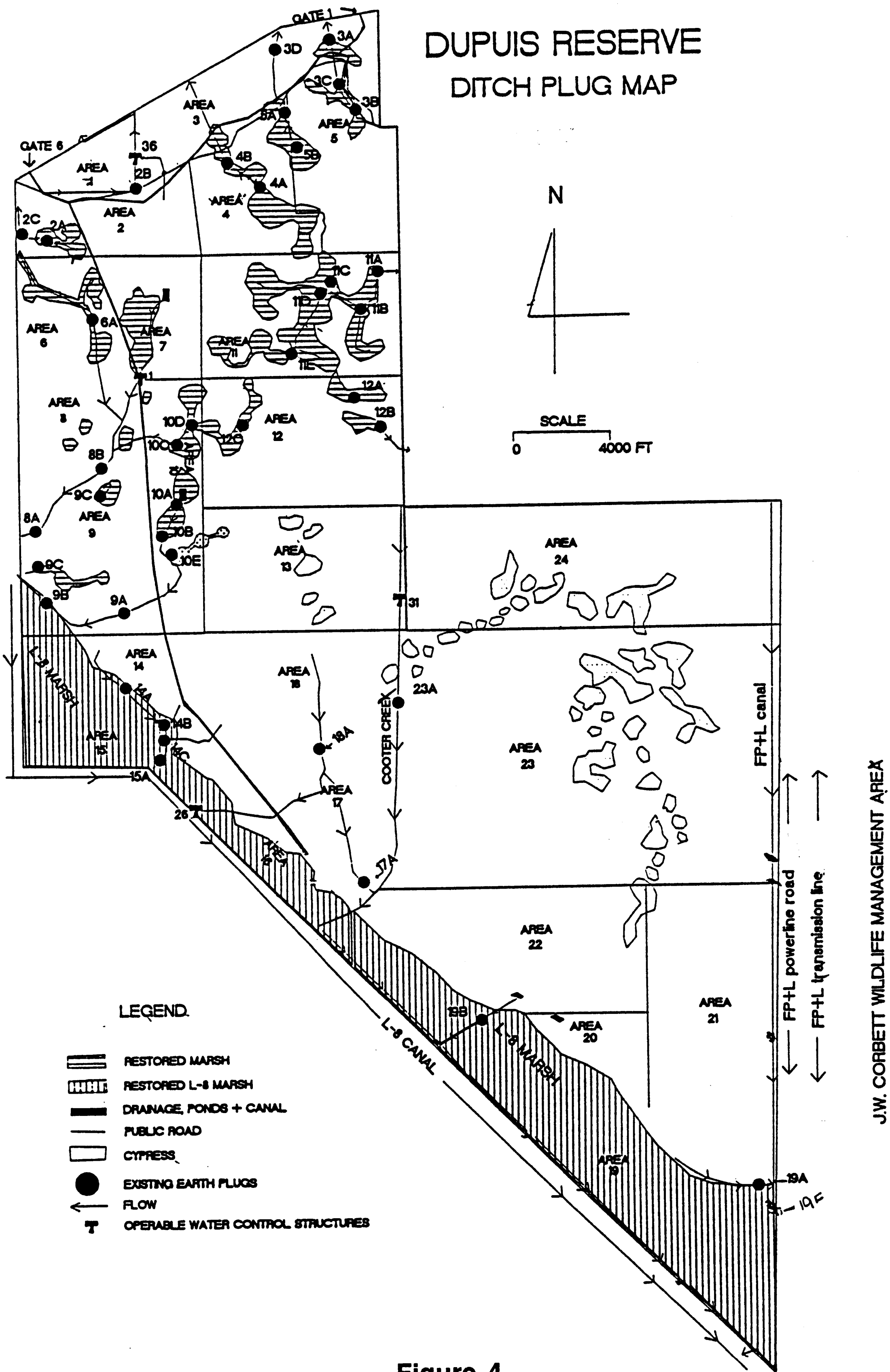
DUPUIS RESERVE
DITCH PLUG MAP
N
SCALE
0
4000 FT
LEGEND
RESTORED MARSH
RESTORED L-8 MARSH
DRAINAGE, PONDS + CANAL
PUBLIC ROAD
CYPRESS
EXISTING EARTH PLUGS
FLOW
OPERABLE WATER CONTROL STRUCTURES
GATE 1
GATE 6
L-8 CANAL
L-8 MARSH
COOTER CREEK
FP+L canal
FP+L powerline road
FP+L transmission line
J.W. CORBETT WILDLIFE MANAGEMENT AREA

Figure 4

In addition to the transects and water level recorders, two permanent photo points were established in 1992, one each in the L-8 Marsh and a plugged isolated wetland system. The photo points provide pictorial documentation of the restoration program.

Phase 2

Reestablishment of sheetflow between Corbett Wildlife Management Area and DuPuis. Mitigation funds from Florida Power and Light were used to construct two "Geoweb" swales which are equalizing flows between the two areas. Two additional swales will be constructed following Phase 3.

The second phase of hydrologic restoration was to reestablish sheetflow to the L-8 marsh, which historically occurred as runoff from the western 5,600 ha. of Corbett WMA. This flow was severed by a canal excavated in 1978 along the Corbett-DuPuis boundary. The FPL canal diverted flow directly to the L-8 Canal, which lowered ground water levels along the eastern boundary of DuPuis. This resulted in a much shortened hydroperiod in the adjacent cypress domes and sloughs on the DuPuis side encouraged the invasion of upland species, such as wax myrtle (Myrica cerifica), as well as exotics, including cajeput (Melaleuca quinquenervia) and Japanese climbing fern (Lygodium palmatum).

In 1991, the District designed, permitted through DEP, and constructed with District field personnel, three earthen plugs in the FPL canal. The plugs were installed to stop the ground water drawdown in adjacent wetlands and to prepare the site for a future hydraulic connection with the Corbett Area.

A plan was developed in 1992, as mitigation for wetland impacts associated with a new FPL transmission line, to construct four swale crossings in the powerline road, which would equalize the flow of water between the two natural areas.

In 1990, the Florida Game and Fresh Water Fish Commission (GFC) improved the levee along the Corbett portion of L-8, and installed flashboard riser water control structures. The work was paid for as mitigation for the Palm Beach County Solid Waste Authority resource recovery plant.

Phase 3

Construction of a 13 km (8 mi.) long earthen levee with water control structures to reflood 1,000 ha. of drained Everglades marsh. Construction began in October, 1994, and will be complete in October, 1995. It is expected that the entire $1.5 million construction bill will eventually be paid for with mitigation funds, which are being accepted to offset project costs.

Prior to the excavation of the L-8 Canal, the southwestern boundary of the DuPuis Reserve was the northernmost edge of the Everglades. Sawgrass marshes extended from the eastern shore of Lake Okeechobee to the pine flatwoods along DuPuis' western edge. After the canal was excavated, the 1,000 ha. marsh was drained and planted in pasture grasses. It became the most valuable winter pasture on Whitebelt Ranch.

The third phase of the DuPuis hydrologic restoration involves the construction of a seven mile long earthen levee along the DuPuis side of the L-8 Canal. Included in this design would be a series of operable water control structures. A great deal of time has gone into the planning, hydrologic modelling, and preliminary design for such an impoundment.

Construction of the levee along the L-8 canal began in October, 1994. A continuous internal borrow ditch is being excavated. The levee is designed to have a top elevation of 7.3 m NGVD (24.0'). Three sets of flashboard risers will be installed along the length of the levee to control marsh water elevations at 5.8 m (19.0'), which is the same control elevation used in the Corbett portion of the L-8 marsh. The water control structures are designed such that the marsh can be drained, if it becomes necessary for management. The District has accepted financial contributions to satisfy mitigation requirements for wetland impacts on projects elsewhere in Palm Beach County. These payments have included reimbursement for land costs, construction of the levee, and long-term maintenance. It is anticipated that the entire $1,500,000 cost of construction will eventually be offset through the mitigation process.

A monitoring program was developed to evaluate the effectiveness of the restoration efforts. In 1988, two vegetation transects were established in the L-8 Marsh, and baseline data was collected in anticipation of reflooding the marsh. In 1990, two additional transects were established in drained isolated marshes to monitor the effectiveness of ditch plugs. In 1990, four water level recorders were installed, one near each vegetation transect. Since that time, the transects have been sampled annually. Information from the water level recorders is collected quarterly and loaded into the District's data base.

RESULTS

Phase 1

Ditch plugs have significantly changed the hydrologic condition by reflooding nearly 4,000 acres of wet prairies, broadleaf marshes, and low pine flatwoods. Slash pine and wax myrtle, which had been invading many former wetlands, are dying. Much of DuPuis is staying wet for extended periods of time. Normal and above normal wet seasons from 1991 to 1994 kept the reserve wet for extended periods of time.

The ditch plugs have greatly reduced the rapid flow of water off the property, and are dramatically changing the character of the former cattle and sheep ranch bringing it closer to its pre-drainage condition.

Phase 2

The Corbett improvements have reflooded all of that portion of the L-8 Marsh. However, the limited discharge capacity from the water control structures has caused the marsh, and large areas of adjacent flatwoods, to stay wet much longer than normal. Flooding has been the worst near the FPL canal/L-8 Canal corner.

Water backs up along the elevated FPL powerline road. Much lower water stages in the FPL and L-8 Canals have resulted in several major washouts in the FPL road.

To correct this problem, in September, 1993, a contractor for FPL constructed the first two-30 m long "Geoweb" swales in the powerline road. Geoweb is a plastic cellular confinement material which provides a stable roadbed in wet conditions, and has been used by FPL and other utility companies under similar conditions. This material was selected because culverts are effective at equalizing flows where substantial head differences exist. However, under low head conditions (typical for sheetflow) culverts only pass a minimal amount of water. Low velocity flows also cause culverts to clog very quickly, reducing the flow even more. The swale crossings move much more water and do not clog. The road is traveled by either large utility service trucks or managing agency 4-wheel drive vehicles, which are not affected by maximum water depths of 30-45 cm.

The remaining two swales will be installed after the L-8 Marsh levee is constructed. We have already found that one of the benefits of the ditch plugs and swales has been to reflood a cypress dome adjacent to the canal and drown out the Lygodium.

Phase 3

Reflooding the L-8 Marsh will likely have even more dramatic effects on DuPuis than the first two phases. Backwater effects from the flooded marsh will probably extend well inland, reducing even more the ability of the land to drain. Low flatwoods and wetlands will stay wetter longer. More dieoff of terrestrial vegetation can be expected.

The average width of the marsh is approximately 1.6 km, and ground elevations vary from 6-6.4 m NGVD (20-21') near the eastern treeline, to 4.6 m (15') near the toe of the new levee. The elevation gradient will create a range of community types, from shallow wet prairies to open water sloughs. The deep marshes and open water sloughs will provide habitat for fish and waterfowl, and possibly Everglades kites. Wet prairies along the shallow fringe will become more extensive, providing feeding areas for wading birds. Public use benefits will include a hiking trail along the levee, and the possibility of an observation tower and/or boardwalk across the flooded marsh.

CONCLUSION

Hydroperiod enhancement in 1,600 ha. of isolated interior wetlands, reestablishing sheetflow conditions between Corbett and DuPuis, and reflooding 1,000 ha. of remnant Everglades marsh will have major positive impacts on one of south Florida's premier natural areas. The Save Our Rivers legislative wording specifically states that lands shall be managed "in such a way as to restore and protect their natural state and condition." For lands acquired for their water resource values, restoration of those values is our most important task.

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STANDARDIZATION OF MONITORING METHODOLOGY FOR WETLAND MITIGATION PROJECTS

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ABSTRACT

Since the mid-1980's, wetland mitigation has become an integral part of the permitting process under which wetland loss by permitted encroachment is compensated at a 1:1 or higher ratio of created, restored, or enhanced wetland for each 1 acre of natural wetland lost. Over the past 10 years, strategies to improve mitigation monitoring plans for wetland creation, restoration, and enhancement projects have evolved. Success criteria for wetland mitigation projects are varied. A wetland mitigation monitoring plan should address how success criteria will be evaluated and how the results will be used to direct changes in the wetland management plan.

Ecological monitoring of a created, restored, or enhanced wetland site plays a vital role in determining and measuring the success or failure of a wetland mitigation plan. All regulatory agencies require that a wetland mitigation monitoring plan be submitted for approval. However, consistency among the agencies is needed to provide a standardized methodology for monitoring wetlands for mitigation projects.

A standardized wetland mitigation monitoring plan is an important tool for evaluating success and providing guidance to adjust or modify wetland conditions. This paper begins to identify basic elements necessary to provide a standardized wetland mitigation monitoring plan that will be acceptable to the regulatory agencies and enhance mitigation success. A joint effort by all agencies that deal with wetland mitigation monitoring, and an agency that focuses on measuring the success of the project, will enhance the permit review process.

INTRODUCTION

Florida's wetland regulations have recently been modified in order to streamline the permit review process of the Water Management Districts (WMD's) and the Florida Department of Environmental Protection (FDEP). The review is also coordinated with the U.S. Army Corps of Engineers (COE). Permit review would be enhanced by using one set of success criteria and monitoring requirements for these regulatory agencies.

Regulatory requirements to permit wetland activities need to include appropriate compensation for unavoidable wetland loss. Mitigation has been used for over 10 years to compensate for wetland loss (Lee, 1992). As part of the mitigation process, periodic measurements must be made to ensure that a created, enhanced, or restored wetland meets the goals and objectives of the success criteria established in the permit. The monitoring plan should be designed to measure specific characteristics of the wetland and should specify the frequency of the measurements. Results from the monitoring program can be used to direct changes in the wetland management plan as necessary to achieve successful wetland mitigation. The purpose of this paper is to begin to identify standardized monitoring methodology for directing and assessing the success of creating, enhancing, or restoring wetlands.

There is variability in wetland types and in the major wetland components. Therefore, adequate sampling for each type of wetland may vary depending upon soil conditions, the hydrologic regime, and the vegetative community. Careful planning for wetland monitoring is important. Since regulatory agencies will establish success criteria for wetland mitigation in the permit, coordination with the regulatory agencies in preparing the wetland mitigation monitoring plan is very important.

SUCCESS CRITERIA

Success can be defined as "achieving an established goal(s)." Examples of specific mitigation success criteria were taken from FDEP, WMD's, and COE permit conditions, as follows:

A successful wetland must meet the following criteria for creation or restoration for a period of at least one growing season, without intervention in the form of irrigation, dewatering, removal of undesirable vegetation, or replanting of desirable vegetation.

1. Success will be evaluated annually based on monitoring for at least 5 years for restored wetlands, and 3 years for created wetlands. Final evaluation will be in terms of a jurisdictional determination.

2. Nuisance and exotic species in the wetlands will be limited to 5 percent or less of total cover.

3. Within the wetlands, at least 80 percent of the ground shall be covered with non-nuisance, non-exotic vegetation. At least 80 percent of this vegetation will be listed wetland plant species reproducing naturally.

4. Forested wetlands will average at least 400 trees per acre above the herbaceous stratum. Tree cover shall exceed 33 percent of the total area and in no area of one acre in size shall the tree cover be less than 20 percent total cover. Cover measurement shall be restricted to native species listed in the planting plan or listed wetland trees that occur in the drainage basin. Forested wetland will have at least 30 consecutive days of inundation for five consecutive years.

A study was conducted to determine the number of years required to attain 33 percent canopy cover using 1.2 liter (1-gallon), 3.6 liter (3-gallon), and 8.4 liter (7-gallon) container sized trees at various planting densities (BRA Inc., 1989). The results indicated that the 33 percent canopy cover was attained within 3.5 years using the 8.4 liter (7-gallon) sized containers at a planting density of 2,470 trees per hectare (1,000 trees per acre) (Table 1). The longest time period to attain the 33 percent canopy cover was 8.5 years using the 3.6 liter (3-gallon) container size at a planting density of 988 trees per hectare (400 trees per acre). This suggests that a greater planting density than required by permit will be necessary to ensure success.

Table 1. Number of years required to attain 33 percent canopy cover using 1-gallon, 3-gallon, and 7-gallon container size trees at various planting densities.

Initial Container Size		Planting Density (Trees/Hectare)		Years to Attain 33 percent Cover
Gallon	Liter	Trees/Ac	Trees/Ha	
1	1.2	400	988	9
1	1.2	600	1482	7
1	1.2	800	1976	6
1	1.2	900	2223	5.5
1	1.2	1000	2470	5
1	1.2	1100	2717	5
3	3.6	400	988	8.5
3	3.6	600	1482	6.5
3	3.6	800	1976	5.5
3	3.6	1000	2470	4.5
7	8.4	400	988	7.5
7	8.4	600	1482	5.5
7	8.4	800	1976	4.5
7	8.4	1000	2470	3.5

Estimated cost for installation, maintenance, and monitoring are not included. The type of soils, hydroperiod, and plant installation factors may vary these estimates (BRA, Inc., 1989).

Success criteria can also include qualitative comparable observations over time. For example, photographs of each community taken at fixed points will indicate qualitative changes and conditions in plant community composition over time.

Wetland Mitigation Monitoring Plan

"Ecological monitoring is the acquisition of information to assess the status and trends of the structure and function of biological populations and communities, and their habitat, and larger-scale ecosystems (i.e., landscapes) over time, for the purpose of assessing and directing management activities" (Gordon et al., 1995).

Ecological monitoring includes the following characteristics:

1. Identification of the population that is being monitored,
2. Establishment of parameters and methodology,
3. Establishment of the frequency of measurement,
4. Development of a maintenance program to control nuisance and exotic species, and
5. Identification and adjustment of management goals as needed.

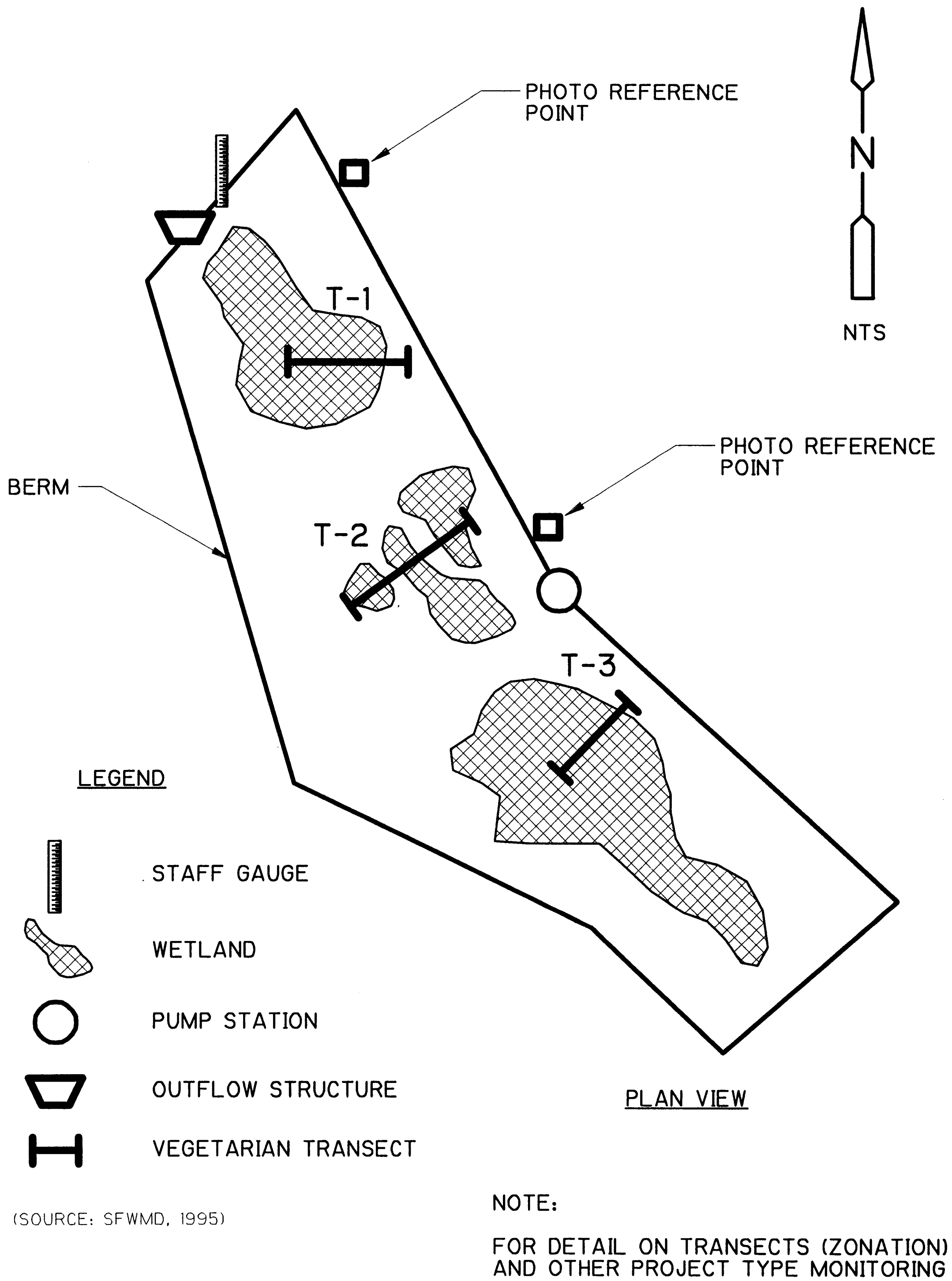
A wetland mitigation monitoring plan should include these characteristics and specifically address how the success criteria will be evaluated and how the results will be used to direct changes in the wetland management plan. Once the success criteria are established, assessment can be made through sampling and measurement. A cost-effective technique that combines a minimum number of observations taken for a given measure (i.e., cover, density, and biomass) with an evaluation of the variance of the measurement, is an important consideration. This technique should include accurate and repeatable results.

A wetland mitigation monitoring plan should also consider the following vegetation sampling methods. Two methods are generally used. The line-intercept method and the quadrat sampling method. Baseline transects for both methods are established in the wetland (Figure 1). Several transects may be required in large wetlands. Each wetland is sampled in the upper transitional zone, lower transitional zone, and in the submergent zone (Figure 2). This could include both emergent and submergent vegetation, if present (Neilsen, 1995).

It has been shown that the line-intercept method can be used in both heterogeneous or homogeneous communities to provide meaningful results with little effort (Neilsen, 1995). This method is used to determine numerical abundance, frequency, percent cover, and other characteristics (Figure 3).

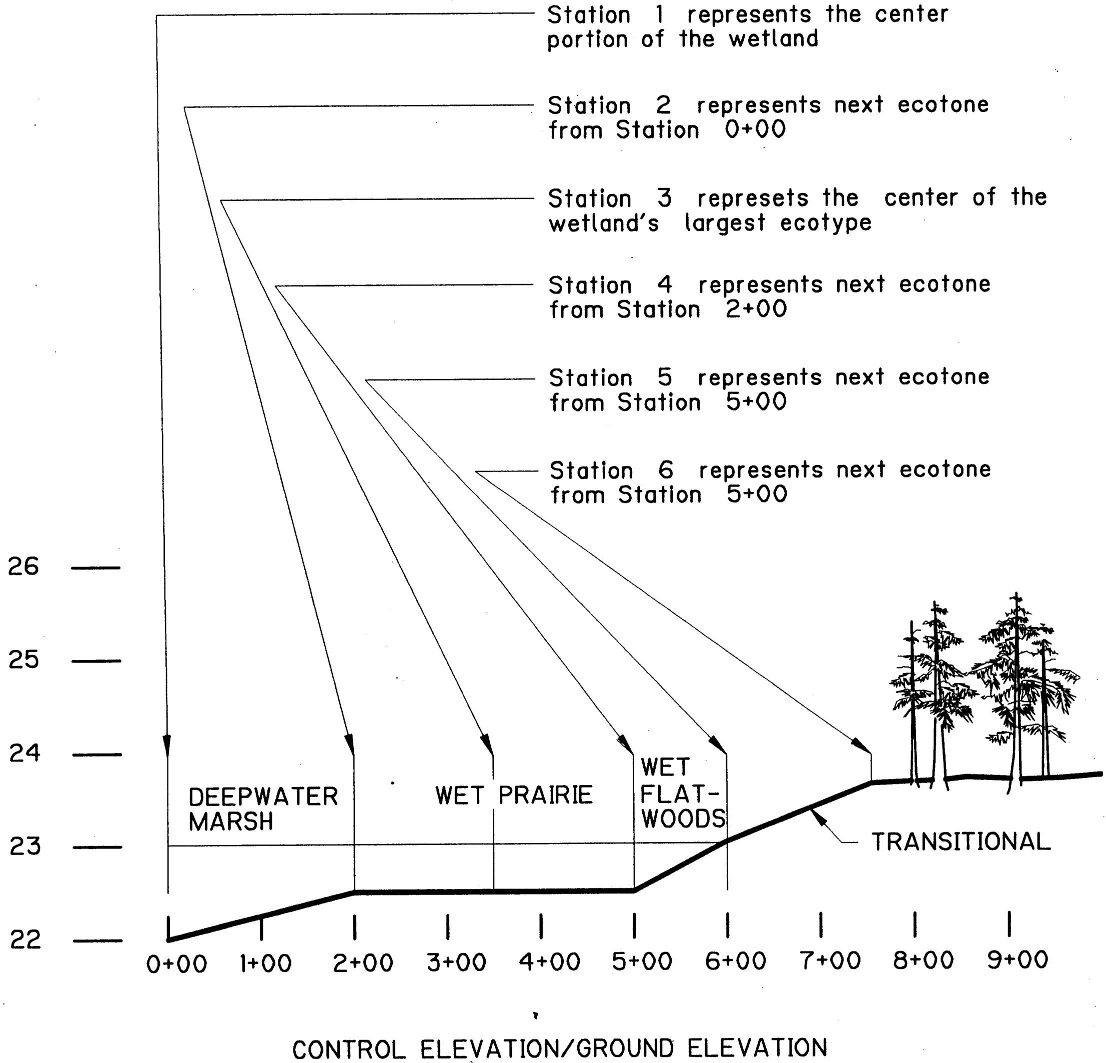
For the line-intercept method sampling will occur in transect segments for each stratum; for the canopy substitute one 10-meter segment; for each subcanopy quadrat, substitute one 2-meter segment.

The line-intercept method is used to determine aerial coverage along the transect. The length of each intercepted plant is measured (Figure 3). The length of the transect and the total length intercepted by vegetation are used to estimate percent cover for all strata minus bare ground. Bare ground is only considered when plant species of crown cover do not occur in any strata (Clapham, 1932). Percent herbaceous cover, percent tree survival, dominance, frequency, and percent cover of nuisance and exotic species can be recorded (by stratum).



TYPICAL MONITORING PLAN

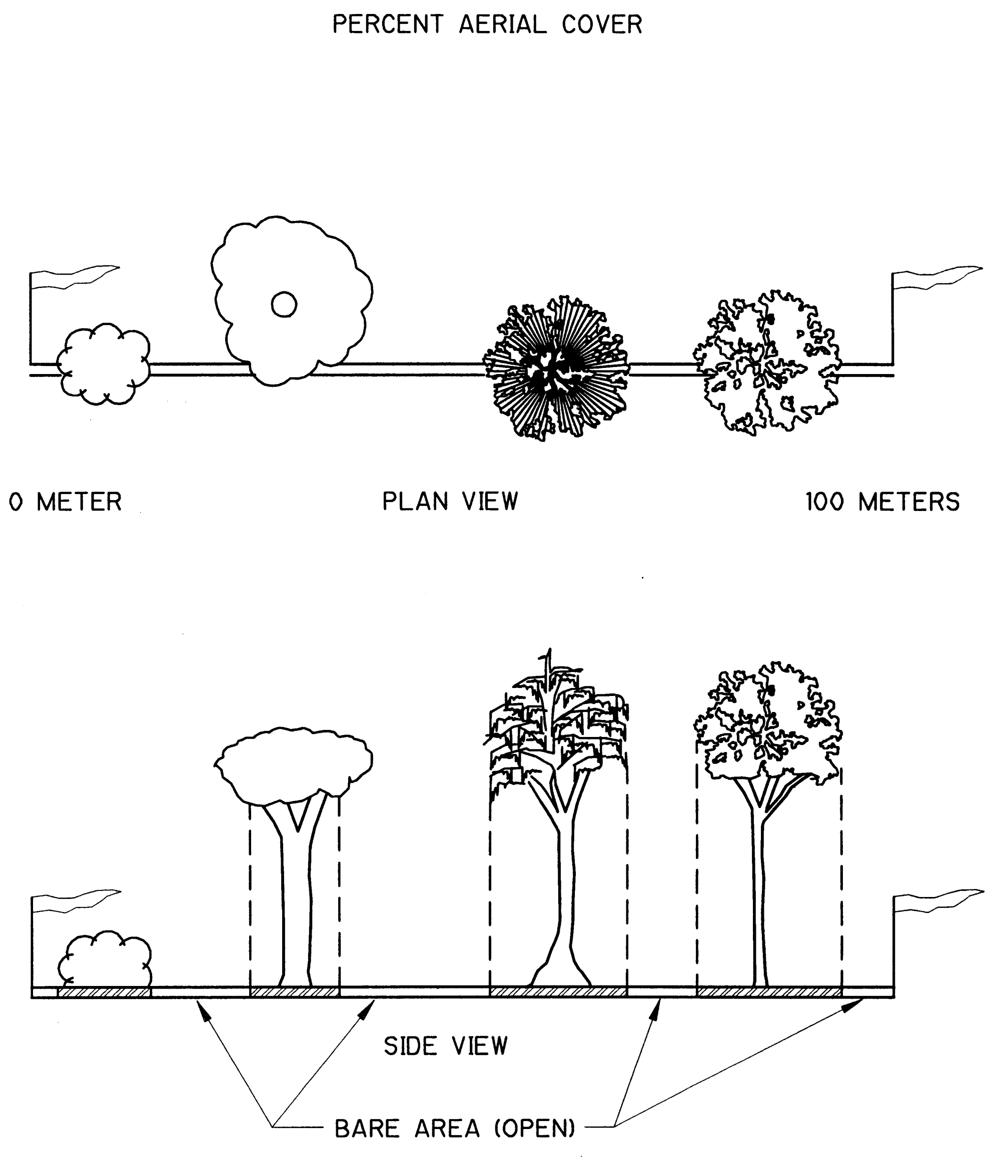
Figure 1



(SOURCE: SFWMD, 1995)

TRANSECT/ELEVATION CROSS SECTION RELATIVE TO CONTROL ELEVATION OF ISOLATED WETLAND

Figure 2



LINE-INTERCEPT METHOD

Figure 3

Sampling should occur within each ecotone or change of vegetation (e.g., upper transition zone, lower transition zone, and the submerged zone). In this way, changes in vegetation cover, dominance, and density can be observed over time.

Quadrat sampling is more widely used in heterogenous communities and depends upon the type of wetland community and stage of succession (Neilsen, 1995). Typically, a transect is randomly placed across the greatest extent of a large wetland, or several transects are placed perpendicular to an elongated wetland. The quadrats are evenly spaced along these transects. Daubenmire (1968) suggests that the transects be laid out in a systematic fashion to ensure that a broad and even coverage be evaluated rather than randomly as preferred by some researchers.

The number of quadrats needed within a site can be determined by the size of the wetland. The number of quadrats along a transect may include a minimum of 1 transect per site for a site less than or equal to 0.2-ha (0.5 acres). For a site within an area between 0.2 ha (0.5 acres) to 0.8 ha (2.0 acres), a minimum of 2 quadrats along a transect are needed. For a site with an area larger than 0.8-ha (2.0 acres), Figure 4 can be used.

The nested plot size used to sample ground cover, subcanopy, and canopy is shown in Figure 5. For quadrat sampling, each transect will include at a minimum:

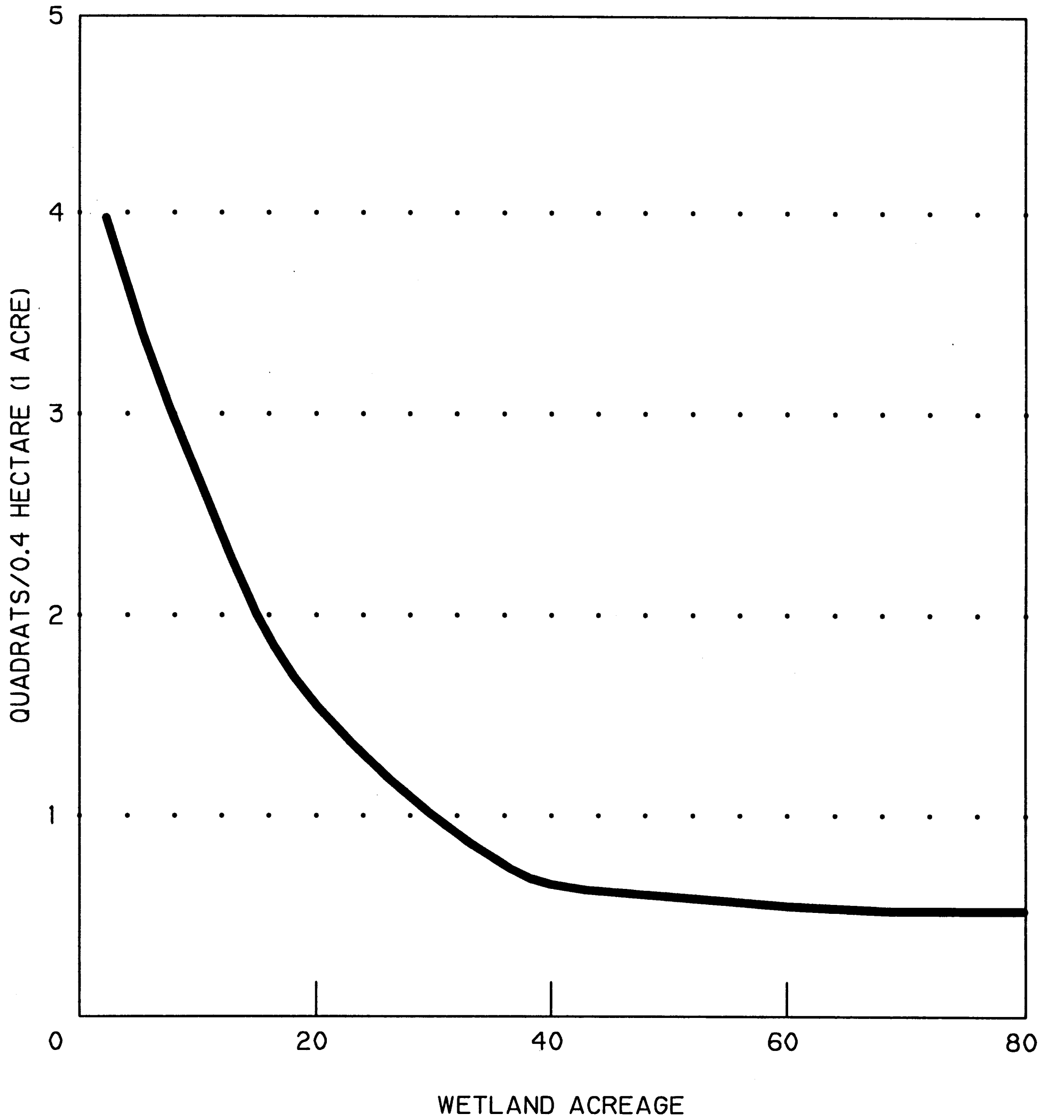
a. For groundcover - Two 1-square-meter quadrats per zone
b. For subcanopy - One 16-square-meter quadrat per zone
c. For canopy - One 100-square-meter quadrat zone

For example, a small wetland with 3 zones will have a total of 3 canopy quadrats, 3 subcanopy quadrats, and 6 groundcover quadrats.

The seasonal time frame in which wetland monitoring occurs provides important comparable and measurable characteristics of wetland species. Bonham (1989) states that variation in measures of vegetation are affected by species life form, species composition, seasonality, edaphic (soil) sources, and human and animal influences. Growing season (phenological stages) and stage of development (successional stage) are also considerations to determine sample time. To assess success criteria, the WMD's and the FDEP recommend sampling semi-annually, during the dry season (October - April) and the wet season (May - September).

To easily identify the location of quadrats along transects that are used for repeated sampling, a 1-foot long rebar is placed vertically in the ground at one corner of each quadrat (i.e., the northeast corner, as a point of reference) and covered with PVC pipe. Additionally, treated wooden stakes (lathes) can be placed at the three remaining corners of the quadrat to accurately locate the sample plot.

In areas of repeated sampling, care must be taken to avoid destruction of the vegetation in the quadrat by creating a permanent foot trail. Animals may use these paths or foot trails as well to migrate and forage. In some cases, stormwater or shallow wetland surface waters may sheetflow across these paths creating small intermittent drainage ways.



NOTE:

1. DIVIDE BY 2 FOR SUBCANOPY OR CANOPY QUADRATS
2. ROUND TO NEAREST WHOLE NUMBER
3. CHART REFERS TO ACREAGE OF EACH INDIVIDUAL WETLAND, NO CUMULATIVE ACREAGE

(SOURCE: NIELSEN, 1995)

GROUND QUADRATS REQUIRED PER 0.4 HECTARE (1 ACRE)

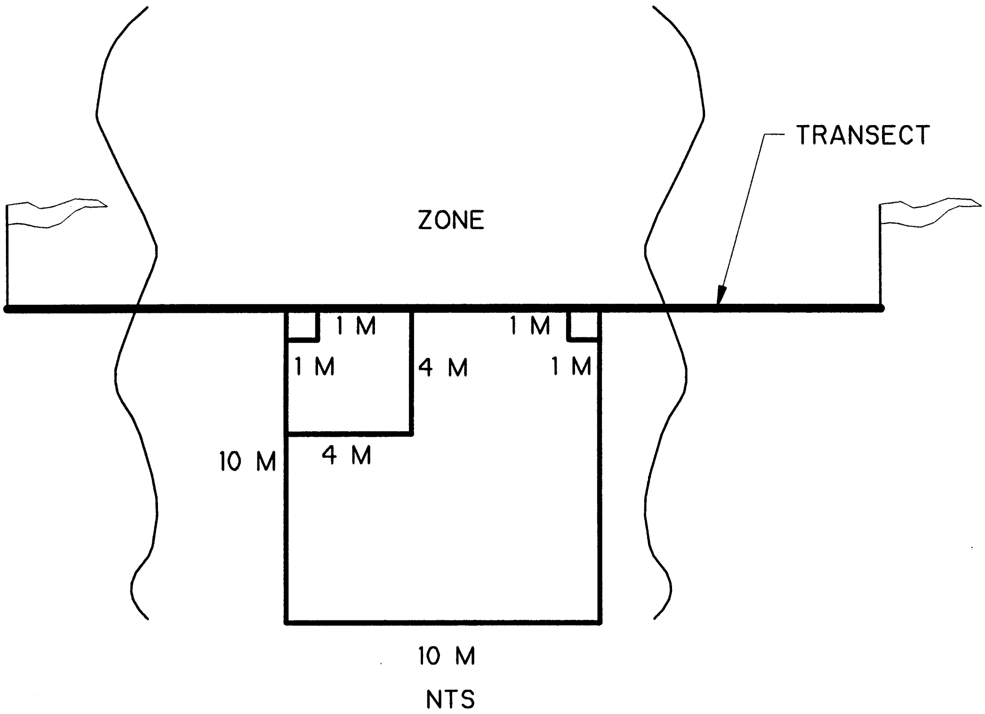
Figure 4

(2) 1 METER SQUARE - GROUNDCOVER

(1) 4 METER SQUARE - SHRUB STRATUM

(1) 10 METER SQUARE - CANOPY STRATUM

WHERE MORE THAN ONE STRATUM OF VEGETATION OCCURS WITHIN A ZONE, NESTED QUADRATS WILL BE USED AS SHOWN BELOW.



QUADRAT SAMPLING

Figure 5

Observations, numbers, and types of wildlife that use the wetland for foraging and nesting should be documented. No quantitative measurements have been required other than noting animal tracks, scat, rooting activities, shrubs, and nesting sites. Reptiles, fish, birds and mammals observed or heard, and the location of their nesting sites should also be documented. The South Florida Water Management District is looking at evaluating fish and macroinvertebrate communities to represent a quantitative (relative abundance) and qualitative (presence) measure of the aquatic fauna of the created, restored, or enhanced wetland.

DISCUSSION

A standardized methodology for a wetland mitigation monitoring plan can be developed by including a set of parameters that address the goals and objectives of specific success criteria. Input from the various agencies in developing guidelines for monitoring mitigation has not as yet been provided. As a result, monitoring reports are provided with incomplete or inaccurate data that do not allow for proper evaluation of success criteria. Standardized methodologies to perform a wetland mitigation project and prepare a complete monitoring report may help regulators, and consultants and their clients obtain successful mitigation. A working group from the Central Florida Association of Environmental Professionals (CFAEP) is preparing draft guidelines for a standardized monitoring methodology that will be submitted to the regulatory agencies for review. A draft will be forthcoming to review and comment by the regulatory agencies.

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PRELIMINARY STUDIES OF SAWGRASS SEEDLINGS FOR EVERGLADES RESTORATION

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ABSTRACT

Understanding seedling establishment of sawgrass (Cladium jamaicense Crantz) is crucial for restoring the Everglades. In this experiment, we explored nutrient factors controlling survival of seedlings in the early stage of development and various techniques for seedling transplanting. Nutrient pulse treatments were applied approximately four months after transplanting. Photosynthetic rates were measured six weeks after treatments and prior to plant harvesting. A range of 61% to 95% survivorship was observed when seedlings were transplanted into a moist commercial potting soil mixture without standing water. In general, it was found: 1.) the biomass of the plants receiving nutrient pulses was significantly higher than plants with no nutrient addition (control); 2.) photosynthetic rates of pulsed plants were significantly greater than control plants; 3.) no differences in stomatal conductance were observed between the two nutrient treatments; and 4.) instantaneous leaf water use efficiency significantly increased with nutrient pulses.

INTRODUCTION

Sawgrass (Cladium jamaicense Crantz) is a large clonal freshwater wetland species that plays an important role in controlling the structure and function of the Everglades ecosystem (Davis, 1994). It is recognized as the dominant Everglades species and is also found in fresh water wetlands throughout subtropical North and Central America. In recent years, cattail (Typha domingensis Pers.) has invaded 4,800 ha previously occupied by sawgrass in Water Conservation Area 2A in the northern Everglades (Davis, 1994). To restore the Everglades and reestablish sawgrass vegetation, we must increase our understanding of sawgrass seed germination and seedling establishment. Unfortunately, knowledge about the growth and ecophysiology of sawgrass seedlings is very limited (Steward and Ornes, 1975).

Although limited to growth in wetland acres, sawgrass possesses leaves with several xeromorphic characteristics, such as thick cuticle, numerous bands of lignified fibers, marginal spines, and a waxy bloom on the leaf surface. This suggests a restricted water-supply for developing leaves (Conway, 1940). Sawgrass also has low nutrient requirements for growth (Alexander, 1971). However, it is capable of maintaining nearly monospecific stands under a range of nutrient concentrations (Davis, 1994; Steward and Ornes, 1975) and grows mostly only a autochthonous peat. Although sawgrass can reproduce sexually, seed propagation does not appear to be the major mechanism maintaining the established stands in the Everglades (Alexander, 1971). It appears that seed germination occurs under very restricted conditions. This experiment is part of a long-term program to restore the Everglades as required by State and Federal Legislative actions. This experiment explores techniques for transplanting sawgrass seedlings and for enhancing their early growth after establishment. We also want to understand the underlying physiological responses (photosynthesis, transpiration, stomatal conductance, and biomass allocation).

METHODS

Sawgrass seedlings were obtained from a native plant nursery (Plant for Tomorrow, Inc., Loxahatachee, Florida) in June, 1994. Seeds were originally collected from marsh wetlands along Route 27, south of Belle Glade, Florida. The seeds were treated to increase germination percentages and planted in February, 1994 by Plants for Tomorrow, Inc. One flat, with thirty liners with fifteen seedlings per liner, was purchased. The purchased seedlings were transplanted in three densities and grown in one-gallon (16 cm dia.) plastic pots with potting soil composed of equal amounts of sawdust, perlite, and peat. Twenty liners were directly transplanted (full liner). The remaining 10 liners were split apart and planted in densities of two to four individuals or single individual seedlings. A total of 20 pots had 15 seedlings/pot (high density), 30 pots had 2 to 4 seedlings/pot (medium density), and 62 pots had one seedling/pot (low density). Initially, all of the transplants were thoroughly watered and kept in the shade for three weeks to allow them to acclimate slowly to full sunlight. Plants were grown under full sunlight and watered weekly for approximately two months, after which the plants received only rainwater. Survivorship of transplanting was surveyed for the three densities, four months after transplanting.

Nutrient treatments were applied to only two of the three densities (two to four seedlings per pot and one seedling per pot) because there was an inadequate amount of full liner seedlings to allow for replication. Ten plants from both medium density and individual density pots were randomly selected at the time of nutrient addition for determination of plant height, number of live leaves, and pre-treatment biomass. Regressions between height and number of live leaves and biomass were used for estimating initial biomass of the seedlings used for the duration of the study. All ramets within medium and low-density pots were counted and their heights measured prior to nutrient addition. Initial seedling heights of the medium and low-density plants were similar, 30.5±8.7 cm and 28.8±6.5 cm, respectively. The number of leaves for the medium and low-density plants was 8.3±1.5 and 4.6±1.0, respectively. After the survey, the seedlings were randomly divided into two groups for each density. One group received nutrient solution twice (pulsed) and the other received the same amount of tap water (control). An all-purpose plant food fertilizer (Schultz-Instant ultra pure, St. Louis, Missouri) was used for the nutrient additions. An initial dose of 240 ml; 2.8 N $g.m^{-2}$, 4.1 P $g.m^{-2}$, 2.7 K $g.m^{-2}$) was added to each pot. The second nutrient dose (240 ml; 3.8 N $g.m^{-2}$, 5.7 P $g.m^{-2}$, 3.8 K $g.m^{-2}$) was initiated one week later. After the first nutrient dose, growth measurements were taken every two weeks.

Plant morphological measurements, such as height (longest live leaf length), number of live leaves, and number or ramets were measured every two weeks for six weeks after the first nutrient addition was applied. Following the final growth measurement, physiological characteristics (photosynthesis, transpiration, and stomatal conductance) were measured for five pulsed and five control plants using a LI-COR 6250 portable photosynthesis meter. During the physiological measurement, photosynthetically-active radiation was 1187±59 mol $m^{-2}s^{-1}$, leaf chamber temperature was 30±0.03° C, and relative humidity was 48.72±0.12(%). Two to five leaves (2.86±0.03 cm^2) were put into the leaf chamber due to the small size of the live leaves. Fresh weight, dry weight, and area were determined for the leaves used in this portion of the study.

The effects of nutrient additions on biomass and relative growth rate were analyzed with a two-way analysis of covariance (ANCOVA) using SuperANOVA. The ANCOVA model treated nutrient pulse and density as fixed factors and the initial plant biomass as a covariate. First, interactions between nutrient pulse and the covariate and between density and the covariate were tested to meet the ANCOVA's assumption of homogeneity of slopes. ANCOVA was conducted only when there was no interaction between fixed factors and the covariate. For final biomass and growth rates, there was no interaction between fixed factors and the covariate. Thus, ANCOVA and least square means for these variables were used. Biomass and growth rates were estimated, assuming that initial plants were all of similar average weight. Log-transformed final biomass and growth rates were used in the ANCOVA model to meet the normal distribution of analysis of variance.

Physiological data did not meet the assumption of homogeneity of ANCOVA. The data were analyzed by two-way analysis of variance (ANOVA) in which nutrient pulses and density were treated as fixed factors. Residuals from these models were examined for normality and homoscedasticity and transformed as necessary to meet the assumptions of ANOVA.

RESULTS AND DISCUSSION

Percentage of seedling survival was surveyed two months after transplanting. Results were consistent with the hypothesis that survivorship of seedlings is the interaction between physiology and environment. Overall, the most successful seedling transplants were achieved under moist soil conditions without standing water. The highest survivorship (95%) of sawgrass seedlings was found in high density. The next highest survivorship (90%) was medium density, and the lowest one (61%) was in low density. Although the transplants in high density exhibited the highest survivorship early in the experiment, they suffered mortality later in the study. Unlike the other transplants, roots in the 15 seedings/pot transplants were tightly bunched together. The fact that the high survivorship was achieved in soil without standing water suggests that transplanting sawgrass for restoration should be conducted in the dry season.

Biomass, growth, and physiology, studied for seedlings grown in medium and low density, were significantly affected by nutrient treatments. Total and leaf biomass of pulsed plants were significantly higher than those of control plants (Table 1).

When individually grown, total and leaf biomass of pulsed plants were 30% and 57% larger than control plants, respectively. The positive effects of nutrient pulses were most apparent for medium density. Total and leaf biomass per pot in the pulsed treatment were 150% to 225% larger than those in control, respectively. In low density, total leaf area and height of pulsed plants were 100% to 112% greater than control plants, respectively. Pulsed plants for both densities produced more ramets, longer leaf lengths (height), and greater leaf area than control plants (Table 2).

Nutrient additions enhanced the growth of sawgrass seedlings, particularly above-ground leaf growth. Hence, pulsed seedlings can reach greater size than control seedlings and are more likely to overcome environmental stresses such as severe hydrologic fluctuations.

Table 1. Least square means (±SE) of total, root, leaf, and rhizome biomass of sawgrass seedlings grown for low and medium density with and without nutrient pulses. Each mean represents a sample of 12.

Density	Treatment	Total biomass(g)	Root biomass(g)	Leaf biomass(g)	Rhizome biomass(g)
Low	Pulse	1.254±0.123	0.422±0.061	0.676±0.063	0.156±0.047
	Control	0.967±0.126	0.479±0.062	0.431±0.065	0.058±0.048
Medium	Pulse	1.764±0.123	0.687±0.064	0.919±0.066	0.157±0.049
	Control	0.704±0.144	0.286±0.071	0.282±0.074	0.136±0.054

Table 2. Means (±SE) of morphological variables in sawgrass seedlings responses to nutrient pulses. Each mean represents an average of 12 pots.

Density	Nutrients	Total leaf area (cm^2)	# of ramets	Height (cm)
Low	Control	20.3 ± 7.2	1.5 ± 0.2	30.4 ± 1.4
	Pulsed	43.0 ± 10.3	1.9 ± 1.0	36.6 ± 1.4
Medium	Control	39.1 ± 6.1	2.3 ± 0.2	29.2 ± 1.6
	Pulsed	122.9 ± 46.2	3.1 ± 0.2	32.1 ± 1.5

Biomass allocation was analyzed by regression between leaf and root biomass. Although nutrient additions did not change the slopes of the regressions (0.802±0.113 vs. 0.875±0.056 for control vs. pulsed plants, respectively), control plants exhibited a greater y-intercept than pulsed plants (0.128 g vs. -0.142 g for control vs. pulsed plants, respectively). This indicates that control plants had a greater biomass allocation to roots relative to pulsed plants. One of the major mechanisms by which plants adjust to resource limitations is by shifting biomass to the organs that acquire the most strongly-limiting resources. In general, when plants have a relatively high surplus of nutrients, they compensate by producing proportionately more shoot and less root biomass. This shift in biomass allocation tends to favor higher leaf growth and increased resource use efficiency.

Photosynthetic rates (Ps) of pulsed plants were significantly greater (by approximately 32% to 45%) than control plants (Table 3). However, no differences in stomatal conductance were observed between the two nutrient treatments. As a result, instantaneous leaf water use efficiency increased in pulsed plants compared to control plants. Although instantaneous leaf water use efficiency increased in pulsed plants, whole-plant water use increased as well, largely due to enhanced leaf area in pulsed plants.

A significant ($p<0.05$) linear relationship of Ps as a function of stomatal conductance was found for both pulsed and control plants. However, pulsed plants showed higher Ps than control plants at the same stomatal conductance. A linear correlation between stomatal conductance and photosynthesis has been observed in many terrestrial plants. This correlation reflects the consistency of water-use efficiency and the ratio of internal and ambient CO_2. This relationship suggests that stomatal factors are primarily responsible for photosynthesis.

Table 3. Least square means (±SE) of photosynthetic rates (Ps mol.m^{-2}.s1, stomatal conductance (CS, cm.s^{-1}), and water use efficiency (WUE) of sawgrass seedlings grown for low and medium density with and without nutrient pulses. Each mean represents a sample of 5.

Density	Treatment	Ps	CS	WUE
Low	Pulse	12.78±1.99	1.24±0.11	1.48±0.23
	Control	9.68±0.66	1.012±0.08	1.27±0.14
Medium	Pulse	13.55±1.42	0.88±0.12	1.96±0.11
	Control	9.37±1.27	0.95±0.11	1.10±0.08

In summary, high success of sawgrass transplants was achieved under moist soil conditions without standing water, and growth was increased by nutrient additions at an early growth stage. The greater growth of pulsed plants primarily resulted from increases in both photosynthetic rates per unit leaf area and total photosynthetic area per plant. Although the transplanting of sawgrass in the Everglades is too large a task to be a viable restoration option at this time, this experiment demonstrates that small-scale stormwater treatment areas or experimental field pots can be managed to enhance sawgrass seedling survival and growth.

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DEVELOPMENT OF PARADIGMS FOR QUANTITATIVE ASSESSMENT OF THE SUCCESS OF WETLAND CREATION AND RESTORATION PROJECTS FOR WILDLIFE

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ABSTRACT

Successful replication of wetland functions in creation and restoration projects is difficult to quantitatively document. Specified monitoring plans and measurable success criteria in permit conditions and mitigation plans are lacking. In particular, wildlife criteria are rarely specified. The objectives of this study were to devise and evaluate alternative, quantitative paradigms for assessment of success of wetland creation and restoration projects, using the avian community as the metric for success. The avian community and vegetation at a wetland creation site and nearby reference sites in the coastal plain of South Carolina were sampled between February 1993 and February 1995. These data were used for examination of several different approaches that were developed to evaluate success. One paradigm evaluated was temporal changes in relevant biological parameters, such as species richness, diversity, and density, of each site. A second approach was to compare these parameters with those of reference wetlands. To overcome the inherent problems of variability among individual reference wetlands, a third approach was to construct a hypothetical avian community, through a literature search, for the wetland types in this study, and classify bird species based on their degree of wetland dependency. The possibility of adapting existing wetland evaluation techniques for use as evaluators of success was explored as a fourth approach. Paradigms were compared and contrasted in terms of their feasibility and sensitivity.

INTRODUCTION

The practice of wetland restoration and creation has increased considerably in recent years, especially for the purpose of mitigation for losses of wetland area, functions, and values (Atkinson et al. 1993). Although great biological and engineering strides have been made in restoring and creating wetlands, it remains difficult to document success of these projects in terms of replicating wetland functions. There are 4 main reasons for this lack of rigorous evaluation. First, our knowledge of how wetlands function is far from complete (Bacchus 1991). Second, there is a lack of conclusive long-term studies which demonstrate man's ability to recreate functions of natural wetlands (Adamus 1988, Bacchus 1991). Third, post-construction monitoring has been lacking or insufficient to address the issue of functional success (Clewell and Lea 1990). Fourth, measurable success criteria in permit conditions and mitigation plans have been rare (Josselyn and Bucholz 1982, Shisler and Charette 1984, Eliot 1985, Maguire 1985, Reimold and Cobler 1985, Dial and Deis 1986, Quammen 1986, Clewell and Lea 1990). When success criteria are included, they are often simple, vague or qualitative (Adamus 1988, Clewell and Lea 1990). The most frequently used quantitative criterion is survival rate of planted vegetation (R. Banks, pers. comm.), but it is doubtful whether this adequately addresses successful duplication of wetland functions.

Wetlands are highly valuable as wildlife habitat because of their combination of plant communities and water regimes that provide the necessary resources for feeding, reproduction, and cover of a great diversity of wildlife (Shaw and Fredine 1956, Toburen and Windell 1977). Wetlands are especially important to birds. Approximately one-third of all North American bird species use wetlands to satisfy some or all of their life functions (Kroodsma 1978). This is particularly impressive because wetlands comprise only 5% of the total land area in the continental United States (Dahl 1990). Avian diversity is generally higher in forested wetlands than in surrounding uplands (Wharton et al. 1981). Avian densities have also been found to be significantly higher in forested wetlands than in surrounding uplands (Reese and Hair 1976, Dickson 1978). Approximately one-third of birds on the U.S. endangered and threatened species list are wetland dependent (Williams and Dodd 1978). Despite the widely accepted belief that providing excellent wildlife habitat is an invaluable function of wetlands, wildlife criteria are rarely considered in mitigation plans and permit conditions. Thus, there is a need for objective, quantitative methods for evaluation of success of wetland mitigation projects that consider wildlife resources.

A wetland mitigation project in the upper coastal plain of South Carolina provided an opportunity to address these issues. Construction of a French Drain at the Pinewood Hazardous Waste Landfill, near Pinewood, South Carolina, destroyed a tract of bottomland hardwood forest. To mitigate for this loss, Laidlaw Environmental Services, the operator of the landfill, built a mosaic of wetland impoundments, totalling approximately 33 ha, on a site approximately 6.5 km south of the landfill, near Rimini, South Carolina. This site had previously been mined for clay, to use as liner in construction of the landfill.

The location, size, and shape of the impoundments were largely determined by the previous mining operation. Topsoils removed during the mining process, including Cantey loam, Rains sandy loam, Persanti very fine sandy loam and Dothan loamy fine sand, were stockpiled on-site. The depressions made by mining were graded to specified contours, including littoral shelves ranging from 20 to 70 m in width. Dikes, flashboard risers, and connecting canals were constructed, and stockpiled soils were spread over the impoundments. Herbaceous vegetation was planted on 1 m centers around the middle portions of the littoral shelf. Sapling trees were planted on 5 m centers in the middle and upper portions of the littoral shelf, and some larger trees were transplanted from surrounding wooded areas.

The first impoundment completed was a 6.9 ha wading bird habitat (Unit 2). Construction of this unit was completed in late 1992, and planting was conducted in February 1993. Construction of a 15.8 ha reservoir (Unit 1) and a 10.7 ha moist-soil emergent unit (Unit 5) was completed in late 1993. Trees were planted in January 1994, and herbaceous vegetation was planted in late March 1993. Also on site, but not required for mitigation, are 2 agricultural impoundments of 5.7 and 3.6 ha (Units 6 and 7). These impoundments are planted with agricultural crops that are attractive to waterfowl and flooded in winter. Surrounding habitat includes small woodlots, agricultural fields, and a pine plantation. A large lake is approximately 2 km away. The entire site is managed by a private conservation organization as a wetlands and wildlife management and education center.

The objectives of this study were to (1) devise and evaluate alternative, quantitative paradigms for assessment of success of wetland creation and restoration projects, using the avian community as the metric for success, and (2) describe avian use and vegetation of the created wetlands to aid in management of the site and add to a lack of available data on bird use of created wetlands.

MATERIALS AND METHODS

Bird Sampling

Field data was collected from the study areas approximately monthly between February 1993 and February 1995. Impoundments were scanned every 5 minutes for 30 minutes with spotting scope and binoculars from observation towers. All birds seen were counted by species. Each impoundment was sampled at 3 randomly selected times, within 3 time strata, during 2 days in each visit. Unit 2 was sampled starting in February 1993, Unit 1 starting May 1993, and Units 5, 6, and 7 starting January 1994. Units 6 and 7 were only sampled when flooded.

Vegetation Sampling

Ground vegetation was sampled by estimating percent cover of vegetation by species by ocular estimation in 1 m^2 quadrats. Quadrats were located at 5 m intervals along transects running perpendicular to the littoral shelf. Transects were established with a systematic random sampling design. Trees were sampled by counting and measuring diameter at breast height (dbh) by species in 10 m^2 quadrats. These quadrats were randomly located on half of the transects.

Temporal Changes Paradigm

With this approach, one could assess the rate of change of avian and vegetation community parameters to look for a pattern of improvement. Alternatively, time specific goals for these parameters could be specified in mitigation plans. As an example, estimates of several parameters of the avian community and vegetation of Unit 2 were compared between Year 1 and Year 2. Data presented from the mitigation site will be limited to Unit 2 to simplify examples, and because 2 years of data collection have been completed.

Comparisons with Reference Sites Paradigm

A second approach was to compare created wetlands on the mitigation site with reference wetlands. Our criteria for selection of reference wetlands were: short distance (< 50 km) from mitigation site, permission to access private property, and similar type and size of wetland. Avian scans of open water were conducted with the same methods and schedule as the mitigation site except that distance and thick vegetation forced exclusion of passerines and other small birds. Therefore, comparisons were limited to birds that feed primarily in water (waterbirds). However, forested sites adjacent to the water were sampled with point counts to account for all bird species. All birds seen or heard were recorded by species and sex in a 10 min. period within a 50 m radius semi-circle. There were 4 sampling points at each reference site.

One of the reference sites we selected was a private hunt club (Taylor) located < 2 km from the mitigation site. It contained a small body of water with dead standing trees, surrounded by a narrow ring of bottomland hardwoods. The second reference area used was the Cuddo Unit of Santee National Wildlife Refuge. We selected a small open-water wetland (Goose Pen Pond) and a nearby greentree reservoir. We also sampled a small tract of bottomland hardwoods on the same property as the mitigation site (Stewart Tract). Several parameters of the avian communities were compared among Unit 2 and the reference wetlands.

Hypothetical Avian Community Paradigm

A hypothetical avian species list was generated for the mitigation site through a literature search, using Hamel (1992) as the primary reference. A list of birds was determined by selecting the Oak-Gum-Cypress habitat type and shrub/seedling successional stage, which most closely matches the current state of the created wetlands. The list was narrowed by deleting species that do not occur in the coastal plain of South Carolina. This list was compared with the avian species list generated from observations of Unit 2.

Wetland indicators were assigned to bird species on the hypothetical list, as well as all birds observed, in the same manner as the *National List of Plant Species That Occur in Wetlands* (Reed 1988). The indicators are based on the probability of finding the species in a wetland vs. a non-wetland. For example, facultative wetland (facw) indicates that a species is usually found in a wetland (67%-99%), but is occasionally found in non-wetlands (uplands or deep water). The determination of indicator was made with the help of habitat preference descriptions in Hamel(1992) and Peterson (1980). Assignments contain a degree of subjectivity, and they should be considered as a useful example, but not definitive. These indices were used to separate "wetland" from "non-wetland" birds in data analysis, and to provide another tool for describing an avian community that could be used in comparative analysis.

Adaptation of Existing Evaluation Techniques Paradigm

The possibility of adapting existing wetland evaluation techniques for use as tools in measuring success of created wetlands was explored through a review of existing techniques. We focused on the Habitat Evaluation Procedure (H.E.P.) (U.S. Fish and Wildlife Service 1980) and the Wetland Evaluation Technique (Adamus et al. 1987).

RESULTS

Temporal Changes

A clear trend of improvement in the avian community of Unit 2 could not be detected from the first to second year after construction using 3 of the most common measures. Density, species richness, and diversity for the entire avian community were similar between years (Table 1). However, statistics for birds typical of wetlands did show improvement. For example, percentage of individual birds that were classified as "wetland" (facultative wetland or obligate) increased substantially. Similarly, waterbird and waterfowl densities increased. Passerine use of the Unit, so far, has been primarily limited to species that prefer open habitats. Planting herbaceous vegetation and stocking fish in shallow water attracted waterfowl and wading birds in the early stages post-construction, whereas decades will be required to produce a mature bottomland hardwood forest and its associated avifauna by planting sapling trees. A clear trend of improvement is evident in the ground vegetation on Unit 2 between Years 1 and 2. Percent cover by vegetation of all species and by "wetland" species increased (Table 2). Also, richness (number of genera) and diversity showed substantial increases.

Table 1. Mean observed avian density at one instant in time (birds/ha), species richness, diversity (Shannon Index), and percentage of birds classified "wetland" on Unit 2 at Laidlaw mitigation site, Rimini, South Carolina in Years 1 and 2.

STATISTIC	YEAR 1	YEAR 2
Avian Density	0.37	0.40
Waterbird Density	0.15	0.33
Waterfowl Density	0.09	0.23
Avian Species Richness	31	27
Avian Diversity	2.01	2.17
Percent Wetland Birds	40.9	92.0

Table 2. Percent cover vegetation, percent cover wetland vegetation, vegetation genera richness, vegetation diversity (Shannon Index) on Unit 2 at Laidlaw mitigation site, Rimini, South Carolina in Years 1 and 2.

STATISTIC	YEAR 1	YEAR 2
Percent Cover Vegetation	16.2	45.5
Percent Cover - Wetland Vegetation	15.8	34.9
Vegetation Genera Richness	8	15
Vegetation Diversity	0.49	1.62

Comparisons with Reference Sites

Waterbird density on Unit 2 was similar to Taylor, but both were much lower than Goose Pen Pond, where much greater use by wintering waterfowl was observed (Table 3). Waterbird richness was surprisingly highest on Unit 2, probably because of shallow water and a lack of vegetation that attracted mixed flocks of migrating shorebirds in Spring and Fall. Waterbird diversity was the same on Unit 2 and Taylor, but lower on Goose Pen Pond where deeper water limited access by many species of waterbirds. The index of similarity, which is the number of species observed in both wetlands divided by the number of species observed in either wetland and expressed as a percentage, showed Unit 2 to be most similar to Taylor in species composition of waterbirds, perhaps because of their close proximity (< 2 km). Goose Pen Pond was much less similar, probably due to deeper water and different vegetation. When the scans of the bodies of water were combined with the forested point counts, species richness of the reference sites was considerably higher than Unit 2 (Table 4). This further shows that a created forested wetland cannot reproduce natural forested wetland functions in a short period of time.

Table 3. Mean observed density (birds/ha), species richness, diversity (Shannon Index), and index of similarity (%) of waterbirds on Unit 2 at Laidlaw mitigation site and Taylor hunt club, Rimini, South Carolina; and Goose Pen Pond, Santee National Wildlife Refuge, South Carolina, February 1993 - February 1995.

STATISTIC	UNIT 2	TAYLOR	GOOSE PEN	GOOSE AND TAYLOR
Waterbird Density	0.24	0.28	1.20	0.39
Waterbird Richness	21	13	18	23
Waterbird Diversity	1.86	1.86	1.51	1.88
Index of Similarity	X	43.5	22.6	34.8

Table 4. Avian richness and index of similarity (%) (scans and point counts pooled) on Unit 2 at Laidlaw mitigation site, and Taylor Hunt Club, Rimini, South Carolina; and Cuddo Unit of Santee National Wildlife Refuge; and pooled references, February 1993 - February 1995.

STATISTIC	UNIT 2	TAYLOR	SANTEE	POOLED REFERENCES
Richness	38	63	60	83
Index of Similarity	X	24.7	22.5	24.0

Hypothetical Avian Community

Bird species included in the hypothetical avian community for Unit 2, as well as all species observed on Unit 2 and its reference sites, are listed in Appendix 1, along with their assigned wetland indicator and location. The index of similarity between observed species on Unit 2 and the hypothetical list was 20% in Year 1 and 19% in Year 2. These low indices, and the fact that they don't show improvement, further confirm that a bottomland hardwood cannot be duplicated quickly.

Adaptation of Existing Wetland Evaluation Techniques

There are several standardized techniques that have been used to evaluate wetlands for wildlife. These techniques have been mostly used for prioritizing habitat preservation or assessing a wetland before a permitted disturbance. The same techniques could be applied to created and restored wetlands to assess how well wildlife habitat had been replicated. The Wetland Evaluation Technique (W.E.T.) (Adamus et al. 1987) and the Habitat Evaluation Procedure (H.E.P.) (U.S. Fish and Wildlife Service 1980) have been used for this purpose in a few instances (Adamus 1988).

H.E.P. is probably the most useful method for wildlife. It is designed to quantify the relative value of wildlife habitat of different sites at the same point in time or the relative value of the same area at future points in time. Indicator species(s) are selected for the site, habitat suitability indexes (HSI) are calculated for each species by surveying the habitat and relating the data to an existing model, and the HSI is multiplied by area to obtain habitat units (HU's). In wetland mitigation, this procedure could be used before a wetland experienced a known, permitted disturbance, and then on the created or restored wetland. One would then assess the differences between these values, or pre-specified, time specific values could be set as success criteria.

DISCUSSION

Each paradigm considered in this paper has strengths and weaknesses, and no one method emerged as clearly superior for our situation. Using temporal changes of individual created or restored wetlands allows for the cleanest comparisons since the wetland essentially serves as its own reference. It is also less labor intensive than using reference sites. The significant problem with this approach is deciding what rate of change or goal value to consider "successful." One must have some knowledge of how the biological characteristics of the wetland should change over time, or at the very least, the characteristics of a mature, natural wetland of similar size and type.

Use of reference sites would seem to be a logical approach to evaluate success, and in some circumstances they may be. However, a high degree of variability among individual wetlands makes using one or even a few wetlands as comparisons difficult. How well the wetland in question replicates the reference wetland may depend as much on the selection of the reference as it does on the ability of the created or restored wetland to replicate natural wetland functions. Using several randomly selected references may help with variability and bias in selection. However, this will likely be too expensive and impractical in most situations. Perhaps a more important problem when dealing with created forested wetlands is that references will almost always be in the wrong successional stage. Comparing a 1 year old created wetland with sapling trees to a mature bottomland hardwood forest cannot be considered a fair comparison. The reference, in this case, could only be considered a goal.

Construction of a hypothetical avian community using known species habitat relationships may help overcome some of the variability associated with reference sites and may help to avoid confounding caused by different successional stages. It is also much less time consuming then using reference sites. However, there will likely be some subjectivity, bias, and inaccuracies in the list.

Adaptation of existing modeling procedures for use as tools to measure success is attractive because the methods already exist. They provide quantitative output for use in comparisons. However, these methods rely heavily on habitat models that require questionable assumptions of species-habitat relationships.

We recommend that a collection of well designed, long and short-term studies be conducted to help determine what is typical of the avian community (as well as other wetland functions) in the early stages of created wetlands and how these communities change over a long period of time, especially in forested wetlands. These studies should be done in a variety of habitat types, on both created and restored wetlands, as well as natural sites. An adequate collection of these studies would provide a database sufficient to eliminate use of references specifically for each project. A "reference" could be culled from this database in a manner similar to the hypothetical list discussed here.

Because providing excellent wildlife habitat is an acknowledged function of wetlands, wildlife should be included with other wetland functions as measurable success criteria in mitigation plans. Data on wildlife habitat and use should be collected, whenever possible, before wetland site is altered. In this manner, the establishment of success criteria for mitigation is less arbitrarily determined.

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Appendix 1. Avian species list from hypothetical list and observations of created wetland (Unit 2) and its reference wetlands in Clarendon County, South Carolina, February 1993 - February 1995.

SPECIES	INDICATOR	LOCATION
Pied-billed grebe (Podilymbus podiceps)	facw	3 4

SPECIES	INDICATOR	LOCATION
Double-crested cormorant (Phalacrocorax auritus)	facw	2
Anhinga (Anhinga anhinga)	facw+	1 2 3 4
Great blue heron (Ardea herodias)	facw+	1 2 3 4 5 6
Green-backed heron (Butorides striatus)	facw+	1 2 3 4 5 6
Little blue heron (Florida caerulea)	facw+	1 2 4
Cattle egret (Bubulcus ibis)	facw	1
Great egret (Casmerodius albus)	facw+	1 2 3 4
Snowy egret (Egretta thula)	facw+	1 2 4
Tri-colored heron (Hydranassa tricolor)	facw+	1 2
Black-crowned night heron (Nycticorax nycticorax)	facw+	1
Yellow-crowned night heron (Nyctanassa violacea)	facw+	1
Least bittern (Ixobrychus exilis)	obl	1 3
American bittern (Botaurus lentiginosus)	obl	3
Wood stork (Mycteria americana)	facw+	1
Turkey vulture (Cathartes aura)	fac	1
Black vulture (Coragyps atratus)	fac	1
White ibis (Eudocimus albus)	facw+	1 4
Mallard (Anas platyrhynchos)	facw+	1 2 3 4 5
Northern pintail (Anas acuta)	facw+	
Green-winged teal (Anas crecca)	facw+	3
Blue-winged teal (Anas discors)	facw+	3
Wood duck (Aix sponsa)	facw+	1 2 3 4 5 6
Ring-necked duck (Aythya collaris)	facw+	2 3
Lesser scaup (Aythya affinis)	facw+	3

SPECIES	INDICATOR	LOCATION						
Hooded merganser (Lophodytes cucullatus)	facw+	1	2					
Red-tailed hawk (Buteo jamaicensis)	fac						6	
Red-shouldered hawk (Buteo lineatus)	facw+	1						
Northern harrier (Circus cyaneus)	facw		2					
Osprey (Pandion haliaetus)	facw		2		4			
American kestral (Falco sparverius)	fac		2					
Northern bobwhite (Colinus virginianus)	fac						6	7
Wild turkey (Meleagris gallopavo)	fac	1						
Purple gallinule (Porphyrula martinica)	obl	1		3				
Common moorhen (Gallinula chloropus)	obl	1		3				
American coot (Fulica americana)	facw+			3				
Killdeer (Charadrius vociferus)	fac		2					
Greater yellowlegs (Tringa melanoleuca)	facw		2					
Lesser yellowlegs (Tringa flavipes)	facw		2					
Spotted sandpiper (Actitis macularia)	facw		2		4			
American woodcock (Philohela minor)	facw	1		3				
Common snipe (Capella gallinago)	facw	1						
Semipalmated sandpiper (Calidris pusilla)	facw		2					
Ring-billed gull (Larus delawarensis)	facw		2					
Least tern (Sterna antillarum)	facw		2					
Mourning dove (Zenaida macroura)	fac	1	2			5	6	7
Yellow-billed cuckoo (Coccyzus americanus)	fac					5		7
Ruby-throated hummingbird (Archilochus colubris)	fac+	1					6	
Belted ingfisher (Megaceryle alcyon)	facw		2	3	4	5		

SPECIES	INDICATOR	LOCATION
Norther flicker (Colaptes auratus)	fac	5 6 7
Pileated woodpecker (Dryocopus pileatus)	fac	5 7
Red-bellied woodpecker (Melanerpes carolinus)	fac	5 6 7
Red-headed woodpecker (Melanerpes erythrocephalus)	fac	5 6
Hairy woodpecker (Picoides villosus)	fac	6 7
Downy woodpecker (Picoides pubescens)	fac	1 5 6 7
Eastern kingbird (Tyrannus tyrannus)	fac	2 5 6 7
Great-crested flycatcher (Myriarchus crinitus)	fac	1 5 6 7
Eastern phoebe (Sayornis phoebe)	fac	1 2 5 6 7
Eastern wood-peewee (Contopus virens)	fac	1 5 7
Tree swallow (Iridoprocne bicolor)	fac+	1 7
Rough-winged swallow (Stelgidopteryx ruficollis)	facw	2
Barn swallow (Hirundo rustica)	fac	2
Blue jay (Cyanocitta cristata)	fac	1 6 7
American crow (Corvus brachyrhynchos)	fac	1
Fish crow (Corvus ossifragus)	facw	1 2
Carolina chickadee (Parus carolinensis)	fac	1 5 6 7
Tufted titmouse (Parus bicolor)	fac	1 5 6 7
Brown-headed nuthatch (Sitta pusilla)	fac	7
House wren (Troglodytes aedon)	fac	1 6
Carolina wren (Thryothorus ludovicianus)	fac	1 5 6 7
Northern mockingbird (Mimus polyglottos)	fac	2 5 6 7
Gray catbird (Dumetella carolinensis)	fac	1 6
Brown thrasher (Toxostoma rufum)	fac-	1 5 7

SPECIES	INDICATOR	LOCATION
American robin (Turdus migratorius)	fac	1 5 6 7
Hermit thrush (Catharus fuscescens)	fac	5
Eastern bluebird (Sialia sialis)	fac	5 6 7
Blue-gray gnatcatcher (Polioptila caerulea)	facw-	1 5 6 7
Golden-crowned kinglet (Regulus satrapa)	fac	5
Ruby-crowned kinglet (Regulus calendula)	fac	1 5 6 7
Cedar waxwing (Bomycilla cedrorum)	fac	1
Loggerhead shrike (Lanius ludovicianus)	fac-	2
European starling (Sturnus vulgaris)	fac	1 2 6
White-eyed vireo (Vireo griseus)	fac+	1 5 6 7
Rough-winged swallow (Stelgidopteryx ruficollis)	facw	2
Barn swallow (Hirundo rustica)	fac	2
Blue jay (Cyanocitta cristata)	fac	1 6 7
American crow (Corvus brachyrhynchos)	fac	1
Fish crow (Corvus ossifragus)	facw	1 2
Carolina chickadee (Parus carolinensis)	fac	1 5 6 7
Tufted titmouse (Parus bicolor)	fac	1 5 6 7
Brown-headed nuthatch (Sitta pusilla)	fac	7
House wren (Troglodytes aedon)	fac	1 6
Carolina wren (Thryothorus ludovicianus)	fac	1 5 6 7
Northern mockingbird (Mimus polyglottos)	fac	2 5 6 7
Gray catbird (Dumetella carolinensis)	fac	1 6
Brown thrasher (Toxostoma rufum)	fac-	1 5 7
American robin (Turdus migratorius)	fac	1 5 6 7

SPECIES	INDICATOR	LOCATION				
Hermit thrush (Catharus fuscescens)	fac			5		
Eastern bluebird (Sialia sialis)	fac			5	6	7
Blue-gray gnatcatcher (Polioptila caerulea)	facw-	1		5	6	7
Golden-crowned kinglet (Regulus satrapa)	fac			5		
Ruby-crowned kinglet (Regulus calendula)	fac	1		5	6	7
Cedar waxwing (Bomycilla cedrorum)	fac	1				
Loggerhead shrike (Lanius ludovicianus)	fac-		2			
European starling (Sturnus vulgaris)	fac	1	2		6	
White-eyed vireo (Vireo griseus)	fac+	1		5	6	7
Rough-winged swallow (Stelgidopteryx ruficollis)	facw		2			
Barn swallow (Hirundo rustica)	fac		2			
Blue jay (Cyanocitta cristata)	fac	1			6	7
American crow (Corvus brachyrhynchos)	fac	1				
Fish crow (Corvus ossifragus)	facw	1	2			
Carolina chickadee (Parus carolinensis)	fac	1		5	6	7
Tufted titmouse (Parus bicolor)	fac	1		5	6	7
Brown-headed nuthatch (Sitta pusilla)	fac					7
House wren (Troglodytes aedon)	fac	1			6	
Carolina wren (Thryothorus ludovicianus)	fac	1		5	6	7
Northern mockingbird (Mimus polyglottos)	fac		2	5	6	7
Gray catbird (Dumetella carolinensis)	fac	1			6	
Brown thrasher (Toxostoma rufum)	fac-	1		5		7
American robin (Turdus migratorius)	fac	1		5	6	7
Hermit thrush (Catharus fuscescens)	fac			5		

SPECIES	INDICATOR	LOCATION
Eastern bluebird (Sialia sialis)	fac	5 6 7
Blue-gray gnatcatcher (Polioptila caerulea)	facw-	1 5 6 7
Golden-crowned kinglet (Regulus satrapa)	fac	5
Ruby-crowned kinglet (Regulus calendula)	fac	1 5 6 7
Cedar waxwing (Bomycilla cedrorum)	fac	1
Loggerhead shrike (Lanius ludovicianus)	fac-	2
European starling (Sturnus vulgaris)	fac	1 2 6
White-eyed vireo (Vireo griseus)	fac+	1 5 6 7
Yellow-throated vireo (Vireo flavifrons)	fac+	5 6
Prothonotary warbler (Protonotaria citrea)	obl	5 6
Orange-crowned warbler (Vermivora celata)	fac	1
Northern parula warbler (Parula americana)	fac+	5 6 7
Yellow-rumped warbler (Dendroica coronata)	fac+	1 5 6 7
Pine warbler (Dendroica pinus)	fac	5 6
Palm warbler (Dendroica palmarum)	fac	6 7
Common yellowthroat (Geothlypis trichas)	facw	1 5 6 7
Yellow-breasted chat (Icteria virens)	fac-	1 5 6 7
Hooded warbler (Wilsonia citrina)	fac+	5 6 7
Red-winged blackbird (Agelaius phoeniceus)	facw	1 2 5 6 7
Orchard oriole (Icterus spurius)	fac	5 6 7
Rusty blackbird (Euphagus carolinus)	facw	1
Boat-tailed grackle (Quiscalus major)	facw	1
Common grackle (Quiscalus quiscula)	fac	1 2 5 6 7
Brown-headed cowbird (Molothrus ater)	fac	1 2 5 6 7

SPECIES	INDICATOR	LOCATION
Summer tanager (Piranga rubra)	fac-	5 6 7
Northern cardinal (Cardinalis cardinalis)	fac	1 2 5 6 7
Blue grosbeak (Guiraca caerulea)	fac-	1 2 5 6 7
Indigo bunting (Passerina cyanea)	fac	1 6 7
Painted bunting (Passerina ciris)	fac	1 5 6
American goldfinch (Carduelis tristis)	fac	1 6
Rufous-sided towhee (Pipilo erythrophthalmus)	fac	1 5 6 7
Northern junco (Junco hyemalis)	fac	7
Chipping sparrow (Spizella passerina)	fac-	6
Field sparrow (Spizella pusilla)	fac	6 7
White-throated sparrow (Zonotrichia albicollis)	fac+	1 6
Fox sparrow (Passerella iliaca)	fac	1
Swamp sparrow (Melospiza georgiana)	facw+	1 5 6 7
Song sparrow (Melospiza melodia)	fac+	1 2 5 6 7

1 = Hypothetical List for Unit 2
2 = Unit 2
3 = Goose Pen Pond
4 = Taylor Pond
5 = Greentree Reservoir
6 = Taylor Bottomland Hardwood
7 = Stewart Bottomland Hardwood
obl = Almost always found in wetlands (>99%)
facw = Usually found in wetlands(67%-99%)
fac = Equally likely to be found in wetlands as nonwetlands (34%-66%)
facu = Usually found in uplands (67%-99%)
upl = Almost always found in uplands (>99%)
+ = Higher end of category (more frequently found in wetlands)
- = Lower end of category (less frequently found in wetlands)

WETLAND CREATION AT KISSIMMEE UTILITY AUTHORITY'S CANE ISLAND PROJECT

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ABSTRACT

Kissimmee Utility Authority has constructed a power plant consisting of a combustion turbine unit and a combined cycle unit at the Cane Island Project site near Intercession City, Osceola County, Florida. As a result of construction impacts to wetlands, mitigation was required by the US Army Corps of Engineers (COE). An isolated wetland was created in 1993 at the Cane Island site. The wetland encompasses 2.6 hectares (ha) and includes three islands, each covering 0.1 ha. A total of 5,186 immature trees were planted, including 4,498 pond cypress (Taxodium ascendens) and 400 black gum (Nyssa sylvatica) saplings, and 288 bald cypress (Taxodium distichum) seedlings. Bald cypress and black gum were planted at the edge of the wetland at elevations experiencing less inundation. Pond cypress were planted in deeper portions of the wetland. As required by the COE, success criteria were established for a 3 year monitoring period including 80 percent tree survivorship and exotic and nuisance vegetative cover of less than 10 percent. A monitoring plan was initiated in 1993 to determine whether the criteria are met. After 1 year of monitoring, the wetland creation site met all success criteria. Rate of tree survivorship was 89.0 percent and mean cover of exotic and nuisance species was 0.4 percent.

INTRODUCTION

The Kissimmee Utility Authority (KUA) and Florida Municipal Power Agency have constructed and are operating a electric generating facility consisting of a 40 MW combustion turbine and a 120 MW combined cycle unit at the Cane Island site (Site) near Intercession City, Florida. As a result of construction impacts to wetlands, gopher tortoises and their habitat, and sand pine scrub habitat, mitigation was required by the US Army Corps of Engineers (COE), Florida Department of Environmental Protection (DEP), South Florida Water Management District (SFWMD), and Florida Game and Freshwater Fish Commission (FGFWFC). Under the supervision of Black & Veatch, a comprehensive and multi-faceted Mitigation Plan was developed. The Plan includes wetland creation, incorporation of conservation easements, avian collision mitigation, and management of conserved uplands and wetlands.

The research reported here focuses on the 1993 and 1994 results of the 2.6 ha artificial wetland created in 1993. The objectives of this research are to determine whether COE success criteria are met through a 3 to 5 year monitoring of tree survivorship and growth, exotic and nuisance vegetation abundance, and groundwater and surface water elevations at the Site.

STUDY SITE

The site covers 416 ha in Osceola County, approximately 2.4 kilometers (km) northwest of Intercession City, and 32.2 km southwest of Orlando, Florida (Figure 1).

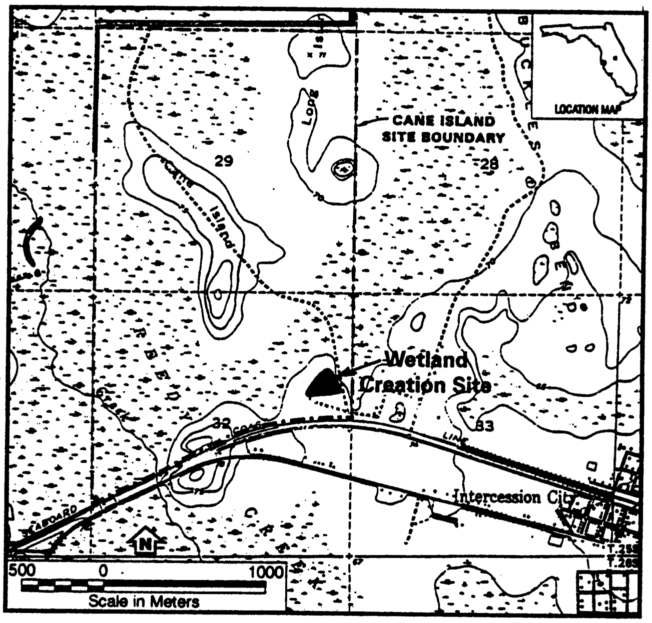

Figure 1. Cane Island project site.

The site lies within the Osceola Plain physiographic section of central Florida (Readle, 1979). The Osceola Plain has little relief locally, with elevations generally ranging from 19.8 to 22.9 meters (m) amsl. Soils are sandy, with the upland soils low in natural fertility and organic matter, and generally well-drained. The wetland soils are poorly drained fine sands, inundated or saturated with water 6 to 12 months of most years. The wetland soils are also low in organic matter and natural fertility.

A mix of forested wetlands (mixed hardwood swamp, hydric hammock, and cypress strand/pond) and uplands (sand pine scrub, improved grassland, old fields, oak scrub, flatwoods, pine flatwoods, pine-mesic oak, and mesic hammock) make up the site (Black & Veatch, 1992).

The wetland creation site is located in the southeast corner of the Site (Figure 1) in an abandoned pasture. This area is a former farmstead, which has been disturbed by cattle grazing and land clearing. The area is undergoing secondary succession and consists of early- to mid-successional plant communities, including old fields of broomsedge (Andropogon virginicus), saw palmetto (Serenoa repens), mixed hardwood, and slash pine (Pinus elliottii). This area was selected as the wetland creation site because of its disturbed nature, low plant diversity, and lack of protected species.

MATERIALS AND METHODS

A team of wetlands ecologists, wildlife biologists, and engineers developed the wetland creation site plan. Construction and design of the wetland follow standard engineering practices and mimics natural wetland ecology and hydrology. The design is based on data from site soil borings and topographical surveys (Black & Veatch, 1993), extensive on-site plant community investigations (Black & Veatch, 1992), local hydrologic and climatic patterns, and scientific literature on wetlands ecology and creation (Ewel and Odum, 1984; Beever, 1986; Myers and Ewel, 1990; Hammer, 1991).

The objective of the wetland creation plan is not only to mitigate for the loss of wetlands as a result of the Cane Island Project, but also to benefit wildlife by the diversification of wildlife habitat on-site.

Mitigation Requirements

Wetland creation was required for the mechanical clearing and filling of 0.8 ha herbaceous/shrubby and 2.1 ha of forested wetlands. Project mitigation ratios to the COE for wetlands creation are 1:0.4 (wetlands impacted: wetlands created) for disturbed shrubby/herbaceous wetlands and 1:1 for forested wetlands. With these mitigation ratios, 2.4 ha of wetlands creation were required.

Wetland Construction

Excavation of the wetland creation site began on June 1, 1993. Construction of the wetland site was finished, with final grading during the week of August 16, 1993. Surveying of the wetland site was conducted between August 8 and September 26. Figure 2 is a plan view of the artificial wetland, showing elevations of each wetland area.

Water sources are groundwater, precipitation, and surface water runoff. Excavation in uplands 0.2 ha started at existing grade, an elevation of approximately 21.6 m, and extended to an elevation of 21.0 m. Slope is 5:1, and the width of this upland-wetland edge is approximately 3 m.

Two wetland zones were constructed. Wetland Zone 1 is a sloping area extending from the upland edge at an elevation of 21.0 m into the wetland at 20.7 m. This area generally experiences periodic inundation or saturation up to an elevation above Zone 1 near 21.6 m. Wetland Zone 1 has a slope ranging from 5:1

to 25:1 and an approximate width of 1.8 to 4.6 m, and covers approximately 0.4 ha. Wetland Zone 2 extends from Wetland Zone 1 approximately 36.6 m to elevation 20.3 m. This area experiences significant periods of inundation. The slope of this area varies from 50:1 to level. Zone 2 covers 1.9 ha.

At the request of the COE, three islands (0.1 ha each) were included in the wetland creation site to preserve some semi-mature trees at the site and provide additional wildlife habitat.

Wetland Planting

Wetland Planting Acclaim Environmental, Inc., of Clearwater, Florida, planted trees on September 23 and October 29, 1993; 5,186 young trees were planted, including 4,498 pond cypress (Taxodium ascendens) and 400 black gum (Nyssa sylvatica) saplings, and 288 bald cypress (Taxodium distichum) seedlings. Three sizes of pond cypress (28.3 liter [l]); 12.1 l; and 4.0 l); and one size of black gum (4.0 l) and bald cypress 15.2 centimeter (cm) cone were planted. Saplings were planted on 1.5 to 3.0 m centers. Black gum and pond cypress were primarily planted in Wetland Zones 1 and 2, respectively. Bald cypress was planted only in Zone 1.

According to wetland planting specifications, all trees that died within the first year after planting in 1993 were replaced in 1994. Consequently, a total of 488 seedlings/saplings, including 45 pond cypress, 52 black gum, 180 red maple, and 171 laurel oak, were replanted at the wetland between August 29 and September 26, 1994. The red maple and laurel oak seedlings were planted in Wetland Zone 1 to replace the dead bald cypress and black gum because these species are more tolerant of the highly variable conditions at the edge of the wetland than bald cypress and black gum. No additional bald cypress seedlings were planted at the wetland. Black gum was replanted in Zone 2. Figure 2 shows the final planting scheme.

Monitoring Methodology

Success Criteria

Monitoring criteria were required for this project. The following criteria, based on discussions with the COE, DEP, and SFWMD, were incorporated into the methodology:

- Survivorship rate must be maintained at 80 percent or more for planted tree species for 3 consecutive years of monitoring. Trees must be living and reproducing as indicated by the appearance of fruit or an increase in height or diameter.

- Percent cover of exotic/nuisance wetland species (i.e., Australian pine [Casuarina litorea], Chinese tallow tree [Sapium sebiferum], downy myrtle [Rhodomyrtus tomentosus], primrose willow [Ludwigia peruviana], cattails [Typha spp.], Brazilian pepper [Schinus terebinthifolius], and punk tree [Melaleuca quinquenervia]) must be maintained below 10 percent for 3 consecutive years of monitoring.

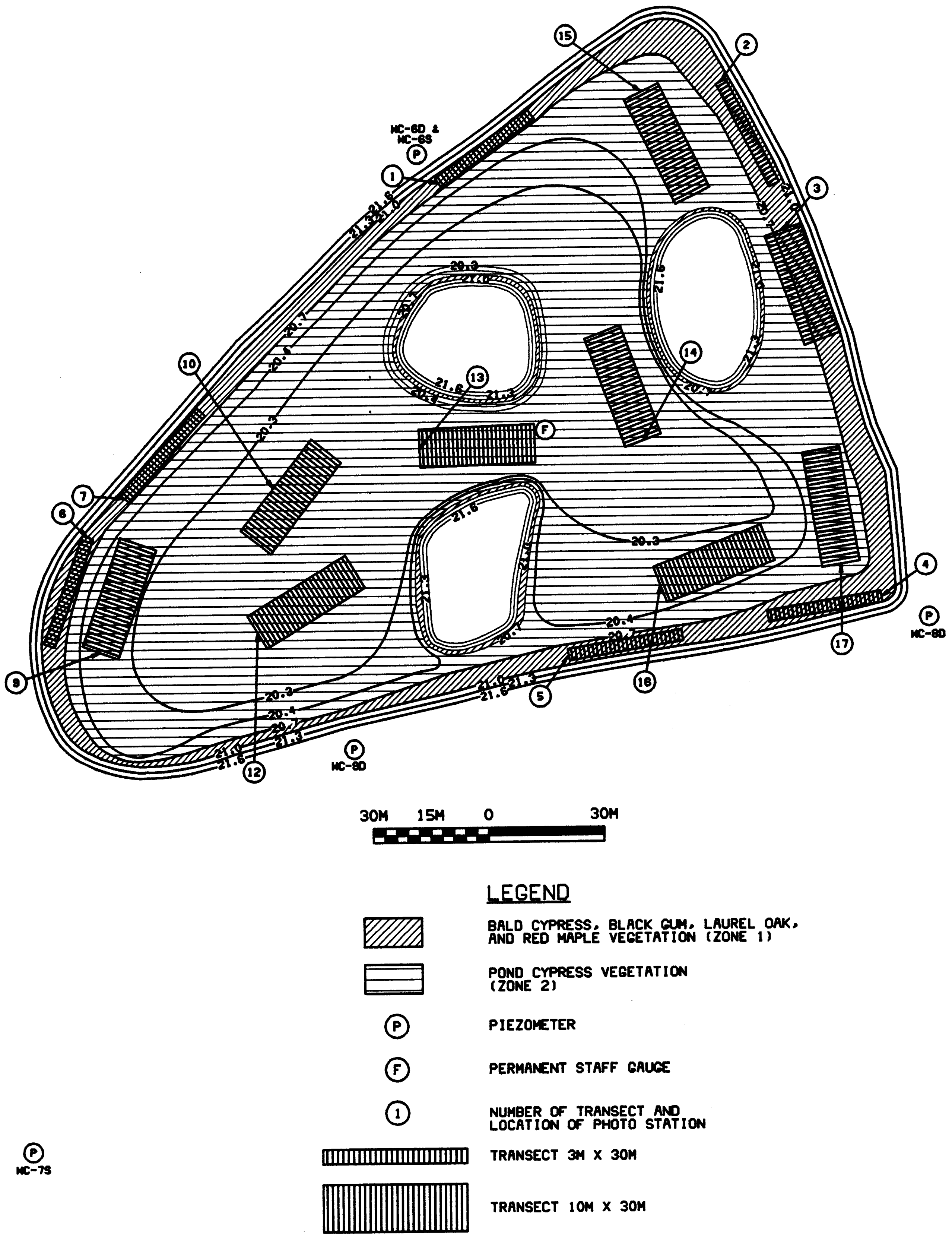

Figure 2 - Plan View

Tree Survivorship

To monitor the success of the establishment of the forested wetlands, fifteen 30 m (98.4 ft) long permanent belt transects were established in each wetland zone. Transect locations are shown on Figure 2. Wetland Zone 1 contained seven belt transects, 3 m or 10 m wide. Wetland Zone 2 contained eight belt transects, 3 m or 10 m wide.

To assess the survivorship rate of planted tree species, the living saplings were counted in April and September 1994. Baseline data were collected in October 1993. Data were pooled and average survivorship per species for each transect and wetland zone was calculated. Spring 1994 data are not shown.

Tree Growth

In each belt transect, sapling heights and diameters were measured for all living trees in April and September 1994 to determine whether or not planted trees exhibited growth during 1994. Newly planted trees in transects, such as red maple and laurel oak, were measured in September only. Heights were measured from the ground surface to the tallest part of the living tree (e.g., branch or apical meristem). Diameters were measured at breast height [dbh: 1.4 m above the ground surface]. Diameters of pond cypress were the only species quantified since all the other species have yet to reach sufficient height. Data were pooled and average heights and dbhs for each species per transect were calculated.

Ground Cover Vegetation

To determine the abundance of exotic vegetation at Wetland Zones 1 and 2, three 1 m^2 plots were placed at regular intervals (3 m, 15 m, and 27 m) along each 30 m belt transect. Percent canopy cover was visually estimated for each species either rooted within or extending into each plot. Ground cover data was collected during the September monitoring period. Cover classes were used to estimate coverage as described by Daubenmire (1959) with modifications by Bailey and Poulton (1968). The percent of non-vegetated area within each plot was also estimated. Mid-points of cover class values were used to calculate mean percent cover for each species. Data were pooled per transect and average cover of native and nuisance/ exotic vegetation per plot was calculated for each transect and Wetland Zone.

Hydrologic Conditions

Since 1992, hydrologic conditions have been monitored in the created wetland with the use of staff gauges and piezometers. Seven piezometers (three pairs and a single piezometer) were installed in December 1992 to assess pre-construction groundwater levels. Each pair of piezometers reaches 3.0 m (shallow: S) and 6.1 m (deep: D) below existing grade. Two depths were used to determine the extent of the clay hardpan layer at the wetland site (Black & Veatch, 1993). Construction of the wetland eliminated two pairs of piezometers as they were installed inside the wetland, which was reconfigured after piezometer installation. Two additional piezometers were installed to the southeast and southwest of the wetland in December 1993 to replace the lost piezometers (Figure 2). These piezometers reach 3.0 m below existing grade. A temporary staff gauge was installed in October 1993. A permanent staff gauge was installed in February 1994 (Figure 2).

RESULTS

Tree Survivorship

Wetland Zone 1

In 1993, Wetland Zone 1 was planted mostly with bald cypress and black gum (Table 1). The greatest number of bald cypress and black gum were planted in Transects 1 and 2, respectively. Transect 3 also had 24 pond cypress planted as the transect extended into Wetland Zone 2, below an elevation of 20.7 m. Average number of saplings planted per transect were 8.0 for bald cypress, 13.1 for black gum, and 3.4 for pond cypress.

By September 1994, 46.5 percent of saplings planted in Wetland Zone 1 had survived (Table 2). Most of the mortality can be attributed to black gum, which had a survivorship of 27.2 percent (Black & Veatch, 1995). Survival of bald cypress (58.9 percent) and pond cypress (91.7 percent) were higher than that of black gum.

Wetland Zone 2

Transects in Wetland Zone 2 were established only in areas planted with pond cypress (Table 1). The number of pond cypress planted per transect ranged from 26 in Transect 9 to 107 in Transect 15. The average number of pond cypress planted per transect in Wetland Zone 2 was 82.6.

By September 1994, only one sapling (in Transect 17) had died in the 8 transects located in Wetland Zone 2 (Table 2). The September 1994 survival rate is 99.8 percent.

Tree Replacement

In agreement with planting specifications, trees which died within the initial year after planting were replaced in the fall of 1994. A total of 488 seedlings/saplings including 45 pond cypress, 52 black gum, 180 red maple, and 171 laurel oak were planted at the wetland. With the tree replanting, tree densities per transect were returned to initial 1993 values (Table 1).

Table 1. Summary of tree density data after planting in 1993 and 1994.

DENSITY

Number of Saplings per Belt Transect in 1993 and 1994

	Belt Transect Number															
	1		2		3		4		5		6		7		Average	
Species Name	1993	1994	1993	1994	1993	1994	1993	1994	1993	1994	1993	1994	1993	1994	1993	1994
Bald Cypress	14	5	7	6	3	2	7	5	9	5	5	3	11	7	8.0	4.7
Black Gum	15	1	18	5	15	12	14	6	9	1	7	0	14	0	13.1	3.6
Pond Cypress	–	–	–	–	24	24	–	–	–	–	–	–	–	–	3.4	3.4
Red Maple	–	12	–	7	–	2	–	5	–	5	–	5	–	10	–	6.6
Laurel Oak	–	11	–	7	–	2	–	5	–	7	–	4	–	8	–	6.3

	Belt Transect Number																	
	9		16		10		12		13		14		15		17		Average	
	1993	1994	1993	1994	1993	1994	1993	1994	1993	1994	1993	1994	1993	1994	1993	1994	1993	1994
Pond Cypress	26	26	94	94	83	83	95	95	99	99	96	96	107	107	61	61	82.6	82.6

Table 2. Tree survival rate in September 1994

1994 Percent Survival Rate			
Belt Transect Number	Wetland Zone 1	Belt Transect Number	Wetland Zone 2
1	20.7 (6/29)	9	100 (26/26)
2	44.0 (11/25)	16	100 (94/94)
3	85.7 (36/42)	15	100 (107/107)
4	52.4 (11/21)	17	98.4 (60/61)
5	33.3 (6/18)	10	100 (83/83)
6	25.0 (3/12)	12	100 (95/95)
7	28.0 (7/25)	13	100 (99/99)
Average	46.5 (80/172)	14	100 (96/96)
		Average	99.8 (660/661)

Tree Growth

Wetland Zone 1

Changes in heights of bald cypress and black gum during 1994 showed no consistent trends (Table 3). Sample sizes for black gum in 1994 in most instances were too small for meaningful comparisons. In those transects where sample sizes were adequate for comparisons (Transects 2 and 3), mean heights of black gum dropped 22.8 percent from 0.8 to 0.6 m and 18.2 percent from 0.8 to 0.7 m, respectively. For bald cypress, average heights remained the same in all transects except Transects 3 and 7, where slight increases were observed.

Field observations support the 1994 height results (Black & Veatch, 1995). Observations show that the living black gum and bald cypress throughout Wetland Zone 1 had little or no growth during 1994. Neither species exhibited many new leaves during 1994, and leaves that did grow were stunted. Bald cypress also exhibited chlorotic and reddening of needles.

Table 3. Mean heights and diameters of tree seedlings/saplings in 1994.

Belt Transect Number	Tree Species	Mean Height (m)*		Mean DBH (cm)*	
		4/94	9/94	4/94	9/94
Wetland Zone 1:					
1	Black Gum	0.8 (8)	0.4 (1)		
	Bald Cypress	0.4 (7)	0.4 (5)		
	Laurel Oak	--	0.6 (11)		
	Red Maple	--	0.6 (12)		
2	Black Gum	0.8 (9)	0.6 (5)		
	Bald Cypress	0.3 (7)	0.3 (6)		
	Laurel Oak	--	0.6 (7)		
	Red Maple	--	0.7 (7)		
3	Black Gum	0.8 (13)	0.7 (12)		
	Bald Cypress	0.3 (3)	0.5 (2)		
	Laurel Oak	--	0.6 (2)		
	Red Maple	--	0.6 (2)		
	Pond Cypress	1.0 (24)	1.1 (23)		
4	Black Gum	0.9 (14)	0.7 (6)		
	Bald Cypress	0.4 (7)	0.4 (5)		
	Laurel Oak	--	0.7 (5)		
	Red Maple	--	0.7 (5)		

Table 3 (Continued). Mean heights and diameters of tree seedling/saplings in 1994.

Belt Transect Number	Tree Species	Mean Height (m)*		Mean DBH (cm)*	
		4/94	9/94	4/94	9/94
5	Black Gum	0.9 (7)	1.1 (1)		
	Bald Cypress	0.4 (9)	0.4 (5)		
	Laurel Oak	--	0.6 (7)		
	Red Maple	--	0.7 (5)		
6	Black Gum	0.7 (8)	--		
	Bald Cypress	0.3 (3)	0.3 (3)		
	Laurel Oak	--	0.6 (4)		
	Red Maple	--	0.7 (5)		
7	Black Gum	0.9 (5)	--		
	Bald Cypress	0.3 (8)	0.4 (7)		
	Laurel Oak	--	0.6 (8)		
	Red Maple	--	0.5 (10)		
Wetland Zone 2:					
9	Pond Cypress	1.4 (26)	1.5 (26)	0.3 (4)	0.3 (14)
16	Pond Cypress	1.6 (94)	1.6 (94)	0.5 (51)	0.8 (63)
10	Pond Cypress	1.6 (83)	1.7 (83)	0.5 (78)	0.5 (79)
12	Pond Cypress	1.6 (95)	1.7 (95)	0.5 (78)	0.8 (76)
13	Pond Cypress	1.4 (99)	1.5 (98)	0.5 (30)	0.5 (59)
14	Pond Cypress	1.5 (96)	1.6 (96)	0.5 (68)	0.5 (68)
15	Pond Cypress	1.4 (107)	1.5 (106)	0.3 (4)	0.5 (32)
17	Pond Cypress	1.1 (60)	1.2 (61)	0.3 (3)	0.5 (8)

*Sample sizes in parentheses.

Wetland Zone 2

In contrast to the results for bald cypress and black gum in Wetland Zone 1, pond cypress showed increases in height in seven of eight transects over the year (Table 3). Average increases in height per transect ranged from 1.0 to 1.8 cm, which represents a change in height of 4.0 to 10.7 percent. Tree diameters also increased, albeit less dramatically than heights. Average tree diameter in Transects 12, 15, 16, and 17 nearly doubled during the year. The trees in the remaining transects showed no increases in dbhs. However, the numbers of trees with measurable dbhs in all but one of these transects increased. Also, a number of trees were observed that had successfully produced one to several cones. In spite of the positive response of the pond cypress in Zone 2, this tree species exhibited yellowing and reddening of leaves in 1994 (Black & Veatch, 1995).

Ground Cover Vegetation

Wetland Zone 1

In 1993, no exotic and nuisance vegetation were found in any of the transects of Wetland Zone 1 (Table 4). However, a few stems of cattails and primrose willow were observed in this wetland area. This undesirable vegetation was removed. Similar results were observed in 1994 (Black & Veatch, 1995). Cattail abundance has increased since 1993, but is estimated at less than 1 percent outside the transects.

Other vegetation was also sparse (Table 4). In 1993, only Transects 6 (10.8 percent) and 7 (18.3 percent) had a mean cover of greater than 2.5 percent, while mean cover in 1994 was greater than 10 percent in each transect. Consequently, average cover increased from 5.9 to 20.8 percent over time. In 1993, the most prevalent species, Cyperus retrorsus, had a mean canopy cover ranging from 0.3 percent to 18.5 percent and an overall average of 5.1 percent cover (Black & Veatch, 1994). In 1994, the most abundant species was Bermuda grass (Cynodon dactylon) which ranged from 0.3 to 11.0 percent in average cover. Bermuda grass was one of four grasses hydroseeded along the shore line to control erosion. Other species occurring in plots included umbrella sedge (Cyperus polystachyos), Mohr's thoroughwort (Eupatorium mohrii), Ludwigia repens, rustweed (Polypremum procumbens), nutrush (Scleria sp.), Sesbania sp., sweet broom (Scoparia dulcis), Rhynchospora spp., Juncus spp., and dog fennel (Eupatorium capillifolium). No woody vegetation was observed in any of the plots during both years of monitoring (Black & Veatch, 1994, 1995); however, a number of wax myrtle (Myrica cerifera) seedlings were observed in Transects 2, 3, 6, and 7 in 1994.

Wetland Zone 2

In 1993, no exotic and nuisance vegetation were found growing in any transect (Table 4) nor was any observed outside the transects. In contrast, cattails were found in and outside of the transects by 1994. However, cattails were found growing only in Transects 15 and 17, where average cover was 1.0 percent and 5.0 percent, respectively (Table 4). Overall, cover of cattails in the transects was 0.8 percent on average.

Table 4. Summary of vegetation cover at the Wetland Creation Site

Belt Transect Number	Mean Percent Cover of Vegetation					
	Exotic/Nuisance Species		Other Species		Total	
	1993	1994	1993	1994	1993	1994
Wetland Zone 1						
1	0	0	2.5	22.5	2.5	22.5
2	0	0	2.5	10.8	2.5	10.8
3	0	0	2.5	10.8	2.5	10.8
4	0	0	2.5	10.8	2.5	10.8
5	0	0	2.5	10.8	2.5	10.8
6	0	0	10.8	30.8	10.8	30.8
7	0	0	18.3	38.3	18.3	38.3
Average	0	0	5.9	20.8	5.9	20.8
Wetland Zone 2						
9	0	0	0	0	0	0
16	0	0	0	0	0	0
10	0	0	0	0	0	0
12	0	0	0	0	0	0
13	0	0	0	0	0	0
14	0	0	0	0	0	0
15	0	1.0	2.5	2.5	2.5	3.5
17	0	5.0	2.5	0.7	2.5	5.7
Average	0	0.8	0.6	0.3	0.6	1.2

Very little non-exotic and nuisance vegetation was found in the transects in 1993 and 1994. Only Transects 15 and 17 were vegetated (Table 4). Mean cover was 2.5 percent for each transect in 1993 and 2.5 and 0.7 in 1994. Only two herbaceous species, Cyperus sp. and Rhynchospora sp., occurred in these transects in 1993 (Black & Veatch, 1994). In 1994, more species were found in Transects 15 and 17 including Juncus dichotomus, Bermuda grass, Hedyotis uniflora, Eclipta alba, and Rhynchospora fascicularis (Black & Veatch, 1995). Other vegetation found in Zone 2 included pickerel weed (Pontederia cordata), soft rush (Juncus effusus), Mikania scandens, Pluchea rosea, dotted smartweed (Polygonum punctatum), mock bishop's-weed (Ptilimnium capillaceum), Ludwigia hirtella, Carolina willow (Salix caroliniana). In addition, duckweed (Spirodela sp. and Lemna sp.) and algae occurred throughout the inundated areas of the wetland.

Hydrologic Conditions

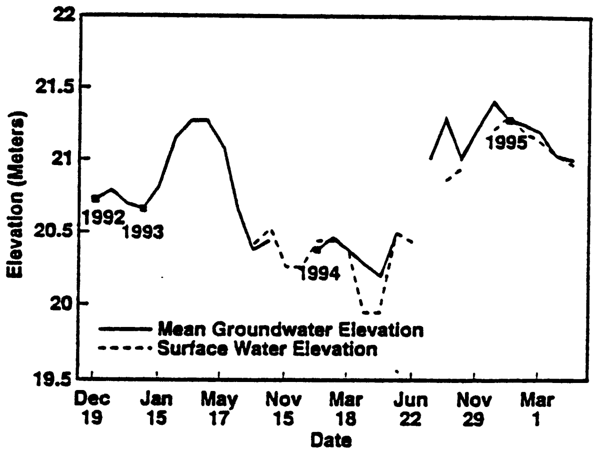

Figure 3. Mean groundwater and monthly surface water elevations.

1993. During 1993, groundwater levels fluctuated from a mean annual high of 21.2 m in late March and early April to a mean yearly low of 20.4 m in early October (Figure 3). A complete dry down of the wetland surface water elevation of 66.5 ft occurred in November and December during a 56 day drought.

1994. Groundwater elevations ranged from a mean annual low of 20.4 m in April to a mean annual high of 21.4 m in December. Groundwater and surface water elevations were high relative to previous years because of an inordinate amount of rain during the summer and fall of 1994.

Groundwater surpassed 21.3 m on average on two occasions, a level not attained at the wetland site since monitoring began in December 1992. The entire wetland was dry only during a short time in April. Furthermore, Wetland Zone 1 was inundated for nearly 4 months (September to December) in 1994. In spite of this long period of inundation, Wetland Zone 1 lacked water for the first 6 to 7 months of 1994.

DISCUSSION

The wetland creation at the Cane Island Project has been highly successful; results which contrast with many mitigation projects in Florida (FDEP, 1991). Only 6.3 percent of wetland creation projects required by the FDEP from 1985 to 1990 were in full compliance with permit mitigation requirements. In contrast, success criteria for this wetland creation project are being met.

In summary, based on the September 1994 transect data, 89.0 percent of the trees planted in 1993 survived at the wetland creation site through 1994. This rate of survivorship is above the required success rate of 80 percent. The high success rate of the planting is primarily because of the survival of pond cypress (99.8 percent) in Wetland Zone 2. Similar results have been observed over the entire wetland where 99.9 percent or 4,453 of 4498 of pond cypress have survived since planting in 1993.

In contrast, the trees planted in Wetland Zone 1, black gum and bald cypress, have exhibited much lower survival rates (27.2 percent and 58.9 percent, respectively) for the first year of monitoring. The relative low rates of survivorship can be ascribed to the droughty conditions at the edge of the wetland during the last year. A 56 day drought soon after planting in October and November 1993 severely stressed the black gum and bald cypress, causing high rates of mortality. Lower water levels during the first half of 1994 also contributed to mortality. The presumed lower nutrient fertility of the substrate (a clay hardpan) may have also have contributed to the low survivorship. Planned fertilization of each seedling and sapling should stimulate growth and increase the chances of survival.

In addition to high survival rates, pond cypress exhibited increases in height and dbh. Several pond cypress also produced cones in 1994. These results are promising given the low fertility of the clay hardpan underlying the wetland site.

Control of exotic and nuisance vegetation has also been successful. Since September 1993, the average exotic and nuisance vegetation cover slightly increased from 0 percent to 0.4 percent. Similar results have also been observed outside of the transects. Cattails have continued to increase since the September 1994 monitoring but are being controlled with herbicide spraying.

Native vegetation has also increased in Wetland Zone 1. Desireable species include marsh pennywort (Hydrocotyle umbellata), dotted smartweed, Pluchea rosea, soft rush, Juncus dichotomus, and Rhynchospora spp. Continued colonization is expected in the future with continued control of exotic and nuisance vegetation. Little change was observed in Wetland Zone 2.

ACKNOWLEDGEMENTS

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ABOVE-GROUND GROWTH AND RADIUM-226 UPTAKE OF COVER CROPS GROWN ON PHOSPHOGYPSUM AND PHOSPHATIC CLAY FROM A PHOSPHATE MINE IN NORTH CAROLINA

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ABSTRACT

German millet (Cynodon dactylon L.) and common bermudagrass (Setaria italica L.) were grown in varying mixtures of phosphogypsum and phosphatic clay to investigate plant growth and the ^{226}Ra uptake on restored phosphate mined lands. These cover crops were grown in the greenhouse in pots containing gypsum-clay mixtures of 0:1, 1:0, 1:1, 2:1, 3:1, and 4:1 to determine the optimal ratio of gypsum to clay for successful colonization and low ^{226}Ra uptake. Dry matter production by german millet and common bermudagrass (harvested after 50 days) increased with increasing gypsum in the mixture, with the best growth in the 4:1 gypsum-clay mixture. There was no significant difference in cover crop uptake of ^{226}Ra with an increase in gypsum in the mixtures and uptake for both species was low across all treatments, $\leq$ 0.90 pCi/g. Uptake of ^{226}Ra by cover crops grown on a gypsum-clay mixture in a field restoration site was not significantly different from uptake by cover crops grown on natural soil. Our results suggest that increasing the amount of gypsum in the mixture, at least up to 4:1, will enhance the growth of cover crops without increasing ^{226}Ra uptake.

INTRODUCTION

Phosphorus, an essential element for plants and animals, is mined by removal of an overburden layer to expose the underlying deposit of clay, quartz sand, and phosphate rock (PR). Phosphoric acid, which is used to produce superphosphate fertilizer, is extracted from the PR with H_2SO_4 and one of the byproducts is phosphogypsum (Tisdale, et al., 1985). One byproduct of the mining process is phosphatic clay which makes up a third of all material excavated at phosphate mines in Florida and often requires large areas for dewatering (Mislevy, et al., 1990). Phosphogypsum, a byproduct of the extraction of phosphoric acid through the acidification of PR with sulfuric acid, is also produced in large quantities and must be used or stockpiled (Tisdale, et al., 1985). In North Carolina, phosphogypsum is mixed with clay tailings to backfill mining pits as a first step in site restoration. However, there is more gypsum material available than clay and excess gypsum must be stored on site. Phosphogypsum and phosphatic clays from mines in Florida (Mislevy, et al., 1989) and North Carolina (Broome, et al., 1990) provide generally adequate media for plant growth with low concentrations of phytotoxic materials, as well as high concentrations of plant nutrients (except for N, Mn, and Cu).

Phosphate rock from Florida mines contain high levels of ^{226}Ra (Menzel, 1968), and measurable amounts of ^{226}Ra have been found in phosphogypsum and clay tailings from a North Carolina phosphate mine (Broome, et al., 1990). Studies of ^{226}Ra uptake by forage and crop plants grown on soil amended with phosphogypsum from PR mining in Florida revealed no significant uptake of ^{226}Ra in vegetables (Million, et al., 1994) or grain crops (Mays and Mortvedt, 1986). However, forage crops grown on phosphatic clay had ^{226}Ra concentrations three times those of forage crops grown on unmined soil (Mislevy, et al., 1990). Likewise, ^{226}Ra concentrations in biomass crops grown on phosphogypsum/quartz sand tailings were six times those of crops grown on an unmined Spodosol (Mislevy, et al., 1989). No information could be found concerning plant uptake of ^{226}Ra on restored phosphate mining operations in North Carolina.

Two warm season grasses, common bermudagrass (Cynodon dactylon L) and german millet (Setaria italica L.) are often used during the initial restoration stages at PR mines located in North Carolina. Cover crops hasten the dewatering of the clay-gypsum ponds while improving soil conditions by adding organic matter that is beneficial for future tree growth (Tisdale, et al., 1985). Because production of phosphoric acid from PR produces much more gypsum than mining produces clay, the objective of this study is to determine if this excess gypsum can be used in higher ratios than the 2:1 and the 3:1 gypsum-clay mixtures that are currently used. A greenhouse experiment was used to examine ^{226}Ra uptake, and the growth of german millet and common bermudagrass grown on gypsum and clay mixed in ratios of 1:0, 0:1, 1:1, 2:1, 3:1, and 4:1. Soil samples were analyzed for pH, nutrients, and ^{226}Ra to investigate the relationship between soil fertility, cover crop growth, and ^{226}Ra uptake. We also measured ^{226}Ra in cover crops and trees grown on a field restoration site and on natural soils to assess the ^{226}Ra uptake in restoration sites containing 1:1-2:1 gypsum-clay mixtures. This information is important in determining the optimal ratio of gypsum to clay for successful cover crop growth along with low uptake of ^{226}Ra.

MATERIALS AND METHODS

Phosphogypsum and phosphatic clay were collected from PR mining and chemical processing operations at PCS Phosphate (formerly Texasgulf) in Beaufort County, North Carolina; brought back to the greenhouse, and mixed in a cement' mixer in 1:0, 0:1, 1:1, 2:1, 3:1, and 4:1 gypsum-clay ratios (dry weight). Three hundred gallons of each soil mixture were placed in 4-quart pots and seeded with german millet (75 pounds/acre) or common bermudagrass (24 pounds/acre) on June 1, 1994. The treatments consisted of 2 grasses, 4 replicates, and 6 soil mixtures (48 seeded pots), in a completely randomized design. The pots were sprayed with distilled water and covered with plastic wrap once each day until the seeds germinated. Upon germination, the plastic was removed and the pots were watered with distilled water once each day. Nitrogen was applied as ammonium nitrate in solution at the initiation of the experiment (June 1, 1994) at the rate of 30 pounds/acre N, and at the midpoint of the study (June 28, 1994) at 70 pounds/acre N.

On July 20, 50 days after the experiment was initiated, above-ground plant material was harvested from each plot. Plant material was dried at 70°C overnight, ground, and sieved through a 20 mm mesh. Soil samples also were collected from each pot, air dried, ground and sieved (2 mm mesh).

^{226}Ra was measured in plant tissue harvested from the greenhouse study and from a field restoration site (R-1) at the PCS Phosphate mining company, Aurora, North Carolina. Soil at the R-1 site consisted of a 1:1-2:1 gypsum-clay mixture. Cover crops (crimson clover, annual ryegrass, alfalfa, yellow sweet clover, and winter rye) and tree species (red maple, bald cypress, green ash, sweet gum, and water oak) were sampled at the R-1 reclamation site. We also collected cover crop and tree leaf samples grown on natural soil for comparison with the field restoration site.[1] Cover crop species seeded in the fall of 1989 were harvested on April 3, 1990. Leaf samples were collected in the fall of 1990. Tree leaf and cover crop tissue from the field and greenhouse studies were dried and sealed in 50 x 9 Falcon petri dishes for 21 days to reach secular equilibrium. Gamma emissions of the ^{226}Ra decay product ^{214}Bi were measured by an EG&G Ortec high-purity germanium crystal (Oak Ridge, Tennessee).

Plant tissue and soils were analyzed for N, P, and Mg to assess potential mineral nutrient deficiencies associated with the gypsum-clay blend. A nitric-perchloric acid digest was used for the total P and Mg analysis of plant material (Sommers and Nelson, 1972). Total P and Mg analysis in soils was performed by acid digestion using the method of hydrofluoric acid in a closed vessel (Lim and Jackson, 1982). Total N in soils and plant tissue was determined using a Perkin Elmer (2400) Series II CHNS/O analyzer. Soil pH was determined in a 2:1 deionized water-soil slurry using a glass electrode and a Beckman 70 pH meter. Total P content was determined colorimetrically by the ascorbic acid blue method (Murphy and Riley, 1968). The NBS standards, buffalo river sediment (No. 2704) and plastic clay (No. 98B), were used to determine the efficiency of digests and instrument calibration accuracy.

STATISTICAL ANALYSIS

We used one-way analysis of variance (ANOVA) to test for differences in cover crop yield and plant tissue concentrations of nutrients and ^{226}Ra. The means comparison test, Ryan-Einot-Gabriel-Welsch Multiple F-test (REGWF), was used to test between the six treatments for differences in mean soil and plant concentrations of ^{226}Ra and nutrients. A one-tailed t-test was used to test for differences in ^{226}Ra uptake between bermudagrass and german millet. The Statistical Applications System (SAS) was used to compute means, standard deviations, standard errors, t-tests, F-tests, and the REGWF test (SAS Institute, Cary, North Carolina). A one-tailed Student's t-test was used to compare whether ^{226}Ra activities were greater for cover crops and leaf samples from trees grown on the R-1 reclamation site than they were for plants grown at a control site. All tests of significance were made a=0.05.

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[1]Field samples collected during the PCS Phosphate Cover Crop and Tree Species Screening Experiment were provided by Dr. Steven Broome of North Carolina State University.

RESULTS / DISCUSSION

Cover Crop Yield

With the exception of the pure clay mixture, above-ground biomass production by both species increased significantly with increasing ratios of gypsum to clay up to 4:1 (Figure 1). Neither bermudagrass nor german millet seeds germinated in the pure gypsum treatment. The low pH conditions may account for the inability of the seeds to germinate in the pure gypsum mixture (Table 1). In the case of both species, the best growth occurred on the 4:1 gypsum-clay mixture, and of the treatments in which the seeds germinated, the 1:1 mixture had the poorest growth.

Differences in growth among treatments could not be explained by soil or plant nutrient content (Table 2). The lowest growth, which occurred in the pure clay and the 1:1 gypsum-clay mixture, could not be explained by nutrient deficiency. In fact, soil and plant concentrations of nutrients such as N, P, and Mg increased with increasing clay in the mixtures (Table 2).

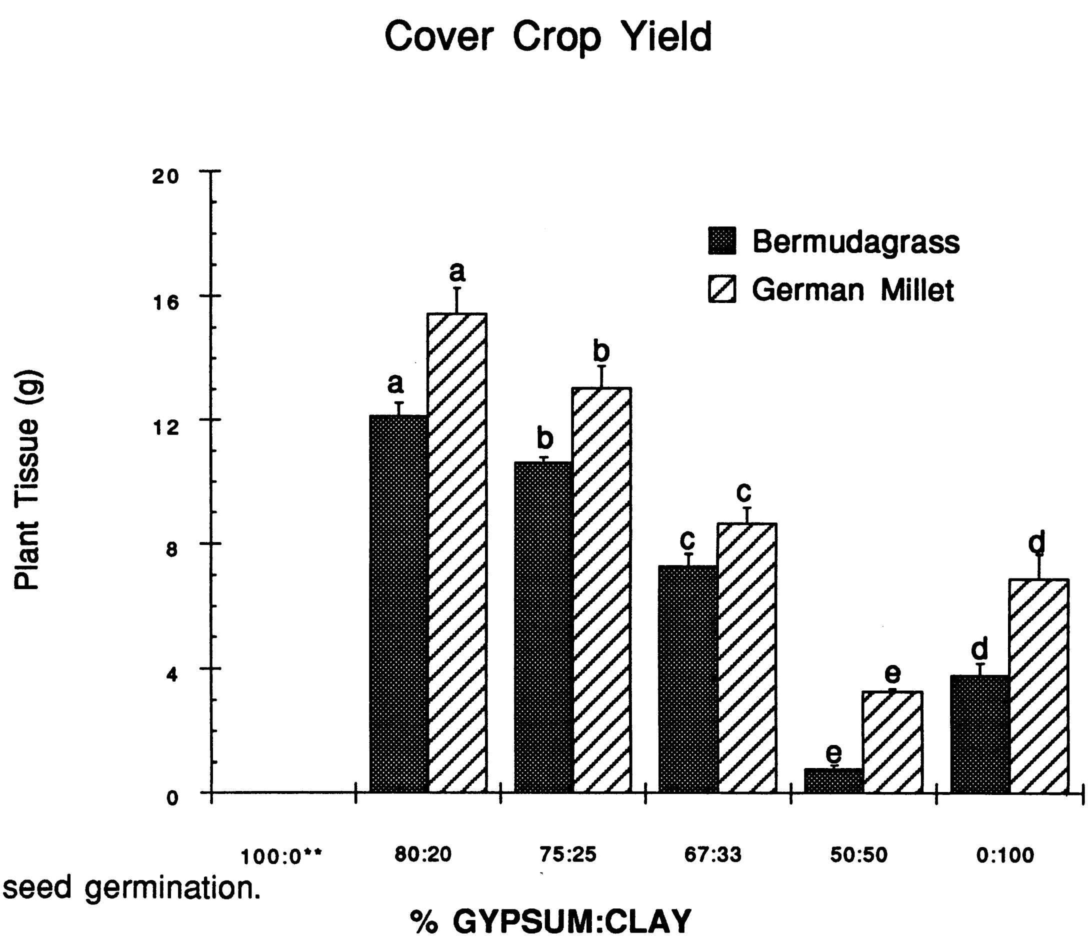

Figure 1. Above-ground production of german millet and bermudagrass is gypsum-clay mixtures. Treatment means (for each species) separated by the same letter are not considered significantly different (a=0.05).

Table 1. Mean soil pH or gypsum-clay mixtures seeded with bermudagrass and german millet. Means separated by the same letter are not considered significantly different (a=0.05).

% GYPSUM:CLAY	pH* POTS WITH BERMUDAGRASS	pH* POTS WITH GERMAN MILLER
0:100	7.4 a	7.4 a
50:50	7.4 a	7.2 ab
67:33	6.7 b	6.7 ab
75:25	6.3 b	5.1 ab
80:20	5.7 c	5.6 b
100:0	2.6 d	2.7 c

*Soil pH measured in a 2:1 deionized water-soil mixture.

Table 2. Total N, P, and Mg content of soil samples and plant tissue from pots seeded with bermudagrass and german millet. Means within the same column separated by the same letter are not significantly different (a=0.05), according to the Ryan-Einot-Gabriel-Welsch Multiple F-test.

BERMUDAGRASS

	PLANT TISSUE*			SOIL*		
% GYPSUM:CLAY	N%	P%	Mg%	N%	P%	Mg%
100:0	*	*	*	0.02 d	0.25 a	<0.04 d
80:20	1.21 c	0:13 c	0:13 c	0.06 c	0.33 d	0.45 d
75:25	1.34 bc	0:15 bc	0:18 b	0.08 c	0.41 c	0.52 cd
67:33	2.00 a	0:20 ab	0:22 ab	0.14 b	0.44 c	0.64 c
50:50	2.13 a	0:20 a	0:22 a	0.14 b	0.57 b	1.13 b
0:100	1.90 ab	0:14 c	0:22 ab	0.17 a	0.92 a	2.18 a

Continued

Table 2 (Continued)

GERMAN MILLET

	PLANT TISSUE*			SOIL*		
% GYPSUM:CLAY	N%	P%	Mg%	N%	P%	Mg%
100:0	*	*	*	0.05 d	0.23 a	<0.04 d
80:20	0.63 ab	0:13 a	0:30 b	0.08 cd	0.35 d	0.41 d
75:25	0.57 b	0:06 b	0:30 b	0.09 cd	0.43 c	0.49 cd
67:33	0.87 b	0:12 a	0:35 ab	0.11 bc	0.47 c	0.63 c
50:50	0.96 a	0:09 a	0:39 a	0.14 ab	0.60 b	2.19 b
0:100	0.77 ab	0:05 b	0:40 a	0.17 a	0.96 a	2.18 a

*Mean (N=4) element concentrations of soil samples and plant tissue.

Physical, rather than chemical characteristics of the gypsum-clay mixtures, may better explain growth differences. Seeds in the pure clay mixture were the first to germinate. However, growth in the pure clay and the 1:1 gypsum-clay mixture was inhibited after germination seed tubers had difficulty penetrating the soil surface. The combination of late germination and poor soil penetration by the seed tubers in the 1:1 gypsum-clay mixture may explain why this treatment exhibited the poorest growth. Plant roots easily penetrated the soil surface in the 2:1, 3:1, and 4:1 gypsum-clay treatments. Thus, an increase in the concentration of gypsum from the 1:1, 2:1, 3:1, and 4:1 gypsum-clay mixtures may have improved growth by improving soil physical properties such as structure and porosity.

^{226}Ra CONCENTRATIONS

There was no significant uptake of ^{226}Ra by cover crops grown on the gypsum-clay blends in the greenhouse study (Table 3).

Table 3. Mean Ra-226 activities (pCi/g) is bermudagrass and german millet in varying mixtures of gypsum and clay.

	BERMUDAGRASS		GERMAN MILLET	
% GYPSUM: CLAY	BIOMASS	SOIL* Ra-226 (pCi/g)	BIOMASS	SOIL* Ra-226 (pCi/g)
100:0	**	9.8 b	***	9.93 a
50:50	0.31 a	10.76 ab	0.11 a	11.02 a
67:33	0.87 a	10.46 ab	0.10 a	10.17 a
75:25	0.75 a	12.74 a	0.15 a'	11.33 a
80:20	0.27 a	10.16 ab	0.31 a	11.14 a
0:100	0.89 a	12.59 ab	0.24 a	12.32 a

* Mean (n-4) Ra-226 activity (pCi/g) of soil in which grass was seeded.
** No seed germination.
Means within the same column separated by the same letter are not considered significantly different (a=0.05) according to the Ryan-Einot-Gabriel-Welsch Multiple F-test.

Uptake of ^{226}Ra by both bermudagrass and german millet was low (≤ 0.90 pCl/g). Generally, ^{226}Ra uptake by bermudagrass was greater than that by german millet. ^{226}Ra activities were somewhat lower in the gypsum as compared to clay (Table 3.) However, no significant (r^2=0.10) relationship was found between ^{226}Ra activities in plants and activities in the gypsum-clay mixtures. Average ^{226}Ra uptake by bermudagrass (0.62 pCi/g) and german millet (0.18 pCi/g) grown on the gypsum-clay mixtures was similar to ^{226}Ra uptake (0.70 pCl/g) by grasses grown on phosphatic clay from a Florida phosphate mine (Mislevy, et al., 1990).

No significant increase in the ^{226}Ra uptake by cover crops grown on the R-1 reclamation site compared to crops grown on natural soil (Table 4) was observed. However, mean ^{226}Ra activity (pCi/g) for alfalfa, annual ryegrass, and crimson clover was somewhat higher in plants grown on the R-1 test site than on natural soil. ^{226}Ra activities in tree leaf samples from trees grown on natural soils and on the R-1 reclamation site at the PCS Phosphate mining company were at, or below, the detection limit (0.1 to 0.2 pCi/g) of the gamma counter. Uptake of ^{226}Ra by cover crops grown on the R-1 reclamation site was low, compared to ^{226}Ra activity in crops grown in the gypsum-clay blend (Tables 3 and 4).

Table 4. Mean Ra-226 activity (pCi/g plant tissue) of cover crops grown on gypsum/clay blend of the R-1 field site and on natural soil. Uptake of Ra-226 on the blend is not significantly different (a=0.05) from the natural soil. Ra-226 activities were corrected for the branching ratio (46%) and the counting efficiency of the detector (2.3%).

		RADIUM-226 (pCi/g plant tissue)	
COVER CROP SPECIES		GYPSUM/CLAY BLEND	NATURAL SOIL
Crimson clover	(n=4)	1.56	0.97
Annual ryegrass	(n=4)	1.60	1.05
Alfalfa	(n=4)	1.56	0.80
Yellow sweet clover	(n=4)	0.30	0.42
Winter rye	(n=4)	0.72	1.67

CONCLUSION

We observed a significant increase in growth by bermudagrass and german millet as greater concentrations of gypsum were incorporated into the gypsum-clay treatments. The highest yield obtained for both species was in the 4:1 gypsum-clay mixture. This suggests that the use of even higher concentrations of gypsum in soil mixtures may be accompanied with greater growth.

Uptake of ^{226}Ra by cover crops grown on the gypsum-clay blend of the R-1 reclamation site was not significantly greater than that by crops grown in the natural soil. ^{226}Ra activity for both grass species was unaffected by varying proporation of gypsum to clay and overall was low compared to that of the soil.

The results of this study suggest that gypsum to clay mixtures in ratios greater than four to one may be used in the backfill of PSC Phosphate restoration sites to improve plant growth with no increase in plant uptake of ^{226}Ra.

ACKNOWLEDGEMENTS

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ENCASED REPLANTING
A RED MANGROVE REPLENISHMENT METHODOLOGY

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ABSTRACT

Inadequacies in conventional red mangrove (Rhizuphora mangle) replenishment methods are primarily a result of their sensitivity to water depth, tidal action, and wave activity. A major problem in successful planting is the difficulty in finding suitable locations with adequate and appropriate environmental conditions favorable to the rooting and sustenance of the mangrove during its early stages of development. To have any potential of establishing thriving mangroves when using conventional methods, the seedlings must be planted only in areas adequately shielded from any substantial wave action or upland run-off. These conditions translate into restrictions not simply on the geographic location of a potential replenishment project, but also on the relative size and range of any replanting. Many areas that would be desirable for mangrove planting present formidable factors that prohibit the successful introduction of the tree.

The necessity of implementing mangrove replenishment projects is supported by the documented reduction in mangroves throughout Florida's estuary systems. Increases in population, water-front development, agriculture, boating and related activities have resulted in significant increases in the types and quantities of pollutants reaching intracoastal and coastal waters. Additionally these factors have contributed to a significant decline in mangrove habitat necessary to maintain commercial and recreational fisheries. Therefore, the importance of mangroves to a healthy marine ecology has dramatically increased. As natural members of estuary systems, mangroves mitigate the environmentally adverse and destructive effects of development and consequential pollution. In an effort to promote mangrove replenishment on a wide geographic basis an alternative planting method, called "Encased Replanting", has been developed. This new planting method is not subject to the limitations of conventional techniques.

Encased Replanting applies new methodology and technique to mangrove replenishment. With employment of the encased method, mangroves can be established in areas with significant tidal action, wave activity, and upland run-off. Mangroves offer a logical contribution to coastline protection, estuary restoration and a healthy marine environment. The encased method effectively enables the replenishment of mangroves where conventional techniques can not succeed.

INTRODUCTION

Ecological Importance

Mangrove trees are an indigenous species to Florida and a major contributor to the state's marine environment. The mangrove tree is a halophyte, a plant that thrives in salty conditions. It has the ability to grow where no other tree can, thereby making significant contributions that benefit the environment. Their coverage of coastal shorelines and wetlands provides many diverse species of crustacea, fish, birds, and mammals a unique, irreplaceable habitat. Mangroves preserve water quality and reduce pollution by filtering suspended material and assimilating dissolved nutrients.

The tree is the foundation in a complex marine food chain and the detrital food cycle. The detrital food cycle was discovered by two biologists from the University of Miami, Eric Heald & William Odum, in 1969 (Odum and McIvor). As mangrove leaves drop into tidal waters they are colonized within a few hours by marine fungi and bacteria that convert difficult to digest carbon compounds into nitrogen rich detritus material. The resulting pieces covered with microorganisms become food for the smallest animals such as worms, snails, shrimp, mollusks, mussels, barnacles, clams, oysters, and the larger commercially important striped mullet. These detritus eaters are food for carnivores including crabs and fish; subsequently birds and game fish follow the food chain, culminating with man. Many of these species, whose continued existence depends on thriving mangroves, are endangered or threatened.

It has been estimated that 75% of the game fish and 90% of the commercial fish species in south Florida rely on the mangrove system (Florida Wildlife, 1984). The value of red mangrove prop root habitat for a variety of fishes and invertebrates has been quantitatively documented. Data suggest that the prop root environment may be equally or more important to juveniles than are sea grass beds, on a comparable area basis (Thayer et al., 1987). Discovery of the importance of mangroves in the marine food chain dramatically changed the respective governmental regulation of coastal land use and development.

The beneficial effects mangroves have on the marine ecology are summarized as follows :

- Basis of a complex marine food chain;
- Creation of breeding habitat;
- Establishment of restrictive impounds that offer protection for maturing offspring;
- Filtering and assimilating pollutants from upland run-off;
- Stabilization of bottom sediments;
- Water quality improvements; and
- Protection of shorelines from erosion.

Framework for Replenishment

Since mangroves grow along the coastlines, lagoons, and estuaries of the state, their domain has been significantly reduced by the dredging, filling, and bulkheading of waterfront property for development. In addition, large concentrations of mangroves were isolated from lagoon waters in the 1950's, by the construction of dikes that established compounds for mosquito control. These compounds continue to be used for controlling mosquito populations through the managed flooding of wetlands during the summer months when tidal waters are low and insect reproductive activity is high. The dikes now prevent the free flow of water between the mangrove wetlands and the lagoon, thereby denying the marine ecology the full benefits of the mangrove population.

Development and population growth will continue to have a negative impact on habitat necessary to maintain commercial and recreational fisheries. Based on analysis of aerial photos from the 1940's, 1950's, and 1980's, one study of the Indian River Lagoon, from Sebastian Inlet south to Vero Beach, documented an 86 percent decline in the availability of mangrove habitat to fisheries over a forty-year period (Florida Marine Research Institute Pamphlet).

Conventional Planting

Literature written on mangrove replanting recommends that seedlings be first planted in containers in order to allow the development of a healthy root system prior to setting the plants out at a replanting site. It is noted that when unrooted seedlings are planted in areas directly on the site, they are most often washed away (Stevely and Rabinowitz, 1982). In some instances stakes are suggested to help secure rooted seedlings to reduce this problem; however, it is generally advocated that plants be placed just above the high tide line since wave action can disturb or displace the seedlings. The transplanting of already established mangrove trees has also been suggested as a desirable approach since this greatly reduces the time required to establish a healthy stand. But even when mangroves up to 4 feet in height are transplanted it is recommended that existing mangroves in the area be used as a guide for placing the new plantings relative to the water line.

Limitations of Conventional Methods

Inadequacies in conventional replanting methods are primarily a result of their sensitivity to water depth, tide, and wave activity. As evidence of this sensitivity, various replanting projects over the last several years have achieved only very limited success. A major problem in successful replanting is the difficulty in finding suitable locations with adequate and appropriate environmental conditions favorable to the rooting and sustenance of the mangrove during its early stages of development. To have any potential of establishing thriving mangroves when using conventional methods, the seedlings or transplants must be placed into shallow water only a few centimeters in depth and only in areas adequately shielded from any substantial wave action or upland run-off. These conditions translate into restrictions not simply on the geographic location of a potential replenishment project but also on the relative size and range of any replanting. Therefore, the search for alternate methods not subject to the limitations of conventional replanting was initiated and is the purpose of this investigation.

Criteria for an Alternative

To overcome shortcomings that can severely restrict the scope of replenishment projects, the fundamental criteria set for the development of alternate replanting methods were established :

1. Ability to directly use readily available, non-rooted, seedlings for planting.

2 Capability of replanting in waters with seasonal high tide up to four feet in depth.

3. Capability of replanting in areas with substantial wave activity and/or upland run-off.

4. High ratio of planted seedlings reaching maturity.

Encased Replanting

In order to overcome the deficiencies in existing replanting techniques, encased replanting was developed as a more productive and adaptable alternative to current techniques. The method focuses on isolating the seedling into a controlled environment at the actual replanting site. The encasement artificially creates an environment favorable to the seedling's initial development while protecting the plant long enough to become well established. The isolation physically separates the seedling from surrounding conditions that are unfavorable to early development of the tree.

By segregating the seedling from harsh environmental factors the encasement provides protection for the seedling during its formative stages and continues through the first three years of growth. By the third year, the young red mangrove starts to sprout its aerial roots which will physically secure the plant to the bottom and ensure its long term survival. The development of aerial roots is a crucial step that ensures viability of the developing tree. The aerial roots provide for the exchange of gases needed for respiration and will enable the tree to root, even in anaerobic sediment. Once aerial roots extend into the bottom they will provide adequate protection from displacement and ensure the continuing subsistence of the plant. Aerial or prop roots are a distinctive root structure of the red mangrove and a characteristic that makes the encased method of planting an effective means of replenishment (Figure 1). The aerial root is the mechanism that will ultimately secure the plant making it resistant to environmental factors that would under normal conditions prohibit development at many potential replenishment sites. The support

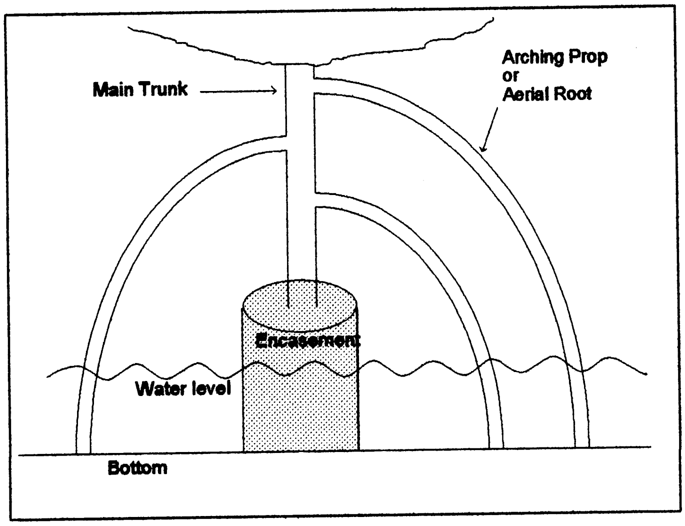

Figure 1. Encased Planting Method

provided by the aerial roots will hold the trunk of mature trees above the mean high tide water level. Over time an intricate maze of roots will protrude from the stem and end the plant's dependence on the encasement for its survival.

In environments that are unfavorable to the seedling's initial development, the encased method will allow the tree to reach a point where its own infrastructure can overcome factors that are hostile to the immature tree. As the plant matures it will establish a dense foundation of prop roots and will continue to develop independent of the encasement. This foundation will enable the mangrove to grow into a healthy, self-supporting tree.

Empirical Evidence

Results from research and field tests conducted over a five year period, from 1990 to the present, involving over 100 test seedlings, demonstrated the encased method to be the most successful alternative in satisfying goals established for the project. The field study proved it to be the only alternative method that effectively achieved all goals, particularly that of replanting in areas with high tide depths of several feet.

An interesting planting application is along bulkheaded properties where mangroves were removed when the sea walls were installed (Figure 2). Although the removal of mangroves when installing sea walls is prohibited today, it was a common practice up until the early 1990's. This application represents one of the most challenging environments since water depth, tidal movement, and wave activity from boats combine to present a number of factors with which the replanting method must contend. In the field project, which is described in detail in the following, the encased method demonstrated its capability in replanting along canal front bulkheading.

Figure 2. Planting along sea walls.

Seedlings were planted in a canal located in south Melbourne Beach where they were intentionally located in waters with significant tidal variances. The seasonal tidal depths at the planting site have varied from 30.5 centimeters (12 inches) during the summer months at low tide, to a maximum 132 centimeters (52 inches) at seasonal high tide. In addition, boat traffic on the canal is a daily occurrence creating periodic and substantial wave activity throughout the year. The mangrove seedlings planted at this location, using the encased method, have matured and shown significant growth since their planting. They have generated the prop roots which are a critical factor in long term survival and in the employment of successful encased replanting. One of the test plants has reached a stem height of 155 centimeters (61 inches) and prop roots now measure over 76 centimeters (30 inches) in length.

In contrast, seedlings planted using conventional methods and located in shallower water but in close proximity to the encased plants were unsuccessful in establishing themselves. All seedlings planted according to conventional methods were dislodged due to wave and/or tidal variances and were subsequently unsalvageable.

METHODS

The encased method is relatively simple and the materials required are inexpensive. The essential component is the encasement itself which effectively accomplishes the isolation and protection previously discussed. Standard PVC pipe was tested as one encasement alternative. PVC proved to be sufficiently rigid to perform the tasks of supporting the seedling through its development of aerial roots. Other materials could also be used but the semi-rigid, light weight characteristics of PVC made it an ideal candidate. Results of the study confirmed that PVC pipe offered suitable properties for the successful planting of mangrove seedlings.

Selecting a diameter that would hold sufficient soil in order to provide adequate nutrition for the seedling to root was a subjective judgement. Since no previous experience could be used for guidance two pipe sizes, 3.8 centimeters (1.5 inch) and 10 centimeters (4 inch) diameters, were tested. The vertical dimension or length of pipe was also somewhat of a subjective decision, therefore the major factor used in determining the lengths for experimentation was the depth of the water during the peak of the seasonal high tide. From evaluation of tidal variances at the site it was decided that two extremes in planting height be tested. The PVC was cut to length in order to accommodate the desired planting elevations. The pipe was then driven into the canal's bottom until it felt secure, approximately 30 centimeters (12 inches). The encasement was filled with soil from the site to 5 centimeters (2 inches) from the top of the pipe. By keeping the soil 5 centimeters below the rim it provided added protection in instances when water would rise to cover the top of the encasement. The seedlings were then planted vertically with the thicker end down and soil covering about 1/3 the seedling's total length (Figure 2).

One seedling was set at an elevation of 76 centimeters (30 inches) above the bottom. The average water depth at the site typically does not exceed 76 centimeters; however, water depth during seasonal high tide has been measured at 132 centimeters (52 inches). This setting ensured that the seedling would not be under water during normal seasonal conditions. A specific concern with this elevation was that during the summer when tide levels are seasonally low, the seedling would not receive adequate moisture for nourishment. However, the soil remained sufficiently moist for the seedling to develop without any unusual curtailment in growth. The seedling has developed normally and the stem has reached 152 centimeters (60 inches) in length and has spawned multiple prop roots.

Another seedling was set at an elevation placing it at 50 centimeters (20 inches) above the bottom. This setting ensured that the seedling would be under water periodically for varying periods of time throughout the year, although the frequency of being submerged would decrease during the summer and increase during the fall through the spring months.

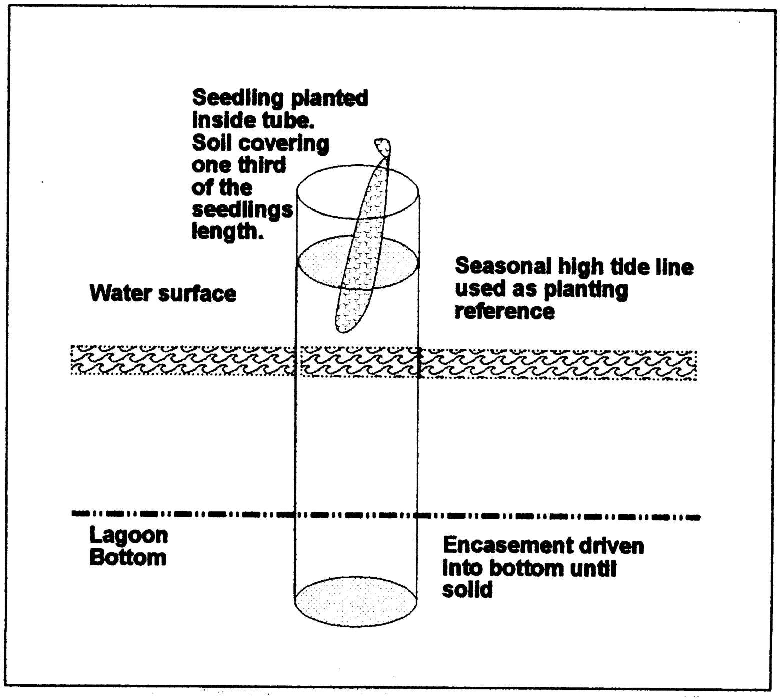

Figure 3. Diagram of encasement method.

The experimentation with extremes in elevation was subjective but was an effort to determine which extreme might reveal a deficiency in the encased method. Obviously the first extreme or highest planting elevation provided the greatest isolation and protection from environmental factors. The second extreme or lowest planting elevation subjected the seedling to the greatest influence of wave and tidal activity, and therefore a higher probability of physical damage or displacement.

For planting of the seedlings, PVC pipe was cut longitudinally for its entire length (Figure 4). This cut allowed for growth in the cross sectional area of the tree without restriction. As the plant's girth increased beyond the diameter of the pipe, the pipe opened and expanded, allowing the trunk and the root system to enlarge (Figure 4). An additional benefit of the longitudinal cut is that it facilitates the removal of the pipe. Once the tree matures sufficiently so that its foundation of prop roots are adequate to provide for the long-term nourishment and support of the tree independent of the encasement, the encasement then can be removed. The initial roots that were sprouted by the seedling followed the pipe, reaching downward toward the lagoon's bottom. The root bundles that developed inside the encasement eventually traveled the vertical length expanding the pipe as they grew. The continuing expansion of the longitudinal split increased to the point where the soil that was originally set inside for planting was being discharged, leaving only the root system. The mangrove was independent of the encasement in approximately 3 years (Figure 5).

Actual removal of the encasement will be heavily dependent upon the specific environmental conditions at the replanting site. Obviously the deeper the water, the stronger the wave activity, the more extreme the tidal variances, and the greater the effects of upland run-off, the longer the mangrove will need to rely on the encasement for support. Since the encasement is a passive form of support it can remain in place for as long as necessary, but typically its removal would take place between the second and fifth year of growth.

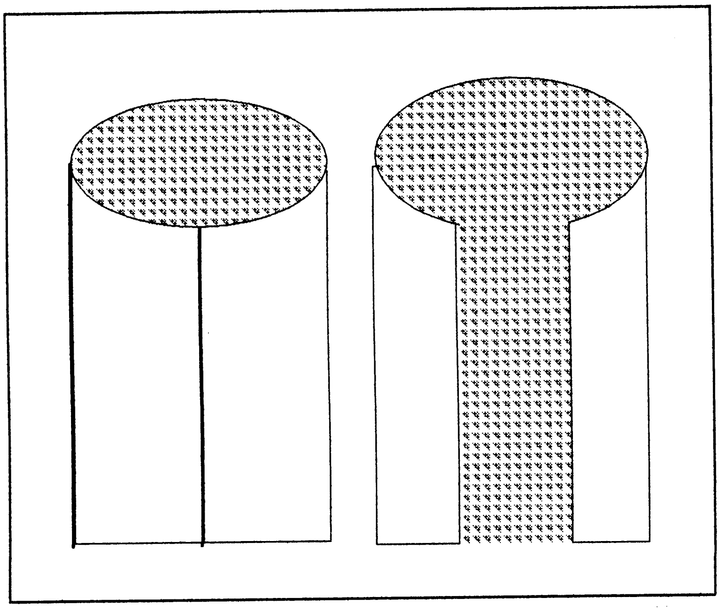

Figure 4. Expansion of trunk causing PVC pipe to open.

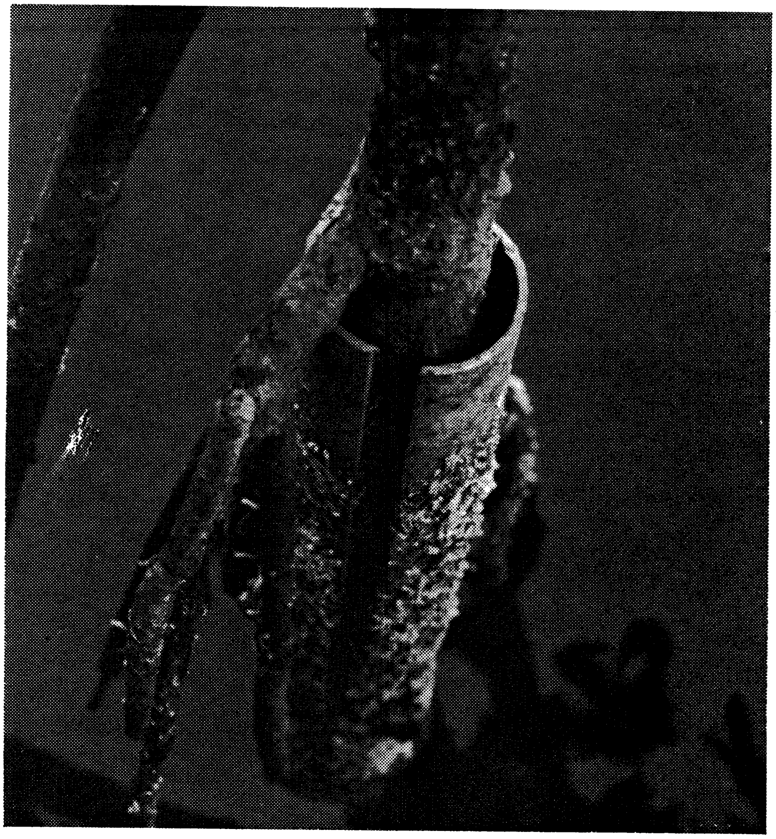

Figure 5. Mangrove becoming independent from encasement.

In the original project design, the intent was to remove all PVC as the tree matured; however, based on the most recent tests using a modified two-piece configuration the concept has changed such that the lower portion driven into the bottom will not be removed. Only the top portion would ultimately be removed after the first two of growing seasons and prior to the formation of aerial roots (Figure 6).

The new two-section system has several advantages over the original single tube design. First, the top section can be removed early in the plants development. This removal conceals the most visible portion of the encasement, thereby offering improved aesthetics over the single piece configuration. Second, it avoids any potential damage to the young tree that might occur with removal of the tube section containing the root system. Third, the foundation portion that anchors the seedling and prevents its dislocation remains in place. Finally, since the lower section will not be removed, holes can be located in the bottom segment to facilitate the migration of the roots into the surrounding soil. Based on experience with the single piece configuration, the lower portion of the encasement will become covered with barnacles and mosses blending with its surroundings.

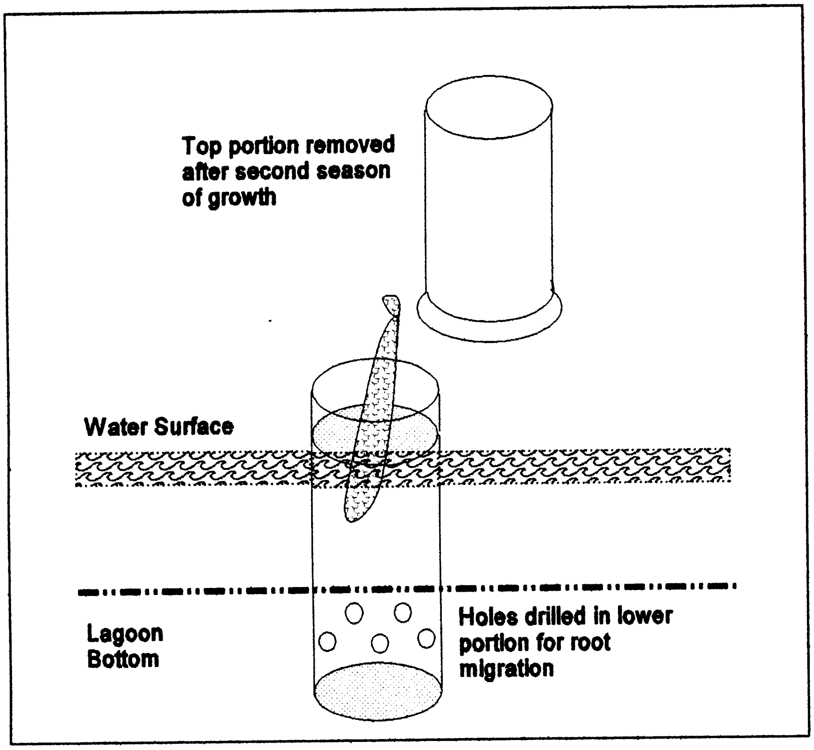

Figure 6. Modified Two-Piece Configuration

RESULTS

Results from experimentation indicate that the seedlings are not highly sensitive to the planting elevation relative to tidal activity. Even when the seedling initial growth has been physically damaged and it is partially submerged for a considerable period of time, as evidenced by the formation of barnacles on the seedling itself, the mangrove will sprout new growth and continue to develop (Figure 7). A characteristic of new growth following physical damage to an immature seedling is a two stem formation. Evidence supports the author's opinion that planting elevation is only critical at the point where the seedling becomes adequately protected from the harsh environmental factors that would result in dislodging the plant prior to it achieving self-reliance. Self reliance being when the tree can stand on its own, supported by its prop roots without assistance from the encasement.

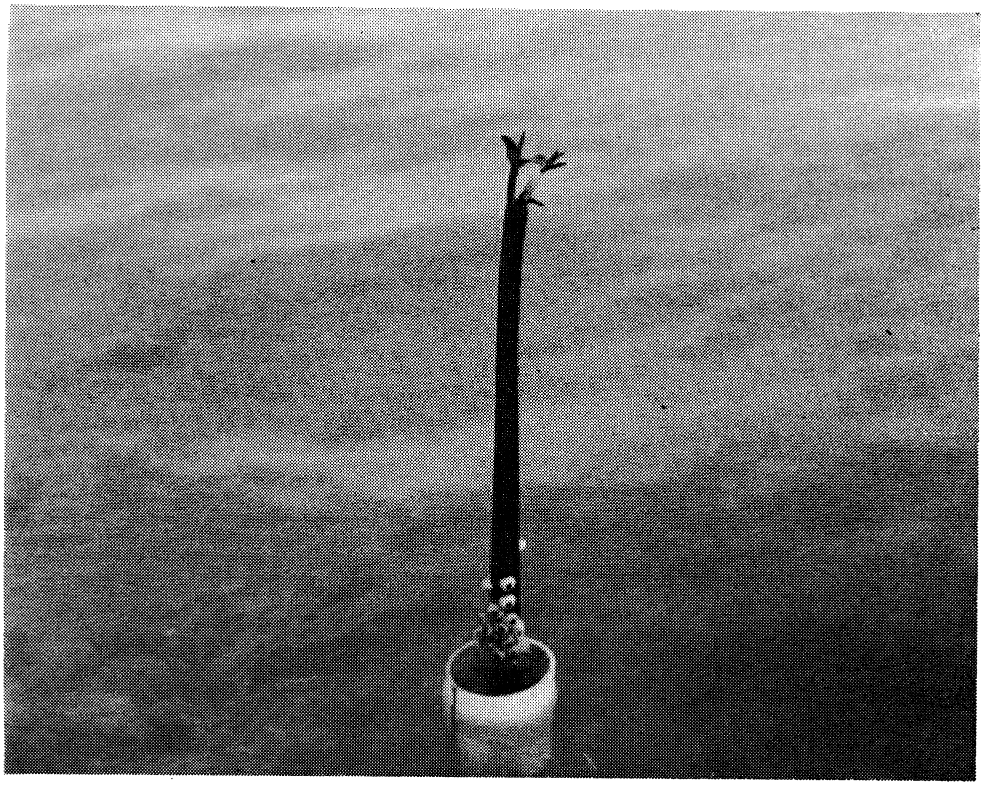

Figure 7. New growth after initial damage or partial submergence.

Differences in diameter of the encasement had no significant effect on the test plants, whether 3.8 centimeters (1.5 inch) or 10 centimeters (4 inch diameter) was used, the plants developed at about the same rate. The longitudinal cutting of the pipe minimized the effect of initial pipe diameter by allowing the plants to grow without restriction. Based on the study results, 3.8 centimeters (1.5 inch) diameter, 160 psi pipe can be effectively utilized for encasement without constraining normal growth.

Additional replenishment sites have been planted and are under evaluation. Each test site, like the Melbourne Beach bulkhead demonstration, has been chosen to analyze the methodology relative to the specific environmental factors that make conventional methods ineffective. A demonstration project which has been underway since fall of 1994, is located at Sebastian Inlet State Park. The park is located on the Florida east coast and is the northern boundary line for Indian River County and the southern boundary of Brevard County. Two previous attempts by St. Johns River Water Management District to replenish mangroves at this site have failed and hence the reason for our site selection. This locale is situated on the lagoon side of the inlet with a southern exposure that is regularly subjected to boat wakes and periodically subjected to extreme wave activity as winter storms pass through the area. Additionally this location is subjected to significant tidal variances (Figures 8 & 9). Both planting techniques, single and two piece configurations, are being tested at the Sebastian Inlet site.

Figure 8. Encased plantings at low tide.

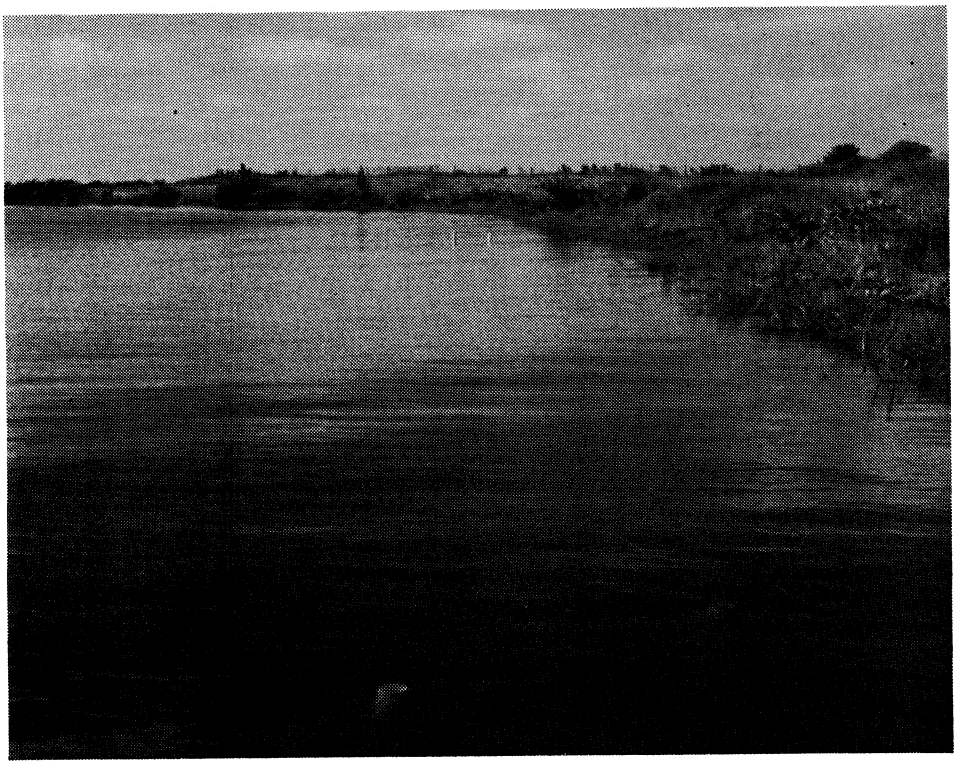

Figure 9. Encased plantings at high tide.

DISCUSSION

The necessity of implementing mangrove replenishment projects is supported by the documented reduction in mangroves throughout Florida's estuary systems. Over the past forty years the increases in population, water-front development, agriculture, boating and related activities have resulted in significant increases in the types and quantities of pollutants reaching intracoastal and coastal waters. With escalating pollution levels, the importance of mangroves to a healthy marine ecology dramatically increases. As natural members of estuary systems, mangroves mitigate the environmentally adverse and destructive effects of development and consequential pollution. Expansion of the mangrove dominion will help ensure the health of estuaries and coastal environments. Therefore, the author advocates mangrove replenishment on an extensive geographic basis as a logical contribution to coastal environmental protection and estuary restoration.

Evidence collected in field study demonstrated Encased Replanting to be a viable method for mangrove replenishment at sites with harsh environmental factors. The Encased method effectively enables the replenishment of mangroves where conventional methods can not succeed.

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RESTORATION OF HARD BOTTOM AND REEF HABITATS: AN ECOLOGICAL ENGINEERING APPROACH

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ABSTRACT

Coastal habitats and the biological communities dependent on them are frequently adversely affected as a result of human activities. Returning these habitats and communities to their predisturbance states is not always feasible when the habitat has been permanently altered or where persistent contamination remains on-site. Innovative ecological engineering methods, such as replacement or repair of injured natural reefs or hard bottom habitat with structures designed to replace critical habitat features, can help replace selected ecological functions and provide refuge for natural resources.

An ecological engineering approach was applied to design and adapt prefabricated reef modules for coastal habitat restoration applications. This approach involves applying small amounts of energy to build and deploy constructed reefs that provide topographic relief, structural complexity, and hard substrate. Natural physical and biological processes then drive recruitment and recolonization of the habitat by a variety of species and communities they support.

An initial mitigation application of this technology in Delaware Bay has proven successful in promoting the development of an abundant and diverse benthic encrusting community at the reef and has also served to enhance recreational fishing opportunities. A second application involved the development and implementation of plans to restore coral reefs in the Florida Keys. The ecological engineering approach, used to identify candidate restoration alternatives, and the multi attribute group decision analysis process, used to help select the preferred alternative, applied in both cases are described below.

INTRODUCTION

The degradation and loss of coastal habitats have been well documented and are likely to persist as population growth occurs along our coasts. We must continue to conserve and restore coastal habitats in a way that accommodates essential development without further degradation of these ecosystems and the natural resources they support.

A number of methods are currently used to compensate or offset coastal habitat losses or natural resource injuries. These include the restoration or creation of wetlands, mangrove areas, sea grass beds, kelp beds, etc. While these approaches are appropriate for in-kind replacement, they are not practical in some areas due to lack of space, poor water quality, or conflicts with other water-based activities. Where these methods are restricted or hard bottom or reef habitats are injured or lost, constructed reefs may provide temporary or permanent habitat replacement. Constructed reefs can function as temporary habitat/refuge until site conditions allow natural vegetation to be replaced or, where appropriate, as permanent replacement for lost or damaged hard bottom or reef structure. Although constructed reefs are frequently different in scale, compared with natural reefs, they function in much the same way as natural reefs: providing substrate, shelter, and altering current patterns etc. (Sheehy and Vik, 1992).

This report describes the use of ecological design and engineering approaches for developing prefabricated reefs for use in mitigation and restoration and briefly presents two different applications of this approach. The first case is a mitigation project in Delaware Bay where designed and prefabricated reefs were used as out-of-kind and off-site mitigation for land reclamation associated with port expansion. The second case is a restoration project for a section of coral reef injured due to the grounding of a large vessel. For the later case, on-site and in-kind structural restoration of the injured reef area was the objective. Both applications successfully incorporated a structured decision analysis process to aid managers in selecting appropriate designs based on multiple criteria.

APPROACHES TO RESTORATION PLANNING

Restoration planning is often a challenging task, characterized by the need to consider multiple factors, input from multiple stakeholders, and considerable uncertainty in performance and cost. Often the planning effort is subject to funding or schedule constraints which make the effort even more difficult.

Retrospective Review

Results of a retrospective review and analysis of a number of coastal mitigation and restoration or habitat enhancement projects (Aquabio, 1994), suggested that performance failures are frequently due to design flaws, siting errors, or inadequate cost analysis resulting in incomplete or poorly maintained projects. Although problems due to stakeholder conflicts and subsequent legal action were also noted, these problems generally restricted implementation rather than directly affecting performance. Our analysis suggested that technical problems noted with restoration projects are often associated with planning perspectives that are:

- Focused on available alternatives rather than clear restoration values
- Based only on a limited subset of the factors that influence final performance
- Based on incomplete or optimistic estimates of total project costs

Alternative-focused approaches are based on existing and readily available methods or technologies as opposed to value-focused approaches in which the designs and/or sites selected are based on a careful analysis of objectives and selection criteria derived from functional analysis of project requirements. An alternative focus constrains the range of options considered to those commonly applied in the past to "similar" habitats. It often prematurely eliminates consideration of a broader set of options or innovative approaches without a full consideration of their expected benefits. The value-or attribute-focused approach begins with a close examination of requirements, based on impacts or injuries, and then considers all relevant factors in the design/development process or in the selection of preferred alternatives.

Projects are commonly selected based on a single objective and/or narrow set of criteria without full consideration of all the factors that influence the final project outcome. Nontechnical issues, such as public opinions, aesthetics, or legal/policy consistency, etc., are often ignored or only considered after an "ideal" technical solution is selected. In other cases, significant technical issues, such as the potential for collateral damage, are ignored. The issues or factors not considered during original deliberations often come to the surface later in project development and may delay or preclude the implementation of the proposed solution.

Restoration project selections are often based solely on initial acquisition or construction costs without consideration of the life-cycle costs of alternative projects. This results in project cost overruns or, when funds are limited, incomplete projects. In addition, even when life-cycle costs are considered, cost uncertainties are often ignored and the most optimistic estimates are used for cost calculation. In either case, project performance is frequently degraded by a reduction in project scale or failure to do required maintenance or adaptive management that reduces long term project performance.

With the wisdom of 20-20 hindsight on our own projects and the opportunity to observe and review a wide range of past efforts by others, we have developed an "Integrated Restoration Planning" (IRP) approach aimed at addressing some of the common problems noted above. This approach is the application of a set of systems and cost analysis methods to the restoration planning process. We have found it useful for our own work, especially, when new technology is applied or where there are multiple objectives and decision makers. The IRP process helps focus the design/development effort on clear functional objectives, structure the problem for group participation, allows an explicit consideration of uncertainty, and documents how the preferred alternative was selected.

Integrated Restoration Planning

The IRP approach evolved over time based on "lessons learned" and is adapted to the specific needs of a particular project. We have found that it helps ensure that all the relevant factors and uncertain conditions that influence the final restoration plan are considered. The IRP process is a set of linked analytic methods centered around a multi attribute decision analysis framework that provides the underlying structure for group restoration planning. Key elements include:

- Ecological engineering/design to provide a basis for value- or attribute-focused design or alternative identification and benefit assessment

- Failure mode, effects, and criticality analysis to assess performance risk

- Probabilistic and life-cycle cost analyses to assess cost and cost risk

- Multi attribute decision analysis methods to help decision makers in integrating information for selecting mitigation or restoration alternatives

For the restoration of hard bottom or reef habitats, an ecological engineering approach is applied to decide how to alter substrate characteristics, structural complexity, and circulation patterns to create conditions suitable for recruitment and colonization by target species or communities. This phase begins with an ecological design process, which is a search for agreement between either past features and functions of the impacted habitat or injured communities (baseline) and a set of functional requirements established by restoration planners and decision makers. It involves translating general restoration goals into a set of clear objectives and specific criteria for developing or identifying mitigation or restoration alternatives. Since more than one functional objective (e.g., the reef must be stable and ecologically effective) is required, design tradeoffs are often necessary.

The ecological design/engineering phase of the process is applied to ensure that clear and comprehensive restoration objectives are articulated early in the planning process and that these are translated into measurable criteria that will focus the design/selection effort and help provide a framework for compliance and performance monitoring.

Failure mode analysis is an analytic approach used to evaluate restoration performance risk and is based on a structured examination of possible modes by which the project might fail, the probability associated with these failure modes, the direct effect of these failures, and the short and long term consequences of these effects on the project performance. It is used to provide a realistic projection of expected restoration benefits. For example, a reef designed to provide a given amount of suitable surface area for epifaunal colonization, may eventually fail due to subsidence, shortening the length of time that it provides support for target species or communities. On the other hand, a reef made of low density material might fail by being transported off-site under severe oceanographic conditions. In the latter case the habitat would be also lost, but there might be substantial collateral habitat damage due to the movement of materials over live bottom. In both cases the reef failed, however, the ultimate consequences (simple loss versus collateral damage) are quite different and must be factored into an evaluation of alternatives.

Cost analysis, as opposed to simple estimating, is conducted to ensure that a complete and realistic project cost is considered when comparing alternatives and that cost risk is fully evaluated during the planning process. One common problem noted in our in review of restoration projects was that all aspects of the project were not considered during planning. For example, maintenance of some sort (such as replanting or restocking based on monitoring) is frequently required, but not included in the project budget. This biases cost-benefit analyses in favor of projects with low initial acquisition or development costs. Life-cycle cost analyses, which include all planning, monitoring, and maintenance costs, are prepared to provide a complete picture of project costs. Probabilistic uncertainty analysis, accomplished using simulation modeling, is used to consider the potential distribution, rather than an optimistic point estimate, of a project's cost. For example, planners considering offshore reef construction that is scheduled to overlap with the hurricane season need to consider dislocation costs due to storms as well as normal construction problems associated with material delays, equipment breakdowns, etc.

Multi attribute decision analysis is a structured framework, usually presented as a matrix of options and selection criteria, used to help evaluate complex problems involving multiple criteria, multiple decision makers or stakeholders, and uncertainty (Sheehy et. al. 1995). It provides a rational and explicit process for identifying and rating the suitable restoration alternatives based on all the relevant factors and considering the decision makers values. In this phase, the results from the cost and benefit analyses are combined with decision makers' values to enable planners to comparatively consider the merits of alternatives.

The overall IRP approach is designed to keep a clear and consistent set of objectives in place and to provide a realistic view of the project early in the planning process where is easier and less expensive to adaptively modify the project scope, schedule, or budget. Ecological design/engineering approach focuses the design or alternative identification process on clear objectives and constraints. The failure mode and cost analyses help provide a realistic technical assessment of benefits and costs, respectively, while multi attribute decision analysis provides an overall framework for the evaluation of alternatives.

The following two case studies provided an opportunity to apply aspects of the IRP approach to actual restoration planning projects. These case studies directly involved decision analysis as a framework the selection of proposed technology.

Brown's Shoal Constructed Reef, Delaware Bay

We applied an ecological engineering approach to plan aspects of a mitigation project proposed for the loss of a shallow water habitat in Delaware Bay. The habitat loss was due to the expansion of the Port of Wilmington, which required a 142 acre land fill. Available sites with suitable conditions for on-site or in-kind restoration were severely limited and, based on recommendations made by Aquabio, resource agencies agreed to develop constructed reefs as off-site, out-of-kind habitat mitigation.

The objectives of this part of the mitigation planning effort were to enhance the benthic forage base and improve fishing opportunities within Delaware Bay. Due to conditions in Delaware Bay and long term performance requirements, we focused the mitigation plans on large scale prefabricated reef technology. Designed and prefabricated units afford considerable design and siting flexibility and thus provided more surface area/ per unit footprint than more traditional constructed reefs made from rock or scrap material. Large scale prefabricated reefs were originally developed for commercial fishing enhancement in East Asia (Sheehy et al, 1983). This technology was introduced to the United States (Sheehy et al, 1983, Aquabio

1982) and demonstrated at several sites off the coasts of Florida (Sheehy, Vik and Mathews, 1983). Extensive performance evaluations conducted in Japan and Taiwan by the senior author and the results of studies in Florida suggested that these types of units were particularly appropriate where suitable sites were limited, site conditions were restrictive, and long term performance was an objective.

A multi attribute decision analysis approach was used to help select both the reef site and conceptual designs for the reef unit. A team of ecologists, oceanographers, engineers, and fishery managers evaluated and ranked 34 designs and 22 potential sites using multi attribute decision analysis methods. Sites were evaluated using multiple factors, some of which where computed from data layers in a simple Geographic Information System (GIS). These factors included a range of physical, biological, and practical factors (such as proximity to shipping channel and boat ramps, etc.). We identified three potential sites that met all critical site attribute levels (Sheehy and Vik, 1985). The final site, sheltered by Brown's Shoal, was selected by resource and environmental agency technical representatives.

Reef unit designs were evaluated based on 11 primary factors (not all of equal importance) related to expected stability, biological performance, and cost. Conditions at the three candidate sites were similar and used as the basis of reef unit selection and/or configuration (many prefabricated reef units are modular and can be built in several configurations permitting a "design to site" approach). Factors considered in the evaluation of alternative designs include those related to unit stability (drag and lift forces under extreme current and wave conditions, etc.), and biological effectiveness (surface area available for colonization, crevice and shelter volume, and current modification, etc.), and life-cycle cost. The three highest ranking designs were selected as the conceptual designs from the remaining alternatives (Sheehy & Vik, 1984). The final design selected for application was a modified version of one conceptual design. It provided a large stable base, a considerable amount of surface area, and, because of significant value engineering in the final design by Van Doren Industries, was more cost-effective than the original design. The selected unit is shown in Figure 1.

The results of the first two years monitoring at the Brown's Shoal Reef Site confirm that the reefs have met or exceeded designers' expectations both in terms of developing benthic biomass and improving recreational fishing opportunities at the site. With minor exceptions, the reef units are stable at the site and suffered only minor subsidence and some minor damage that occurred during placement.

Benthic biomass at the reef site increased significantly (147-895 fold) compared with infauna resident in an area equivalent to the reef footprint (Steimle 1993). This was due, primarily, to the increase in surface area available for the development of encrusting communities. Both juvenile and adult fish densities observed were higher on the reef than on nearby level bottom reference sites and observations (Aquabio unpublished data) suggest that some species are feeding directly on the encrusting community or forage attracted by the micro habitat, or on planktonic food sources concentrated by the water turbulence created by the reef.

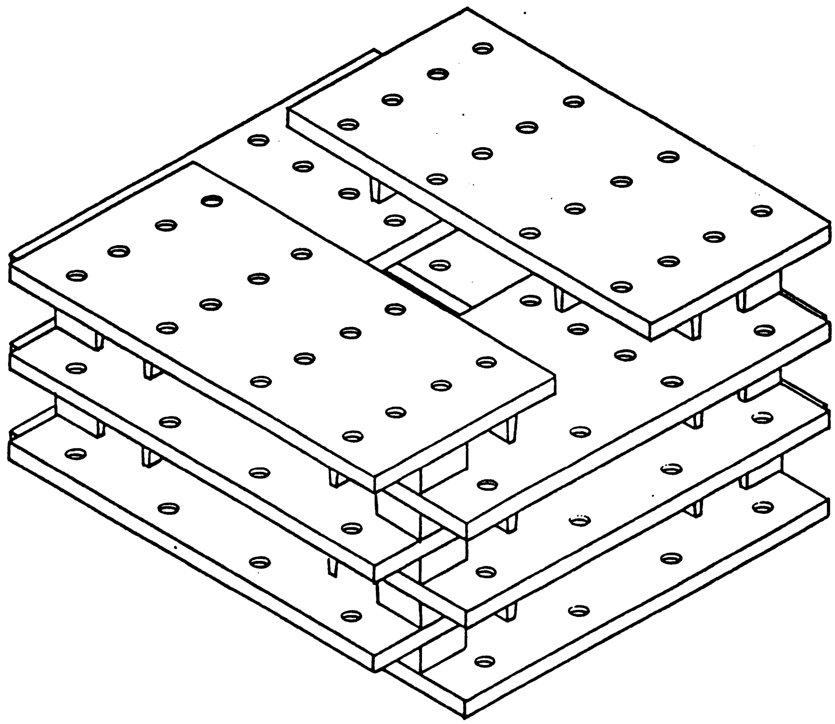

Figure 1. Prefabricated reef module used to construct the Brown's Shoal Artificial Reef in Delaware Bay.
(Dimensions 6.1 x 6.1 x 2.7 meters LxWxH)

Maitland Grounding Site, Florida Keys

Recently, this ecological design/engineering approach was applied to plan restoration for an area of coral reef in the Florida Keys National Marine Sanctuary injured from a vessel grounding (M/V Alex Owen Maitland). National Oceanic and Atmospheric Agency (NOAA), the natural resource trustee, recovered damages from the responsible party as part of a settlement of a Natural Resource Damage claim. Some of these funds were applied to restore the large crater (525 square meters) created at the site as the crew of the vessel attempted to pull it off the reef (Continental Shelf Associates, 1993).

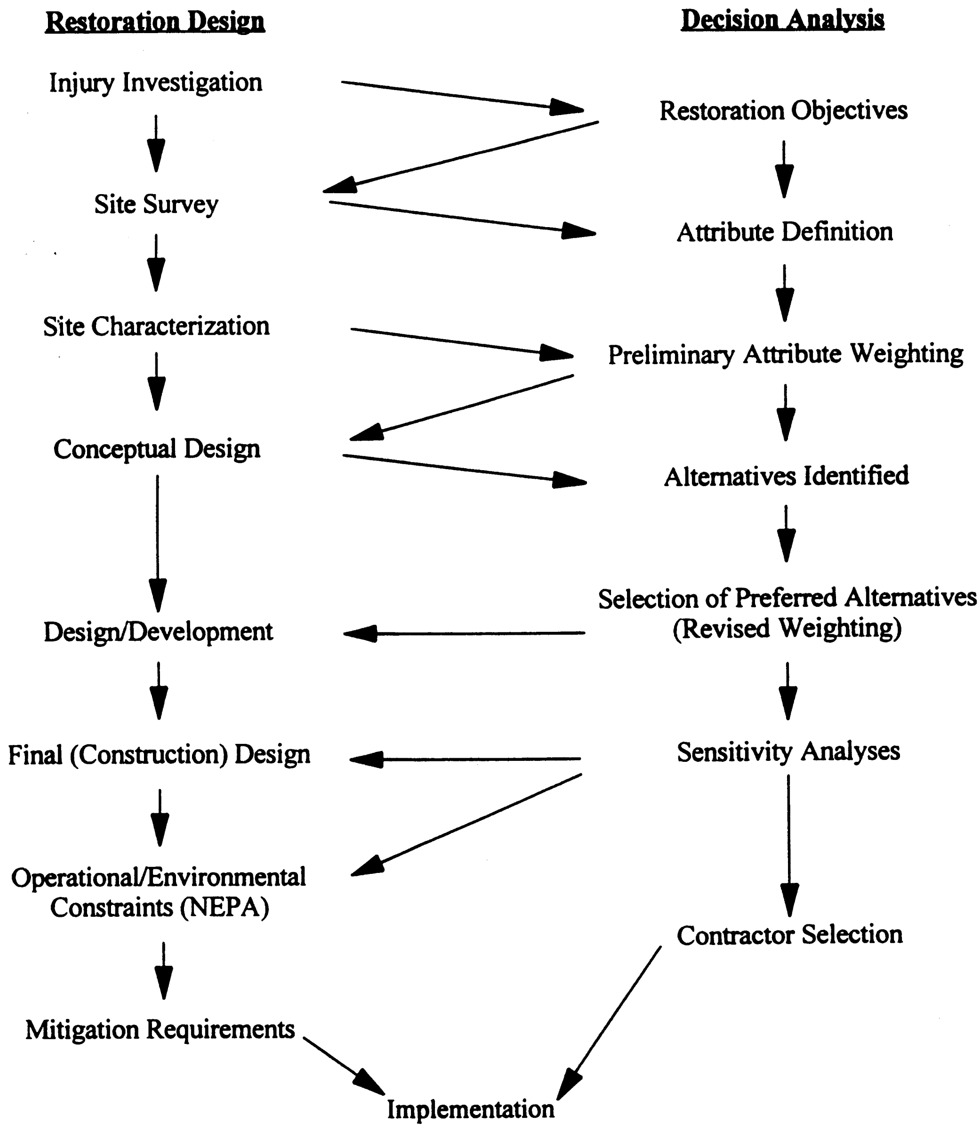

Figure 2. Coral reef restoration planning process illustrating linkage between the design and the management decision processes.

A team of ecologists, coastal engineers, NOAA sanctuary managers and attorneys was established to help plan the structural restoration at the site, prepare necessary permits and environmental documentation, and help NOAA in selecting and monitoring the construction contractor. This was a difficult engineering challenge due to the severe oceanographic conditions at the site during storms, the requirement to meet a complex set of trustee objectives, and the inherent uncertainty in developing new restoration technology. To guide this planning effort and ensure that trustee requirements were integrated into the engineering design process, a group decision analysis framework was developed and applied. The interaction between the trustee decision making and engineering design components of the restoration planning process is illustrated in Figure 2. This design/ development approach allowed us to keep trustee objectives in focus throughout the planning process. A multi attribute decision analysis approach was again applied to help trustees select the preferred conceptual alternative, establish a framework for the comparative analysis required for National Environmental Policy Act documentation, and select construction contractors most capable of completing this task.

An innovative approach, using prefabricated reef replicating armor units designed by Olsen Associates, Incorporated (Olsen Associates, 1995), was selected for the structural restoration at the Maitland site. These armor units were specially designed concrete components fabricated to meet site specific requirements for stability, crater dimensions, and substrate aesthetics. This design provided a stable, biologically effective, and aesthetically acceptable patch for the blowhole area and employed a construction method that reduced the potential for collateral damage.

Unit components were fabricated on shore, transported to the site by barge, and placed using a barge-mounted drag line. Units were unique and incorporated lime rock on the surface to simulate a natural rugosity and replicate natural features at the site. Individual units were placed according to the site plan (Figure 3) and underwater tremie concrete, with inset natural rock, was used to fill gaps provide a seamless appearance for the finished product. The restoration was completed as planned with only minor delays due to greater than expected periods of adverse weather. Figure 4 shows several of the units that made up the armored cap.

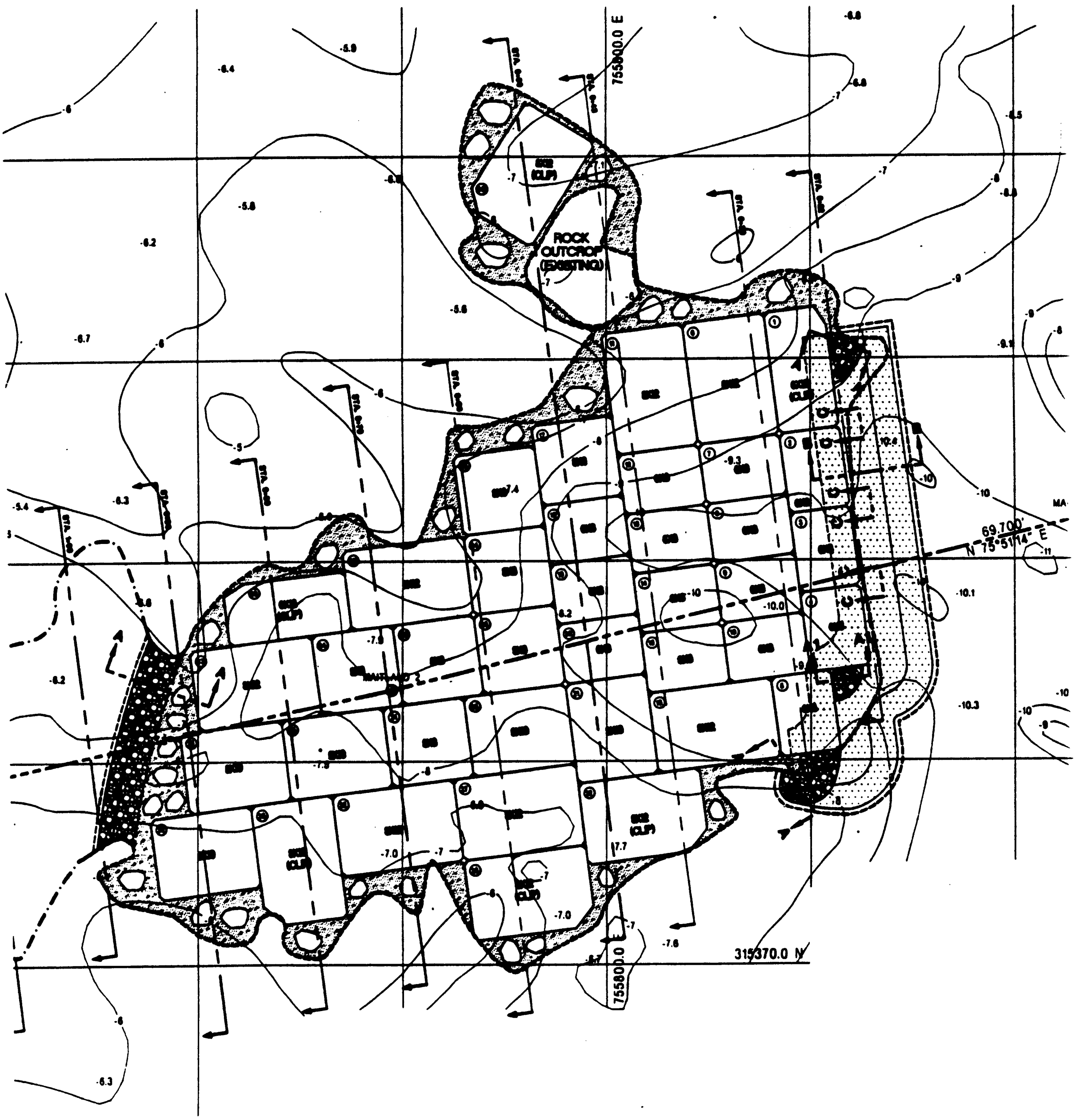

Figure 3. Site plan illustrating placement of designed reef replicating units at the grounding site of the M/V Alex Owen Maitland.

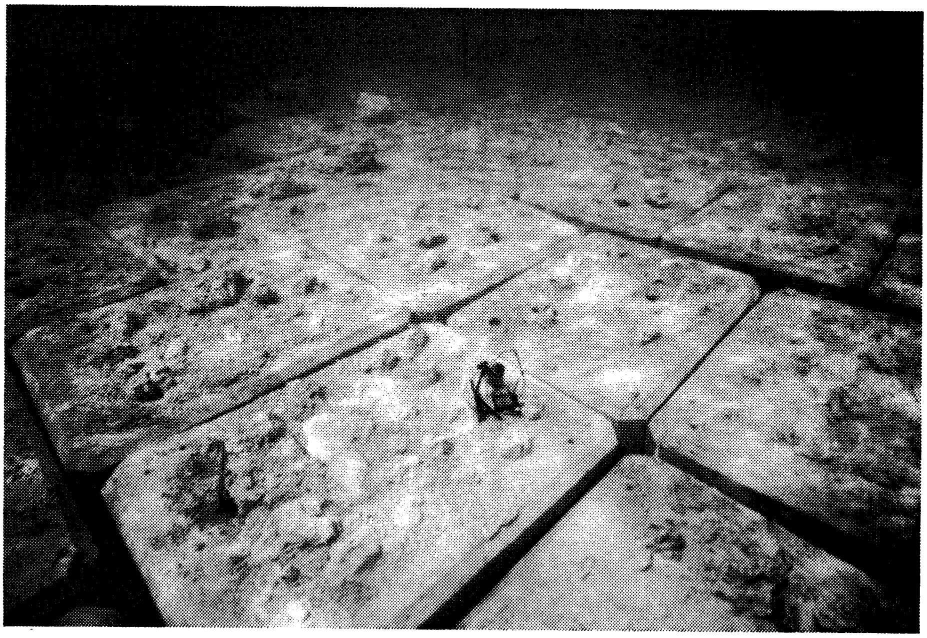

Figure 4. View of a section of the reef repair work prior to application of tremie concrete.

DISCUSSION AND CONCLUSIONS

Compared with a number of earlier mitigation or restoration planning projects, including some of our own, the combination of the ecological engineering and decision analytic approaches applied in these two case studies improved the planning process by:

- Providing clear initial planning guidance to those identifying or designing the technology used for restoration

- Supporting efficient interdisciplinary team deliberations leading to the group selection of a preferred alternative technology or design

- Establishing a consistent framework that described and documented the decision-making process for environmental compliance or external review

In both cases, the clear definition of functional requirements and objectives focused the efforts of technical specialists on appropriate design and selection criteria. This consistent perspective extended from the evaluation of conceptual designs through determining how compliance monitoring would be conducted. It also helped determine the value of various types of data or information needed in the restoration planning process. This allowed resource managers to focus their resources on those items really important to the design or selection process.

The design selection approach involved the development and submittal of evaluation questionnaires, interim ratings, and group meetings to discuss and adjust group rankings. This approach enabled a group of experts distributed across disciplines and geographic areas, to contribute their expertise effectively and efficiently. It focused limited meeting discussions on those areas subject to important differences of opinion. The decision analysts prepared rating and weighting surveys, compiled individual and group evaluations, and performed sensitivity analyses to show what factors were driving the ranking and to highlight conflicting opinions for discussion. Meetings were held to clarify and resolve differences and to reach a recommendation acceptable to all participants or identify critical data gaps.

The systematic design/decision support approaches applied in these two cases helped maintain a consistent theme throughout the planning process and provided a clear record showing why and how the decisions were made. This is particularly important in restoration situations where the recommended technologies are new or controversial, need to be documented for external review, or subject to considerable uncertainty in either performance or cost.

ACKNOWLEDGEMENT

The development of the Delaware Bay artificial reef was supported by the Delaware Department of Natural Resources and Environmental Control and the Philadelphia District of the U.S. Army Corps of Engineers. Support for the coral Reef restoration project was provided by the National Oceanic and Atmospheric Administration under Contract Number. In addition, a number of firms were significantly involved in aspects of this work and their contributions are gratefully acknowledged. These include, Van Doren Industries, Inc. (Delaware Bay reef final design), Olsen Associates, Inc. and Continental Shelf Associates, Inc. (coral reef restoration).

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THE UPPER ST. JOHNS RIVER BASIN PROJECT
THE ENVIRONMENTAL TRANSFORMATION OF A PUBLIC FLOOD CONTROL PROJECT

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ABSTRACT

During the 1980's, the Upper St. Johns River Basin Project was transformed into a national model of modern floodplain management. The project is jointly sponsored by the St. Johns River Water Management District and the U.S. Army Corps of Engineers. With construction nearly completed, the project now represents one of the largest and most ambitious wetland restoration projects in the world.

Since the mid-1950's, the vast Central & Southern Florida Flood Control Project included provision for major flood control works in the marshes of the upper St. Johns River basin. In the late-1960's, changes in the federal environmental arena, coupled with Florida's landmark water management and environmental legislation of the early-1970's, permanently altered the nature of what was once a conventional and highly structural flood control project.

While maintaining its primary flood control mission, today's Upper St. Johns River Basin Project is a multipurpose project designed to balance the multiple uses of the river and to provide for major environmental habitat restoration and water quality benefits. The project continues to evolve to meet the ever-changing basin conditions and public demands. Because the current project relies more on a "semi-structural" approach to manage the river as a natural ecosystem, the Upper St. Johns River Basin Project has been hailed as an ecological model of sustainable water resources development and may well influence the course of future public water projects nationwide.

This paper tracks the dynamic transformation of this traditional federal flood control project to its current design and highlights how, like in the natural world, the project has changed in response to the need for successful adaptation and evolution.

INTRODUCTION

This paper describes the transition of events charted by the SJRWMD (St. Johns River Water Management District) that led to the development of the current GDM (General Design Memorandum) and the several DDM's (Detail Design Memorandums) prepared by the USACE (U.S. Army Corps of Engineers) for the Upper St. Johns River Basin Project. The current project, under construction since 1988, was redesigned in the early-1980's to address several water management problems and environmental concerns. Today's "Upper Basin Project" reflects major design changes since the original project design was completed in 1957. This paper discusses detailed project information, current environmental factors, and traces the evolution of change reflected in the current project design that transformed the project into a model of environmentally sustainable floodplain management.

STUDY SITE

The study site is located in east central Florida, situated southwest and inland from Florida's Space Coast. The project area extends about 121 kilometers (75 miles) from the Florida Turnpike in southern Indian River County northward to Lake Washington in central Brevard County (Figure 1). The upper St. Johns River basin is a watershed 5,180 sq km (2,000 sq mi) in size and drains an area about the size of the state of Delaware.

PROJECT HISTORY

In the late-1800's, pioneers waded deep into Florida swamps with survey rods and steam shovels, opening vast areas of the state's watery interior. Grand "reclamation" schemes of the day, however, were not limited to draining Florida's Everglades. Early plans also included water management "improvements" to control floods and drain extensive areas of the upper St. Johns River marshlands for agricultural production and private development.

The construction of a large drainage system in northwestern Indian River County was one of the first significant water management works constructed in the upper St. Johns River basin. A road grade and drainage canal -- the Fellsmere Grade and Fellsmere Main Canal -- were constructed across the marsh to connect the hamlet of Fellsmere with the small outpost of Kenansville. Other canals followed, cutting through a low coastal land ridge that separated waters in the upper St. Johns River basin from the Indian River Lagoon -- one of the most biologically diverse estuaries in North America. Through these canals, large amounts of fresh water were diverted from the St. Johns River basin to the Indian River Lagoon and the Atlantic Ocean. As more dikes were constructed and large pumping stations installed to meet private flood protection needs, thousands of acres of nutrient-rich floodplains were opened for agricultural production (Figure 2).

Over the past several decades, a significant loss of historical floodplain marsh in the upper St. Johns River basin resulted in major flooding and water quality problems. At the turn of the century, the 164,000 ha (405,000 ac) floodplain of the St. Johns River was a broad shallow marsh. Within seven decades, however, about 70 percent of these fertile wetlands were converted into agricultural fields to support the production of citrus, row crops and beef cattle. (Figure 3). Loss of wetland habitat due to floodplain encroachment practices greatly reduced floodplain storage and conveyance capacity in the river, and severely altered the natural hydrologic and ecological regime of the marsh ecosystem. The impacts of lost floodplain storage and conveyance capacity were especially acute after major storms in the 1920's and 1940's resulted in devastating floods in the central and southern parts of Florida. Thus, the need for a massive flood control project was recognized early in this century.

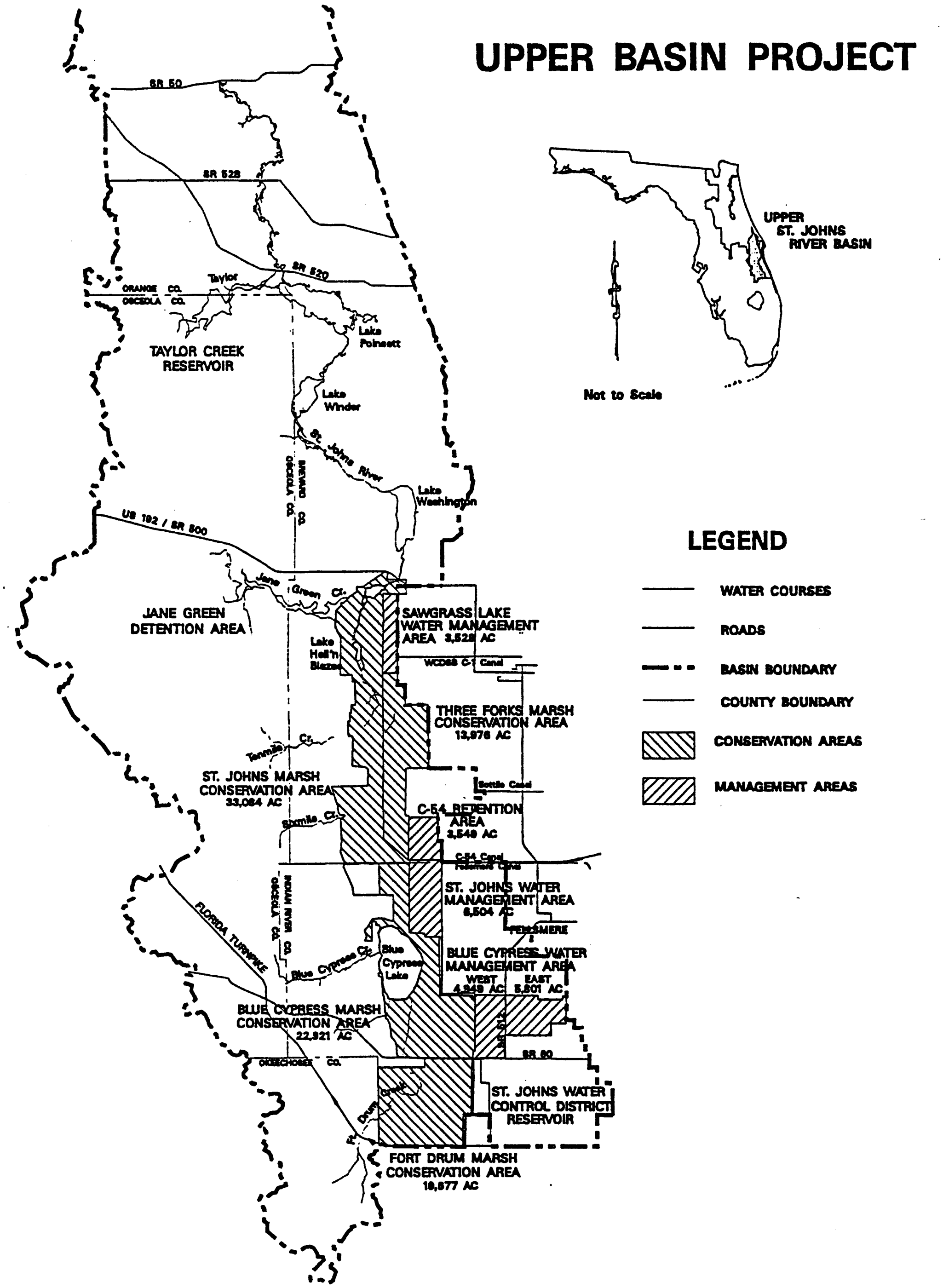

Figure 1. Location Map. *The Upper St. Johns River Basin, located in east-central Florida, drains approximately 2,000 sq. mi.--a watershed the size of the state of Delaware.*

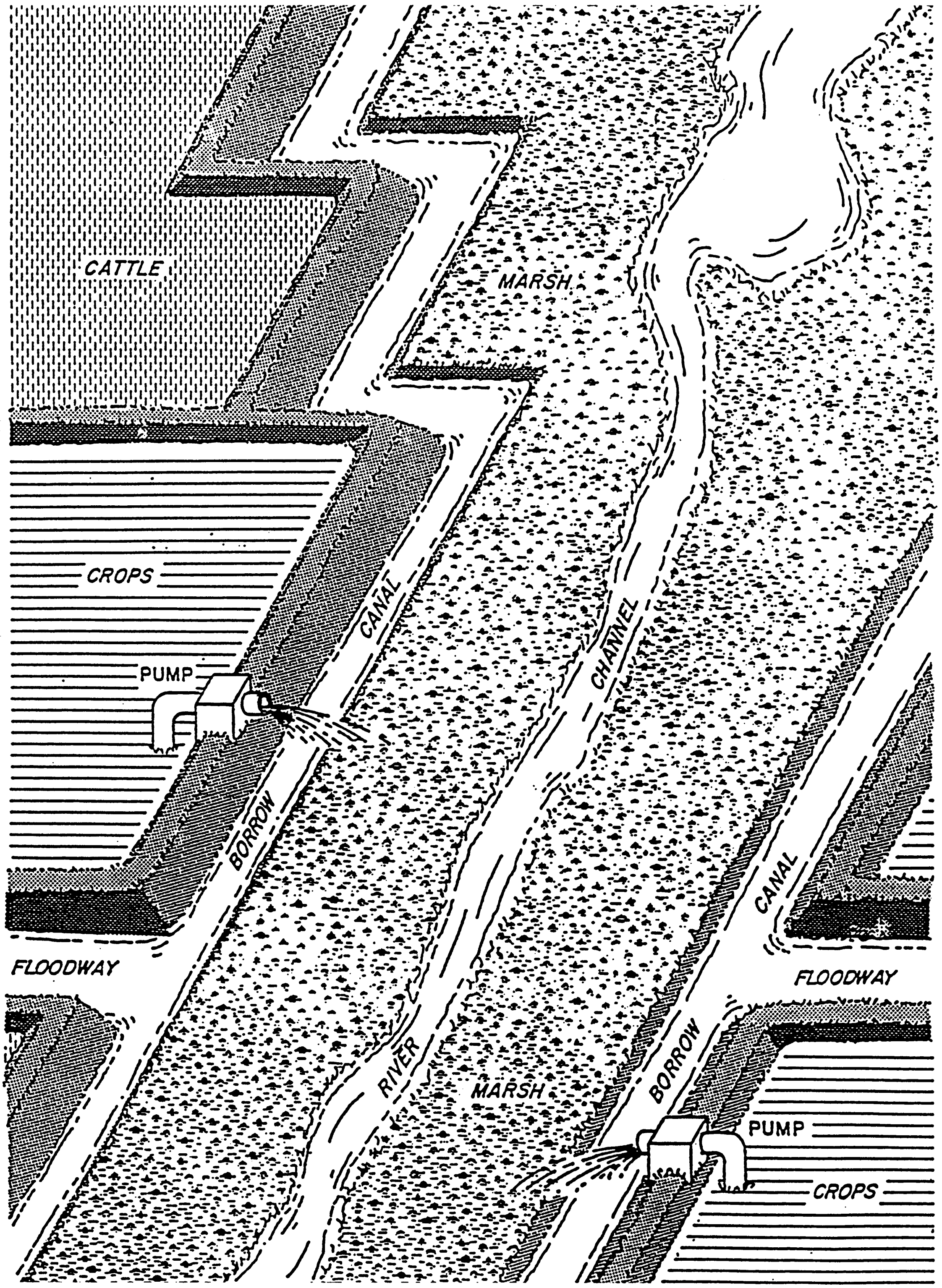

Figure 2. Typical floodplain levee system. *From the 1900s to the early-1970s, over 40 private levee systems were constructed throughout the upper St. Johns River basin. The former wetlands encircled by the levees were pumped dry to grow citrus, row crops and beefsteak.*

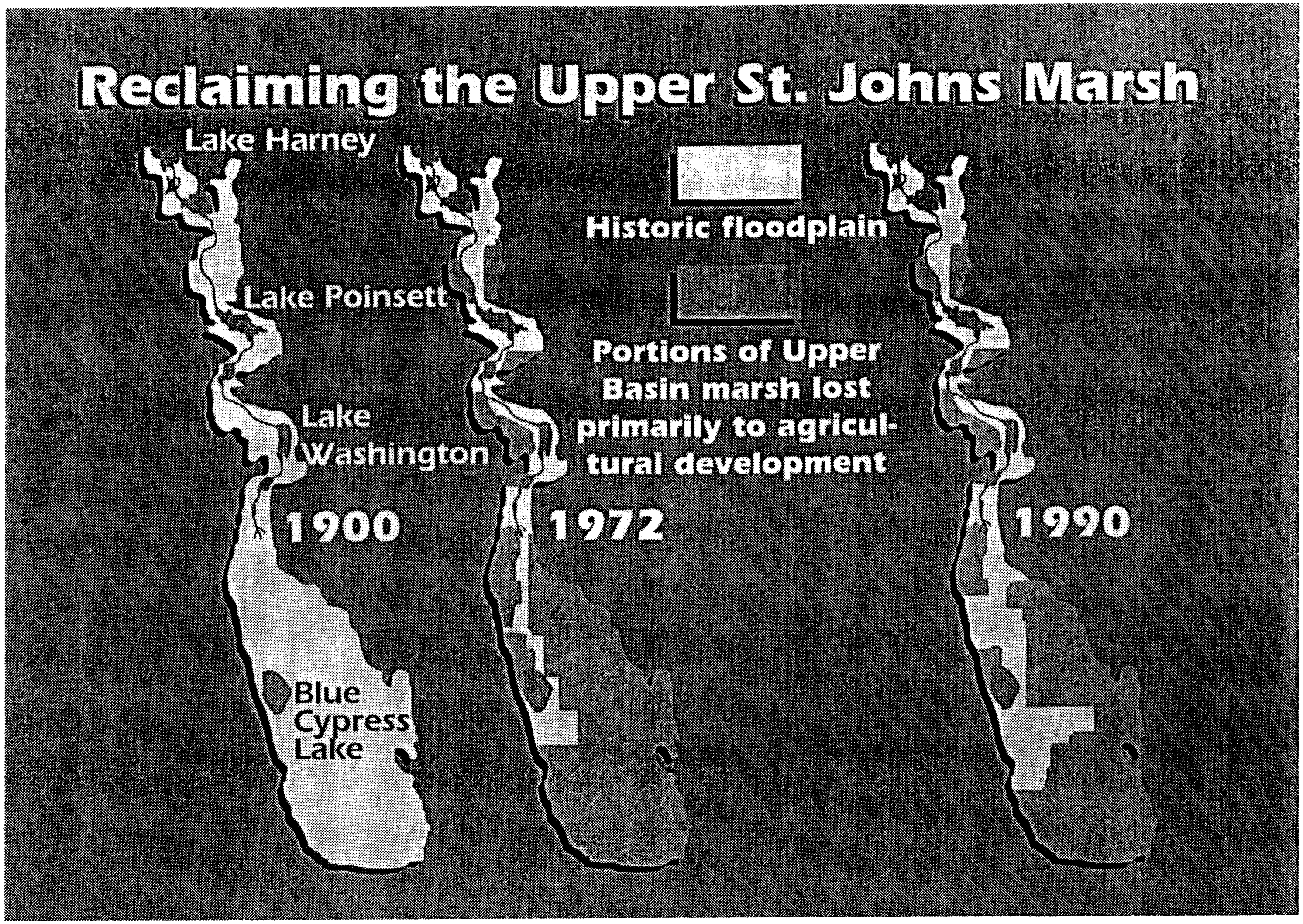

Figure 3. Extent and chronology of floodplain encroachment. *By 1972, agricultural development within the upper St. Johns River floodplain accounted for a loss of 62 percent of the river's headwater marshes. Since the mid-1980s, thousands of acres of floodplain within the historic river valley have been acquired and re-flooded as part of the current Upper St. Johns River Basin Project.*

The history of modern public flood control projects in Florida formally began in 1948 when the U.S. Congress authorized the C&SFFCP (Central & Southern Florida Flood Control Project) and the Florida Legislature created the FCD (Central & Southern Florida Flood Control District) to act as the local sponsor for the C&SFFCP. The original congressional act, which did not include areas within the upper St. Johns River basin, was amended in 1954 to include project works within the St. Johns portion of the larger C&SFFCP. In coordination with the FCD, the Jacksonville District, USACE prepared a project GDM which was completed in 1957. A modified plan was adopted in 1962, and construction of the project began in 1966.

Under the 1962 plan, flood stages would be reduced in the upper reaches of the basin by diverting large amounts of water from the St. Johns River to the Indian River Lagoon during major storm events via the C-54 (Sebastian) canal. Downstream of C-54, flood stages would be attenuated by the detention and storage of surface water runoff in large upland reservoirs located west of the river valley. By 1969, the C-54 canal system was fully operational and a major upland levee and reservoir system (L-73 and associated structures) was near completion. A map depicting the 1962 plan is provided (Figure 4).

Passage of the federal National Environmental Policy Act of 1969 required that an EIS (Environmental Impact Statement) be prepared for federally funded water projects. In 1970, the USACE began preparation of the required EIS. Early findings indicated potential for serious adverse environmental impacts, and in 1972, construction within the upper St. Johns basin was halted pending completion of a more comprehensive EIS. The state of Florida determined that the original project design was unacceptable because of the potential for significant environmental degradation to the upper St. Johns River ecosystem, and in 1974, the state withdrew its formal sponsorship of the project.

Environmental concerns included the potential adverse impacts of large fresh water discharges to the Indian River Lagoon and the potential for severe water quality and habitat degradation to both the natural riverine marshes and to several large upland creeks and hardwood swamp habitats. As a result, project construction was indefinitely suspended, but not before the project's primary flood control component-- the 6,000 cfs capacity C-54 canal -- was fully operational and major flood control benefits to the basin's agricultural interests realized. The interim operational scheme for C-54 called for major fresh water releases to the Sebastian River when water levels in the Blue Cypress Lake basin reached flood stage. Impacts of large stormwater discharges to the lagoon, while not fully documented in earlier years, are now understood to be detrimental to the lagoon s fragile ecosystem which now supports a thriving commercial shellfish industry.

In 1977, local sponsorship for the project transferred from the FCD to the SJRWMD (St. Johns River Water Management District) which was established by Chapter 373, <u>Florida Statues</u>, as adopted in 1972. The SJRWMD completed an extensive reconnaissance report in 1979 describing the then existing basin conditions. A citizens' advisory committee, whose membership was representative of the various interests within the basin, worked with SJRWMD staff to develop a BDC (Basic Design Concept) which the SJRWMD Governing Board adopted in November, 1980.

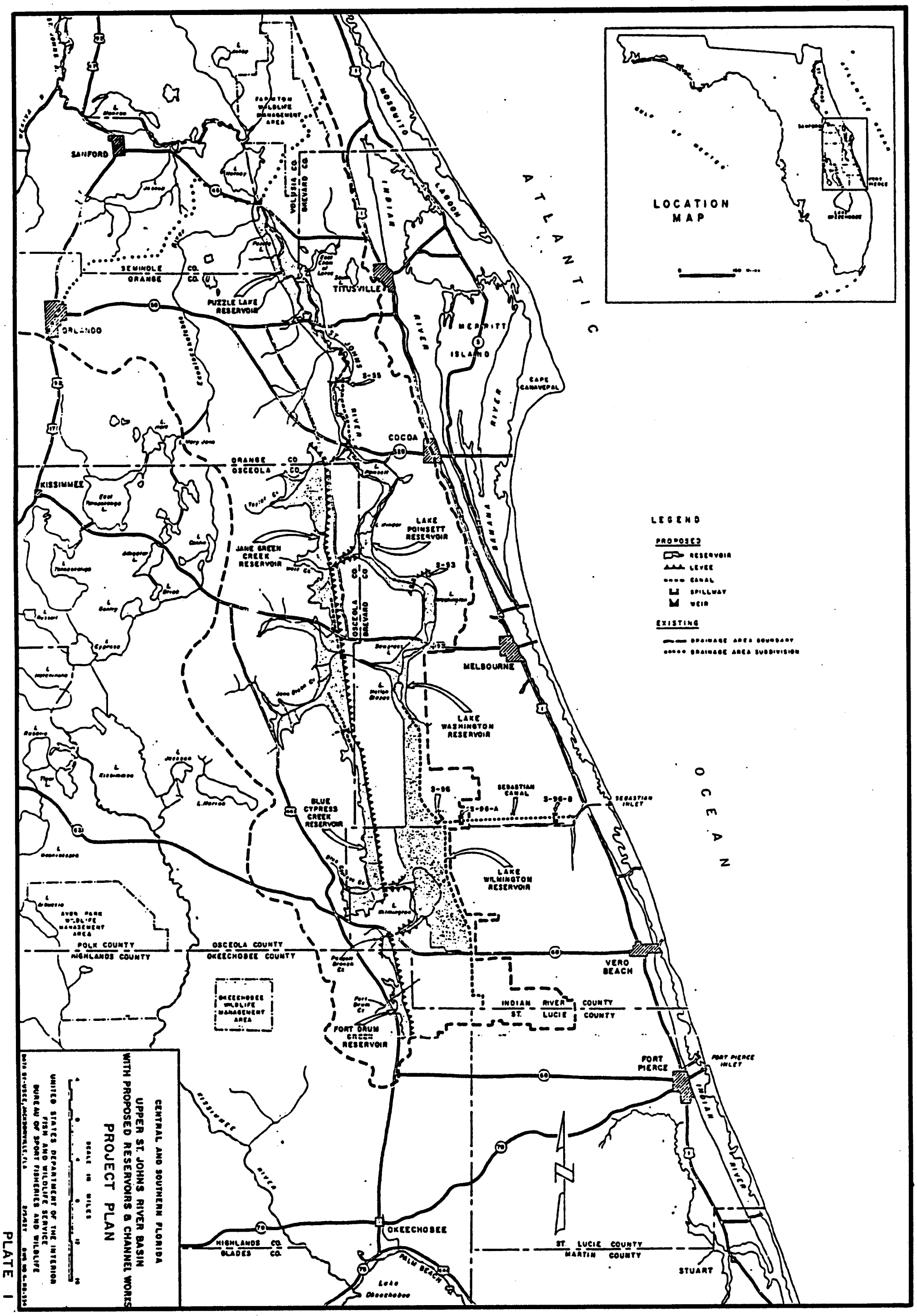

Figure 4. The 1962 Plan. *A project design was first approved in 1957, and modified in 1962. Construction of the upper St. Johns River basin portion of the vast Central & Southern Florida Flood Control Project (C&SFFCP) began in 1966. The project called for channelizing the river's floodplain, diverting floodwater from the basin eastward into the Indian River Lagoon, and storing upland runoff in a vast network of artificial retention reservoirs. The 1962 project was nearly complete before it was halted in 1969 and later abandoned because of its potential to cause environmental harm.*

In 1982, the U.S. Army Corps of Engineers determined that a plan consistent with the BDC would be economically justifiable and warranted federal participation. After considering several alternative plans consistent with the BDC, the SJRWMD Governing Board adopted a Proposed Recommended Plan in February, 1983 and requested the USACE to prepare a project GDM and EIS. The current GDM (including the EIS) was released in June, 1985. The plan was approved by the USACE s Chief of Engineers in August, 1986. Since the GDM was approved, the project design has been modified to meet local sponsor requirements. These changes are reflected in detailed engineering plans prepared by the USACE prior to construction of each planned project component. The current project reflects these design changes (Figure 5).

Passage and subsequent legal interpretation of the federal Water Resources Development Act of 1986 delayed the construction start and resulted in a new funding arrangement for the project, whereby the federal government assumed responsibility for all construction costs for project flood control and water conveyance elements. Construction of the current project began in May, 1988. A phased implementation of the project will be accomplished over about ten years.

MATERIALS & METHODS

A hydrologic simulation computer model--USJHM (Upper St. Johns Hydrologic Model)--was developed to enable water managers to plan and manage the project, as well as assess current and future project performance. Hydrologic simulations of water resources systems are an essential tool not only for planning and managing multipurpose projects, but are also critical for developing criteria for water-related environmental benefits.

Extensive hydrologic simulations were performed to generate hydrologic information not available from historical data to balance flood control and water supply benefits, and to maximize environmental benefits by proper management of the project's four MCA's (Marsh Conservation Areas) and two WMA's (Water Management Areas). The USJHM was used to generate seventy-six years of daily stage and stream flow data for the basin under historical (undeveloped), pre-project (then-existing), and project conditions. These simulation results provided a basis for determining guidelines for management and control of water levels and discharges in each plan component.

The model consists of two main elements, a rainfall-runoff simulation routine and a routing routine. The rainfall-runoff routine takes into account the basin evapotranspiration and continuously simulates soil moisture, surface retention, base flow and surface runoff by applying water balance methods. For routing mainstream discharge, the upper St. Johns River--including its entire valley floodplain from the Florida Turnpike to State Road 46--was divided into 12 hydrologic reaches, including 60 subbasins. Five of the river reaches lie upstream of US Highway 192 (within the immediate project area). Because of the flat topography of the river valley, the storage routing method was used with each reach. Each reach receives runoff from the adjacent subbasin tributaries and discharges into the downstream reach based on a discharge-storage relationship. The stage-storage-discharge data for different reaches

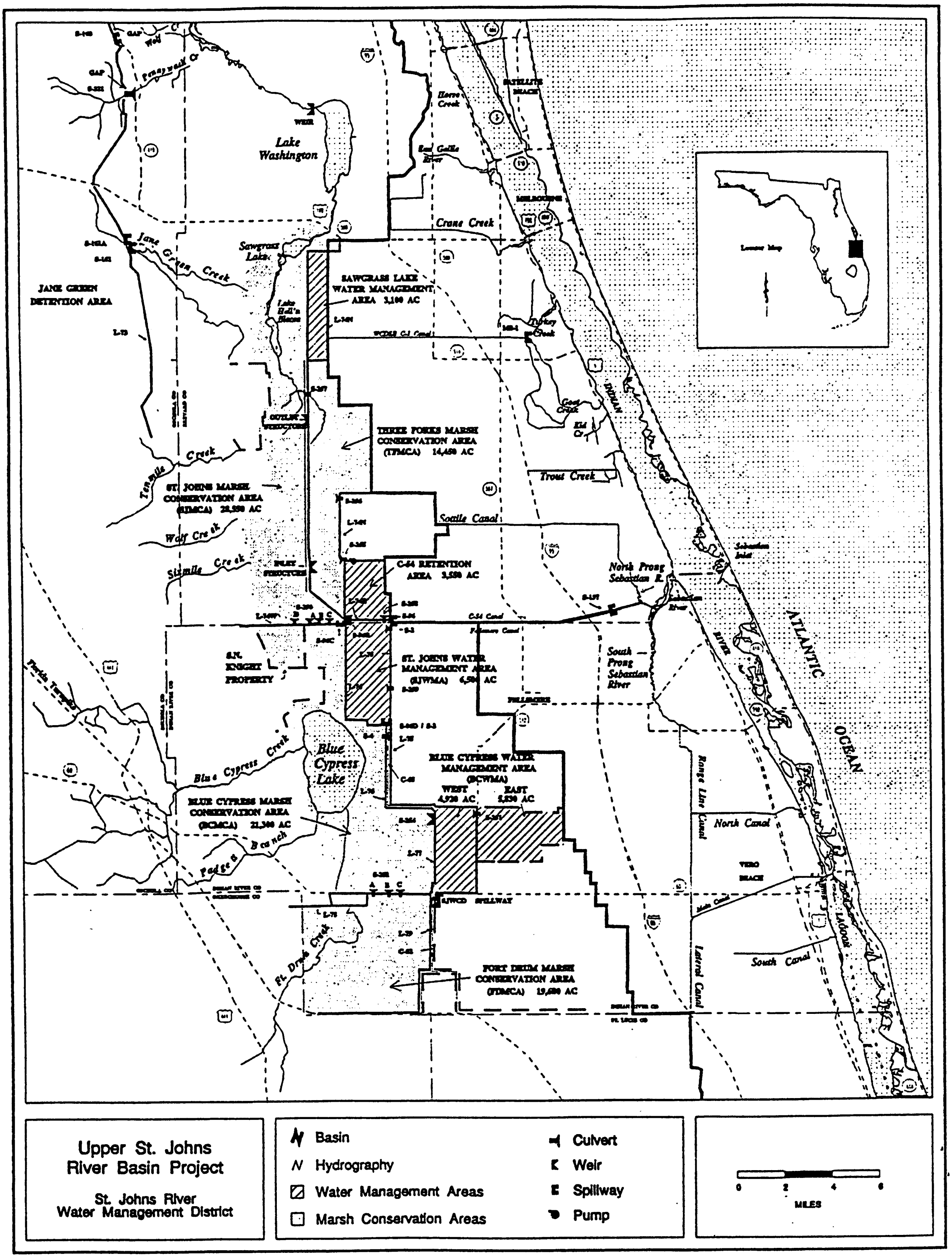

Figure 5. The current USJRBP Plan. *Construction of the current Upper St. Johns River Basin Project began in 1988, and major parts of the project have been completed. The new plan is based on a "semi-structural" water management concept because it relies less on artificial controls, and more on natural river floodplains to store and manage water.*

were developed by the USACE s HEC-2 Water Surface Profiles Program (1982) using river cross sectional data at over 100 locations.

While the primary purpose of the project is flood protection, secondary project goals include environmental enhancement and water supply. The environmental objectives of the project are to preserve and restore freshwater marsh habitats, improve water quality, and decrease stormwater discharges to the Indian River Lagoon (interbasin diversion).

Based on hydrologic simulation model results produced from literally hundreds of analyses of alternative management schemes, an Environmental Water Management Plan was developed to achieve the project s environmental objectives. The environmental plan will be incorporated into the USACE s Upper St. Johns River Basin Master Water Control Plan and will direct operation of project water control structures when water levels within the project are below flood control regulation schedules, i.e., during normal or "low-flow" conditions.

Additionally, to ensure that environmental objectives are achieved, the project attempts to restore, to the greatest extent possible, the natural hydrologic regime of the upper St. Johns River basin ecosystem. By creating a hydrologic regime which mimics natural cycles, optimum soil and vegetation characteristics will be maintained. This, in turn, will help provide other environmental benefits such as enhanced fish and wildlife habitat and improved water quality.

Several hydrologic characteristics were identified as ecologically significant for a healthy marsh system. These are: mean depths, inundation frequencies, maximum depths, magnitude of annual fluctuations, timing of fluctuations, recession rates, and minimum levels for natural lakes. Environmental hydrologic criteria were then developed for each hydrologic characteristic. These criteria comprise a series of hydrologic statistics (or constraints) that form the boundaries of an acceptable hydrologic regime for the basin. To meet environmental goals, the project must be operated within these boundaries.

Using the environmental criteria, environmental water management schemes were developed using simulated hydrologic data derived from the USJHM. Using these model results, a number of management schemes were evaluated to determine which best met the environmental hydrologic criteria. Those management schemes were then incorporated into the plan.

Although the project may be managed to solve occasional short-term problems, the environmental hydrologic criteria are designed to address long-term basin-wide hydrologic conditions. To assure the long-term health of the system, a comprehensive program to monitor biologic responses to project conditions has been developed. If monitoring data indicate that environmental objectives are not being met, or if environmental conditions change, the operating plan may be amended.

PROJECT DESCRIPTION

The USJRBP (Upper St. Johns River Basin Project) is a large, multipurpose public water project. The project design represents a "semi-structural" approach to water management which attempts to balance various environmental and economic goals. The project is semi-structural because it relies less on artificial controls, and more on the function of natural river floodplains to store and manage floodwaters. While maintaining its primary flood control objectives, the USJRBP also provides for major environmental habitat restoration and water quality protection benefits.

Although semi-structural in concept and function, the project does include over 161 km (100 mi) of flood protection levees, six large capacity gated spillway structures, and 16 smaller water control structures, culverts and weirs. The immediate project area totals some 60,703 ha or 609 sq km (150,000 ac or 235 sq mi) and is designed to accommodate the drainage of surface waters from over half of the 5,180 sq km (2,000 sq mi) watershed of the upper St. Johns River headwaters region. During flood conditions, the project may contain over 678,414,990 cubic meters (550,000 ac-ft) of water--an amount which could cover a 223 sq km (86 sq mi) area, approximately 3 meters (10 feet) deep.

A more detailed description of the project's several major hydrologic features follows.

Marsh Conservation Areas

Four large marsh conservation areas are a major environmental component of the current project. Comprised of existing and restored marshes, MCA's are designed to provide temporary storage of floodwaters generated from adjacent upland areas and thus reduce the need to discharge potentially damaging quantities of fresh water to the Indian River Lagoon. Project design criteria included provisions to closely control water levels in MCA's within environmentally acceptable limits for the marsh.

There are two MCA's south of the Fellsmere Grade and two MCA's north of the Fellsmere Grade. Areas south of the Grade include the FDMCA (Fort Drum Marsh Conservation Area) and the BCMCA (Blue Cypress Marsh Conservation Area), a combined area of about 20,234 ha (50,000 ac). Areas north of the Grade include the TFMCA (Three Forks Marsh Conservation Area) and the SJMCA (St. Johns Marsh Conservation Area), which together total over 12,950 ha (32,000 ac).

While some differences exist, MCA's in the Upper St. Johns Project will be similar in function to the large water conservation areas in the Everglades region. Design criteria for the four project MCA's (FDMCA, BCMCA, TFMCA and SJMCA) reflect the increased environmental influences of the 70's and 80's that were absent during the design and construction of the old FCD project. While MCA's will detain floodwaters, and thus occasionally function as shallow reservoir systems, releases from these areas will be made to mimic natural historic flow patterns and to restore seasonal low-flow conditions to downstream marshes. Environmental design criteria calls for untreated agricultural drainage to be initially segregated from the MCA's and stored for reuse in large off-line reservoirs called water management areas. The use of WMA's to store and treat agricultural water is expected to improve water quality of the marsh ecosystem.

Water Management Areas

In addition to extensive marsh conservation areas, over 6,475 ha (16,000 ac) of reservoir systems called water management areas (WMA's) are located south of Fellsmere Grade and east of the BCMCA. These areas include the BCWMA (Blue Cypress Water Management Area) and the SJWMA (St. Johns Water Management Area). WMA's are now under construction on former agricultural lands within the existing river valley. Because of significant soil subsidence on these lands due to agricultural activities, WMA's will be "deep water" reservoirs (in contrast with the comparatively shallow MCA's) and will be operated to provide for long-term water supply and temporary flood storage of agricultural pump and gravity discharges from the eastern portion of the basin. WMA's will also replace a portion of the floodplain storage lost to extensive floodplain encroachment.

The WMA's will also function to improve water quality conditions by separating agricultural discharge from better quality water in the St. Johns River marsh. While the primary purpose of the WMA's is to provide conventional flood storage and agricultural water supply, management efforts are planned to increase and maintain the environmental values of these areas. Because WMA's are smaller than the much larger MCA's, contain larger outlet structures, and are used to segregate and reuse agricultural waters, the use of certain environmental management tools -- such as artificial drawdowns -- may prove effective in achieving environmental objectives. The USACE and SJRWMD staffs are developing a comprehensive water control plan for the WMA's to provide for flood control and water supply benefits, while maintaining timely options for environmental enhancement.

Other Important Project Features

Other notable environmental components of the project are as follows:

<u>C-54 Retention Area</u>

A supplement to the water management area system is the C-54 Retention Area. After release of the current project GDM, the SJRWMD acquired lands to construct and operate a 1,566 ha (3,870 ac) retention area north of the SJWMA. Operation of the C-54 Retention Area will further reduce planned releases of freshwater to the Indian River Lagoon through the C-54 canal, lessening the potential adverse environmental impacts of large fresh water discharges on the lagoon.

A public/private partnership to enhance waterfowl habitat within the C-54 Retention Area is also planned. The private conservation group, Ducks Unlimited, has provided major funding for a waterfowl management area within the C-54 Retention Area. On-site management by the Florida Game and Fresh Water Fish Commission will help ensure a productive waterfowl area as part of the WMA system.

Restored Areas

In the early 1980's, the St. Johns River marsh between the Fellsmere Grade and Lake Washington--the river reach located in southern Brevard County--totaled only 10,930 ha (27,000 ac). This area accounted for less than 20 percent of the historical floodplain in this river reach. Current project plans include the restoration of an additional 14,690 ha (36,300 ac) of riverine floodplain in this area. These restoration areas were previously developed for private agricultural uses. The restoration of these lands will replace a portion of the flood storage and conveyance lost due to encroachment and restore the area to a viable marsh ecosystem.

Canal Plugs

In 1986, the SJRWMD, in cooperation with the Florida Department of Environmental Regulation (now the Department of Environmental Protection), constructed nine earthen canal plugs in existing borrow canals adjacent to remnant marshes in southern Brevard County as part of an early implementation of the Federal project within the SJMCA. The plugs improve sheetflow, water quality conditions, and limit the overdrainage of water from the marsh during dry periods. Eight plugs are located in the C-40 and C-40 Extension canals (a/k/a "No Name Levee") along the east side of the marsh. One plug was constructed in a borrow canal on the west side of the marsh. Additional canal plugs are planned for the western side of the valley and at other river reaches north of the federal project area to further improve marsh conditions.

Improvements to Upland Areas West of the St. Johns River

A 56 km (35 mi) long levee (L-73) and several large gated spillway structures were constructed along the western uplands of the river as part of the 1962 project. If completed, the L-73 system would have created several connected upland reservoirs. However, when project construction was stopped in 1969, only one upland impoundment--the Taylor Creek Reservoir--was fully operational. Under the current project, the L-73 upland system was modified to achieve flood control and environmental benefits.

As part of the L-73 system, the Jane Green Creek Detention Area has been redesigned to provide temporary detention of upland stormwater without causing significant environmental degradation to the system. The detention area now consists of about 6,475 ha (16,000 ac) located west of L-73 and south of US 192. A new gated spillway structure (S-161A) will operate to provide short-term detention of flood waters and to maintain base flow to the creek system and river reaches downstream during normal hydrologic conditions.

PROJECT COSTS

The benefit/cost ratio of the 1957 project was 2:1; two dollars in benefits were projected for each dollar of cost. The 1985 project GDM contains a favorable benefit/cost ratio of 1:.7, with no economic values calculated for purely environmental benefits. Estimated project costs (in 1994 dollars) are as follows:

Project lands	$87.3M
Flood damage reduction	46.5
Planning & design	9.9
Relocation costs	11.3
Recreation construction	4.7
Total project cost	$159.7M

Under the current cost-sharing arrangement, the federal government will pay all engineering design and capital construction costs. The SJRWMD is responsible for all land acquisition. Recreation development costs are shared 50/50.

Since land acquisition costs amount to nearly 70 percent of the total project cost, no discussion of project costs would be complete without describing the climate in which the SJRWMD pursued an aggressive land buying program to support project implementation. In the early 1980's several factors--including more stringent development criteria, greater pressure for private on-site treatment of agricultural waste water, an economic recession marked by high interest rates and rising land values, successful SJRWMD bonding efforts for land acquisition and implementation of the Save Our Rivers documentary stamp tax increase to fund environmental lands, coupled with positive public sentiment toward programs to improve environmental conditions--all contributed to conditions favorable to the public acquisition of environmentally sensitive lands. To date, the SJRWMD has exercised its eminent domain powers to acquire only two of the some twenty-five large land parcels needed for the project. All other land acquisitions have been achieved through voluntary negotiation with local landowners.

PROJECT BENEFITS

The several public benefits of the Upper St. Johns River Basin Project are summarized as follows:

Flood Protection and Flood Damage Reduction

While the USJRBP represents an ambitious environmental river and wetlands restoration effort based on sound ecological criteria, the primary thrust of today's project remains flood control and flood protection. Flood protection benefits are achieved through both lowering peak flood stages and constructing or improving flood protection levees to USACE standards. All project levees provide protection against the SPF (Standard Project Flood)--a storm event which has a return period of more than 200 years.

Upon completion of the project, flood stages will be reduced by about 0.46 m (01.5 ft) in the Blue Cypress Lake region and by about .15 m (0.5 ft) at Lake Washington at the downstream end of the project.

Water Quality Improvements

Water quality improvements are expected because agricultural waters south of the Fellsmere Grade are stored in the two water management areas (BCWMA and SJWMA) and the C-54 Retention Area. These waters are thus separated from the marsh conservation areas (FDMCA, BCMCA, TFMCA and SJMCA) and the St. Johns River marsh downstream of these areas. Better water quality is maintained in the marsh conservation areas and can be discharged to augment downstream flows. The storage of agricultural waters in the WMA's and the construction of canal plugs to restore sheet flow of water through the marsh will reduce the concentrations of suspended solids, turbidity and nutrients.

Improved Water Conservation Storage

The creation of MCA's and WMA's will improve both temporary flood storage and long-term water conservation storage. More water will be available because less water is diverted out of the basin and discharged downstream during high water conditions. Instead, excess water during the wet season can be stored and reused for beneficial purposes during the dry season. Computer simulations show that during a 1-in-5 year drought about 66,385 ha-m (50,000 ac-ft) of additional water, on an annual basis, would be available under project conditions. This water could be allocated from the WMA's for irrigation demands and to augment downstream flows for water supply and environmental enhancement.

Even with low flow discharges made to the marsh north of the Fellsmere Grade, computer simulations show that due to the increase in water storage in the WMA's and the FDMCA, the stage-duration characteristics of other MCA's under project conditions will be improved.

Reduction in Interbasin Diversion

Under project conditions, discharges through C-54 to the Indian River Lagoon will be significantly reduced. Computer simulations show that there will be no discharge through C-54 to the lagoon during the 10-year storm event. Therefore, the frequency of discharge to the lagoon will be less than once every 10 years. By operating the C-54 Retention Area, the frequency and volume of discharge to C-54 and the lagoon will be further reduced. In addition, under the SJRWMD regulatory program, a portion of the water historically diverted from the basin to the lagoon through the Fellsmere Main Canal now drains to the SJWMA.

Improved Marsh Conditions and Wildlife Habitat

Throughout the basin, the environmental quality of the marsh will be improved, resulting in enhanced water quality conditions and an increase in productive fish and wildlife habitat. Upstream from the Fellsmere Grade, the FDMCA and BCMCA will operate to provide major water storage benefits and to augment downstream flows consistent with acceptable marsh conditions. North of the Fellsmere Grade, the functional marsh will be increased by more than 13,070 ha (32,300 ac). The hydrologic regime of the marsh will be improved through the storage and continued discharge of water from the MCA's.

Public Recreational Opportunities

Several public recreational sites are now under consideration as part of the project. These sites include the following: FDMCA multi-use recreation area, improvements to Blue Cypress Lake County Park, BCMCA/BCWMA boat ramp, SR 512 bank fishing site, SJWMA/C-54 boat ramp, C-54 bank fishing, improvements to Lake Washington County Park, and a multi-use recreation plan for the Jane Green (Bull Creek) Detention Area.

The RAC (Upper St. Johns Recreational Advisory Council) was formed in 1985 to aid the SJRWMD in the development of a public recreation plan for the upper basin area. Based on input from the RAC and the SJRWMD, the USACE has developed a comprehensive Master Recreation Plan for the entire project area.

SUMMARY

The Upper St. Johns River Basin Project is a major public water project designed to address several water resource-related needs including increased flood control, improved water supply and water quality conditions without adversely compromising the environmental values within the basin. A major objective of the project is to achieve these benefits through extensive floodplain restoration.

The project is called a "semi-structural" water management project because nature's floodplains -- rather than artificial upland reservoirs -- will function to store and purify flood waters. The project has been recognized by environmental groups and agencies as one of the most unique and ambitious environmental river restoration projects in the world. The Florida Department of Environmental Protection has hailed the current upper basin plan as a "national model of modern floodplain management" citing that the project will result in significant public benefits to the entire upper St. Johns River basin area.

Overshadowed by larger and more politically sensitive and controversial water projects, public plans have emerged and evolved to manage the headwaters of the St. Johns River for nearly fifty years. Had earlier versions of the Upper St. Johns Project been completed, state water managers would be faced today with a much more costly cleanup of Florida's longest river and viable options to protect the Indian River Lagoon from unacceptable fresh water discharges would now be economically non-existent. Compared with the skyrocketing price tag for planned Kissimmee River restoration efforts and cleanup of Florida's Everglades system, the $100 million in state funds spent to restore the St. Johns River may yet prove to be the best public water deal of this decade.

ACKNOWLEDGMENTS

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implementing, and managing the current Upper St. Johns River Basin Project. To this team belongs credit in large measure for the project's success.

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HEAVY METALS REMOVAL WITH WATER MILFOIL (MYRIOPHYLLUM SPICATUM) IN CONSTRUCTED WETLAND

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ABSTRACT

Toxic heavy metals contamination is often detected in wetlands, impoundments, estuaries, coastal areas and other ecosystems. The sources of metal contamination are usually from marina or harbor activities, boat and shipping industry, canal run-off, sewage effluent, industrial waste discharge, etc. Conventional remediation methods such as precipitation chemical oxidation or reduction, ion exchange, filtration, membrane technologies or evaporation process are generally impractical or ineffective to be applied in an ecosystem. The use of living plant biomass for removing metals from an ecosystem provides a potential for cleanup of the wetland or estuarine coastal environments.

INTRODUCTION

Growing public awareness and increased scientific knowledge have enunciated environmental problems such as coastal water pollution, loss and degradation of habitats, contaminated water and sediment in ecosystems and the continuing conversion of open land to urban development. Stormwater runoff from urban sites and other non-point source pollution that carries heavy metals, oil, hydrocarbons, pesticides, nutrients and other suspended solids have degraded water quality in many of the nation's wetland ecosystems. One of the key elements of the U.S. Environmental Protection Agency Stormwater runoff regulation is the promotion of the use of best management practices for controlling non-point pollution sources (Yu, 1995). These management practices include detention ponds, infiltration basins, vegetative stripes and constructive wetlands. Pollutant removal through constructive wetland or detention ponds are mainly by gravitational particle-settling, chemical precipitation of metals, filtration of organic matter, and biological uptake or adsorption by aquatic vegetations. Using aquatic plants for bioremoval of pollutants plays an important role in wetland ecosystem remediation.

Aquatic plants are known to accumulate metals and other toxic elements from contaminated water (Wolverton and McDonald, 1975; Muramoto and Ohi, 1983; Green and Bedell, 1990; Wilde and Benemann, 1993). Bioremoval process using aquatic plants often exhibits a two-stage uptake process: an initial fast, reversible metal binding process (biosorption), followed by a slow irreversible, ion sequestration step (bioaccumulation). The initial biosorption by different parts of cells can occur via complexation, coordination, chelation of metals, ion exchange, adsorption and microprecipitation. The second state of bioaccumulation process is an active mode of metal accumulation by living cells. This process of sequestration is dependent on the metabolic activities of the cell, which in turn can be affected by the presence of the metallic ions (Wilde, W.E., and Benemann, J.R., 1993).

Plants exhibit large variability in their capacity to sequester toxic elements. The adsorption capability varies with its physiological state, age, growing water quality as well as other environmental conditions during the actual biosorption process such as pH, temperature, and presence of certain co-ions. Therefore, a plant screening program is the required first step in development of a bioremoval process in a constructed wetland systems. A successful candidate would exhibit (1) capability to reduce metal concentrations to the required regulatory level; (2) high specific metal adsorption capability; (3) capability of removing several metal ions, simultaneously; and (4) high productivity in a low cost cultivation system; ease of harvesting, processing, storage, transportation and disposal.

The objective of this study is to describe methods to identify promising aquatic plants which possess the best characteristics for use in metal removal process in the wetland or detention pond. This was accomplished by first screening a number of potential plants and then choosing the promising plants to establish adsorption isotherms in order to obtain the maximum specific adsorption capability (mg metal adsorbed/kg of dried biomass) and bioconcentration factor (the ratio of specific adsorption to residual metal concentration) of the plants.

METHODS AND PROCEDURES

A small scale biomass metal contacting experiment was performed to screen the optimal plant species. With 50 mL of metal solution and any desirable amount of biomass added to 125 mL polyethylene flasks, the flasks were placed in a shaker and allowed to contact for a sixty minute period. At the end of each experiment, the content of flasks were filtered or centrifuged to separate the biomass from the solution. Both biomass and the filtrate were analyzed to determine metal contents. The experimental results were used to calculate (1) percent of metal remaining in solution; (2) percent metal recovered by biomass; (3) specific adsorption per unit weight of biomass; (4) bioconcentration factor. These parameters evaluated the metal adsorption characteristic to select suitable plant for treating stormwater runoff in the wetland ecosystem. After initial screening of plant species, a series of experiments were performed with various metal concentrations in the selected plants. The obtained specific adsorption was plotted against the residual metal concentration to define the sorption characteristic for the plant biomass. The maximum adsorption was calculated using the Langmuir adsorption equation (Volesky, 1990). $C/Y = C/Y_m + 1/kY_m$, where Y_m was the maximum adsorption; k, the equilibrium constant related to the affinity of the binding site; and Y, the specific adsorption at residual metal concentration C. From a plot of C/Y vs. C, the slope ($S = 1/Y_m$) gave Y_m and the intercept ($I = 1/kY_m$), gave k constant. Each metal biomass adsorption characteristic was evaluated with Y_m, C and k values. The maximum adsorption capability and residual metal concentration were important features shown in the sorption isotherms. A steep isotherm from the origin at low residual concentration indicated high affinity of the biomass for the given metal species.

RESULTS

Water milfoil (Myriophyllum spicatum), hydrilla (Hydrilla verticillata), hygrophyllum (Hygrophila polysperma), water lettuce (Pistia stratiotes), and alligator weed (Alternanthera phiuxeroides) were initially examined in the screening program. Water milfoil was found to be one of the most promising plants in the study. The initial screening results show that with biomass density of 0.01 kg/L in a solution containing 0.51 mg/L of Cd and 3.97 mg/L of Zn, the plant exhibited the specific adsorption of 525 and 4,770 mg/kg and bioconcentration factor of 4,770 and 2,615 for Cd and Zn, respectively. With these promising results, five metals, Cd, Zn, Ni, Pb and Cu were then used to establish adsorption isotherms with initial metal concentrations ranging from 2.5 to 20 mg/L and contact time at 60 minutes, the maximum adsorption capability for water milfoil were 8,200 mg/kg Cd; 13,500 mg/kg Zn; 55,600 mg/kg Pb; 5,800 mg/kg Ni and 12,900 mg/kg Cu. The initial pH in water solution was 3 and then quickly increased to pH 7.0 after biomass was added to the solution. Experiments were also conducted to test the ability of the biomass to lower the metal concentration below the EPA surface water discharge criteria. The results indicate that the minimum residual concentration for Cd, Ni and Cu was about 0.01 mg/L and for Zn and Pb were 0.1 and 0.004 mg/L, respectively. All the metals except Cd were within the EPA quality criteria (USEPA, 1986). The effect of contact time on Cd adsorption by water milfoil was tested at pH 7. More than 50% of the metal was adsorbed by the end of the first minute and 95% of adsorption was complete within 30 minutes.

CONCLUSIONS

Water milfoil is naturally immobilized and is a very common component of many natural communities. Those characteristics as well as the high metal adsorption capability and relatively short adsorption time serve as an indication that this plant can be used for metal bioremoval process for a constructed wetland or detention pond.

Using Water milfoil for treating storm water runoff in wetland system offers some advantages such as: (1) metals at low concentration can be selectively removed to meet regulatory level. (2) this process can result in the recovery of the metals bonded by the biomass. The metal laden biomass can be reduced in volume by drying or incineration, thereby reducing the volume of hazardous waste. (3) Biosorption process can be performed in detention pond or constructed wetland with contaminated water flowing over plants for metal removal. (4) Water milfoil biomass has a very low affinity for calcium and magnesium ions in natural water bodies. This serves as a very attractive characteristic for Water milfoil to treat stormwater. (5) The system offers low capital investment and low operating cost. They can be operated over the broad range of pH value (3-9) and temperature (4 to 90°C).

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THE SEMINOLE EXPRESSWAY MITIGATION SITE; CONSIDERATIONS FOR FUTURE DESIGN AND CONSTRUCTION OF SIMILAR LARGE-SCALE RESTORATION SITES

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ABSTRACT

Careful consideration of design features during constructibility reviews of large-scale mitigation projects can help to ensure a successful construction phase. The construction of the Seminole Expressway Mitigation Site illustrated this concept repeatedly during its 21-month construction duration. In order to mitigate wetland impacts associated with the construction of the expressway, Florida's Turnpike completed a project which included restoration of historic water levels on a 701-hectare (1,735-acre) site and the installation of 60,900 trees of various wetland species on 57 hectares (142 acres) of the site. The three most problematic issues requiring resolution during construction included 1) obtaining plants which met plan specifications, 2) providing water for plant establishment, and 3) providing accessibility for construction, replanting, monitoring and maintenance activities. The logistics of resolving these issues were further complicated by the size of the site and harsh site conditions. On-site resolution of these issues was challenging and costly in terms of both schedule and budget. The need for constructibility review of critical design elements is addressed in order to aid future designers of large mitigation sites.

INTRODUCTION

Most environmental scientists and construction engineers who have experienced wetland mitigation construction in the field would probably agree that these construction projects usually require a high degree of in-field design modification. Conducting detailed constructibility reviews for wetland mitigation designs could greatly reduce the need for in-field modifications. The U.S. Army Corps of Engineer's (USACE) definition of constructibility is well suited for application to wetland mitigation construction:

> "the compatibility of the design with the
> site, materials, methods, techniques, schedules
> and field conditions".

The USACE considers constructibility review to encompass three major goals:

1. ease of construction and enhancement of contractor productivity;

2. adaptation of design features to site conditions and restrictions; and

3. analysis of tradeoffs between standard and alternative components.

Constructibility issues which surfaced during the construction phase of the Seminole Expressway Mitigation Site correspond directly to the three major goals presented by the USACE within their definition. These issues, along with preventive measures or proposed resolutions, are discussed with the intent of providing a positive application of a challenging construction experience to future designers and contractors.

THE SEMINOLE EXPRESSWAY MITIGATION SITE: A CASE STUDY

Project Introduction

Construction of the Florida Turnpike's (Turnpike) largest wetland mitigation project was completed in 1993 to offset wetland impacts associated with the 29-kilometer (18-mile) Seminole Expressway. The mitigation project was designed to offset impacts to Florida Department of Environmental Protection (FDEP), St. John's River Water Management District (SJRWMD), and USACE jurisdictional wetlands amounting to 56, 75 and 66 hectares (139, 185 and 164 acres), respectively. Most impacts were to forested wetlands and included 6.7 hectares (16.7 acres) in environmentally sensitive forests such as Black Hammock Swamp.

Through a joint agreement, the Turnpike used a 701-hectare (1,735-acre) SJRWMD land tract located on the northwest shoreline of Lake Jesup in Seminole County (Figure 1). This broad, flat expanse of rangeland formerly constituted a large portion of the Lake Jesup floodplain. It had been grazed by cattle for several decades which was made possible through considerable dewatering efforts. A 4.6-meter (15-foot) wide, 3.2-kilometer (2.0-mile) long levee had been constructed along the eastern site boundary, and an elaborate network of drainage canals, ditches and swales had been designed to outfall to Lake Jesup with the aid of a pumping station.

Elevations on site ranged from below 0.3 meters (1 foot) NGVD (National Geodetic Vertical Datum) at the south end, to above 1.8 meters (6 feet) NGVD at the north end; most of the site was 0.3-0.6 meters (1 - 2 feet) NGVD. Site vegetation was dominated by rangeland species. Wet prairie, the historic plant community, surrounded the site with sand cordgrass (Spartina bakeri) as the dominant groundcover. A forest of cabbage palm (Sabal palmetto), sugarberry (Celtis laevigata) and live oak (Quercus virginiana) occupied the higher elevations at the north end of the site.

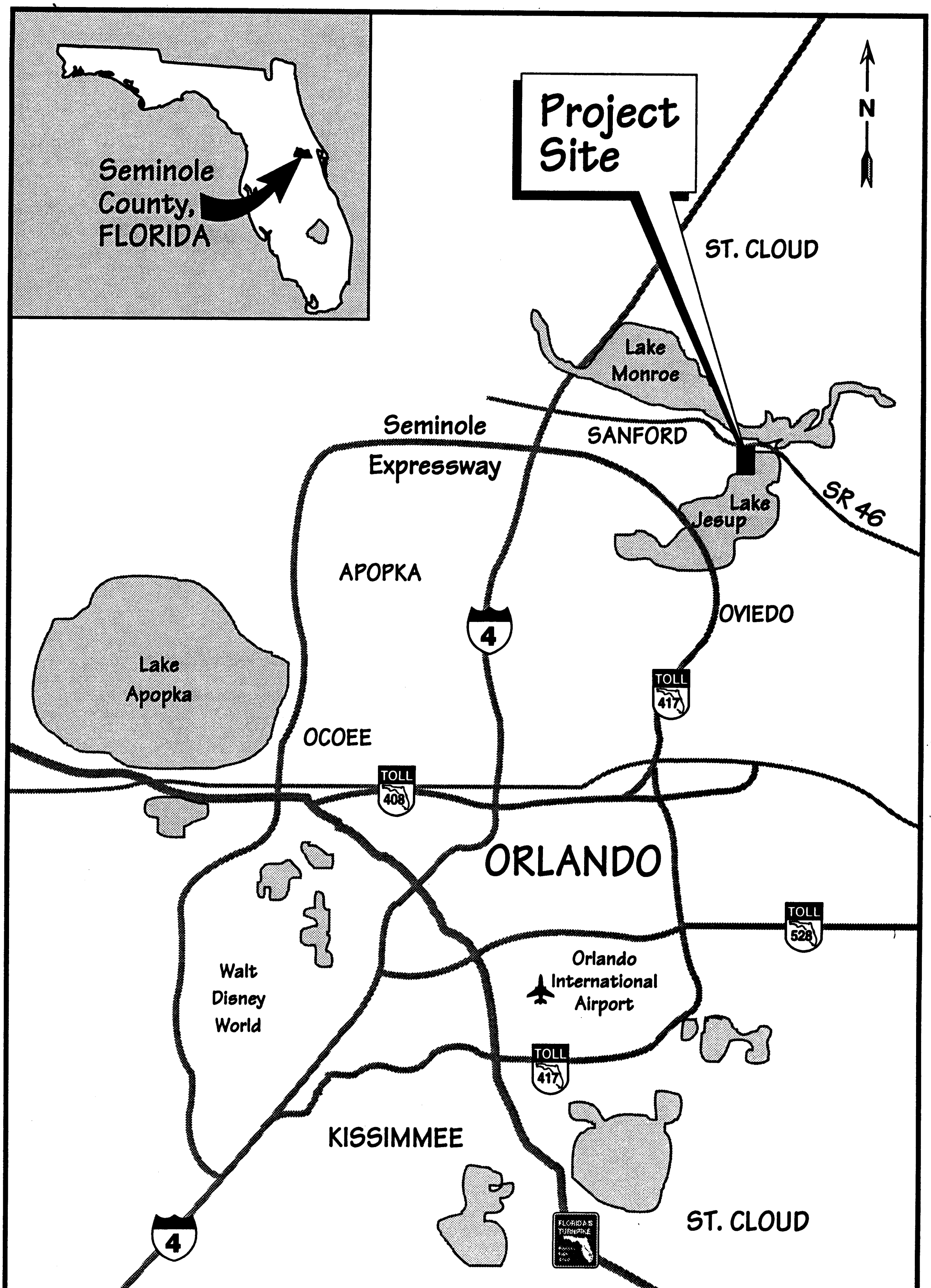


Figure 1. Project Location Map

Mitigation Design

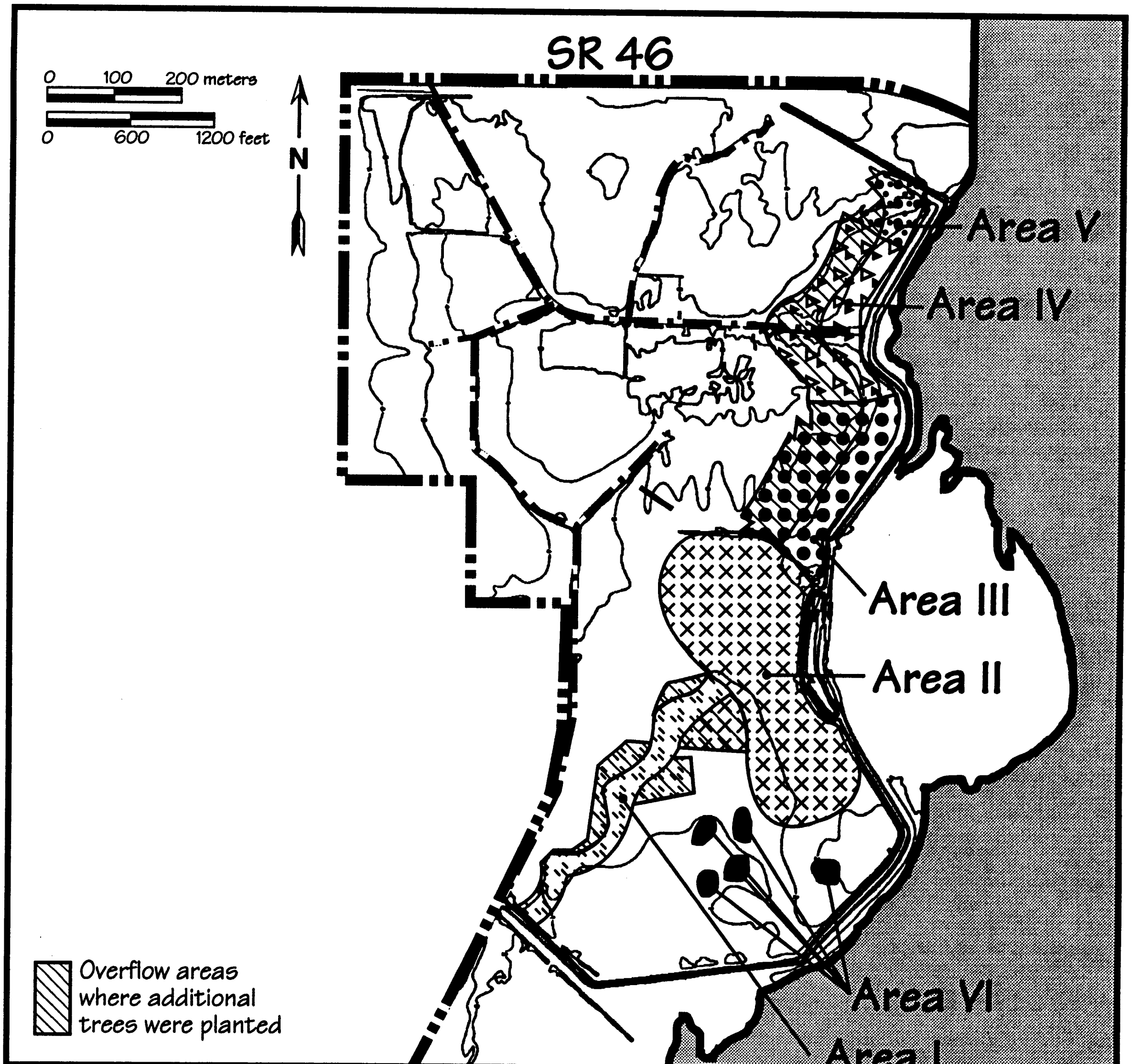
Mitigation design specified restoring the historic hydroperiod (determined using SJRWMD hydrographic data) over a 222-hectare (550-acre) restoration/enhancement area and establishing a 57-hectare (142-acre) mixed hardwood wetland forest. To accomplish restoration and enhancement, project design required restoration of wetland hydrology and removal of nuisance species. In order to circumvent the elaborate system of swales, ditches and canals, earth plugs were installed at numerous locations within these artificial drainage features. The most significant rehydration measure involved excavating 7 levee breaches and restoring 304 linear meters (1,000 feet) of the levee at the south end of the site to natural grade.

Mitigation design also included wetland creation, which is divided into six (I-VI) different planting areas (Figure 2). Area I occupies the lowest elevations (to 0.5 meters, or 1.6 feet, NGVD) and Area V occupies the highest elevations (to 1.5 meters, or 5.0 feet, NGVD). Area VI, nicknamed the Rookery Islands, consists of isolated plantings in the southeast portion of the site. The Rookery Islands were designed to provide rookery habitat for wading birds. Within these creation areas, the design specified the installation of 60,900 3-gallon trees on 3-meter (10-foot) random centers. Species selected by the designers were those typically found in the local Black Hammock Swamp, and included bald cypress (Taxodium distichum), red maple (Acer rubrum), tupelo (Nyssa sylvatica), pop ash (Fraxinus caroliniana), sugarberry (Celtis laevigata), laurel oak (Quercus laurifolia), dahoon holly (Ilex cassine), buttonbush (Cephalanthus occidentalis) and American elm (Ulmus americana). Hydrological tolerances for each species were used as the basis for designing each planting area. The design high and design low water levels were 3.0 and 0.5 meters (9.8 and 1.7 feet), respectively, and normal water level was 0.9 meters (3.0 feet).

Design criteria for plant material were carefully detailed in the construction plans and specifications. At a minimum, all trees were to meet the USDA standards for Florida Grade No. 1 nursery material. Tree height was to be 1.2-1.8 meters (4-6 feet). Plant installation methods outlined by the design and incorporated into the construction plans, required backfilling each tree well with a specific mix of topsoil, sand and fertilizer.

Mitigation Construction

Prior to the initiation of construction, cattle were relocated. In addition, baseline vegetation studies required by permits were conducted (Florida's Turnpike/PBS&J, November 1991). As suspected, the surveyed plant communities reflected the site's history of physical and hydrological disturbance. Forty percent of the sample stations were dry and dominated by graminoid species, the most common being paspalum (Paspalum spp.) and Bermudagrass (Cynodon dactylon). Ninety-five percent of the most frequent taxa were characteristic of pasture or rangeland, and wet prairies associated with a riverine system. The first phase of construction involved disassembly of the pumping station, blocking of drainage features internal to the site, and the required levee earthwork. The second phase involved the installation of planting grid markers, trees, fencing and signing.



AREA	ELEVATION	AREA	TYPE	
I	0.5-0.6 m (1.7-2.0 ft)	12.6 ha (31.1 ac)	Cypress Forest	
II	0.6-0.7 m (2.0-2.5 ft)	33.7 ha (83.5 ac)	Mixed Wetland Hardwoods	
III	0.5-0.6 m (1.7-2.0 ft)	9.0 ha (22.5 ac)	Cypress Forest	
IV	0.6-0.9 m (2.0-3.0 ft)	14.6 ha (36.1 ac)	Mixed Wetland Hardwoods	
V	0.9-1.5 m (3.0-5.0 ft)	2.5 ha (6.2 ac)	Mixed Wetland Forest	
VI	0.2-0.9 m (0.5-3.0 ft)	2.0 ha (5.0 ac)	Rookery Islands	
TOTAL		74.4 ha (184.4 ac)		

Figure 2. Mitigation Site Plan

Modifications to the project design were necessary in order to successfully complete project construction. Ultimately, these field and design modifications required formal modification of the FDEP permit (Table 1). The permit modifications involved changes in planted area, density and species composition, as well as revision to the permitted success criteria. In addition to design and permit modifications, a substantial schedule extension was also necessary to complete construction of the mitigation site. The original budget was $1,243,400 and 365 contract days were allotted for construction. Both the budget and the schedule were exceeded; construction was completed in 558 contract days (57% increase), at a cost of $1,432,340 (15% increase).

Constructibility Issues

Examples of how common mitigation site constructibility issues relate to the three USACE constructibility review goals are discussed below, using the Seminole Expressway Mitigation Site as a case study.

Ease of construction and enhancement of contractor productivity:

Large mitigation sites merit the construction of temporary or semi-permanent roads. Adequate access to the entire site as well as to internal portions of the planting areas is paramount to contractor productivity. Accessibility is not only important during construction for transport of materials and laborers, it is critical to conducting cost effective post-construction maintenance and monitoring. The most serious impediment to construction efforts on the Seminole Expressway Mitigation Site involved site inaccessibility and unworkability caused by extremely wet site conditions. With the historic wetland hydroperiod restored during the first part of construction, site accessibility during the site's wet season became a daily challenge. Four-wheel drive vehicles and heavy equipment bogged down easily in the deep muck soils. Tree planting crews were also slowed considerably by the difficulty encountered in transporting workers long distances through deep water and muck to the planting site. Site flooding also presented physical problems to those workers charged with planting 1.2-meter (4-foot) tall trees in as much as 0.9 meters (3 feet) of standing water.

Large, relatively uniform sites also warrant establishing a permanently numbered grid system, on the ground, for logistical and communication purposes. This quickly became important for the Seminole Expressway Mitigation Site, a large expanse of featureless floodplain. Ready location of field personnel, equipment and materials on the site was critical to the identification of field problems to the contractor, the inspection and enumeration of trees following installation, and site safety.

Accessibility and on-site location challenges were resolved in part through in-field modifications made by field personnel. These modifications included the following:

Table 1. Field and permit modifications required during construction of the Seminole Expressway Mitigation Site.

FIELD MODIFICATIONS	REASON
Planting Area I not completed. Portions of Areas II, II and IV not planted.	Water levels too high to provide access pathways.
Quantities of tree species were changed.	Only *Taxodium* spp. and *Acer rubrum* were available in specified quantities.
Cephalanthus occidentalis was substituted for *Salix caroliniana* in Rookery Islands (Area VI).	*Salix* was not grown commercially.
Selected species were eliminated and substituted with other species.	Either quantities of Florida No.1 were unavailable or species more tolerant of high water levels were required.
Areas I-V were expanded.	Additional area was created to replace dead & substandard trees without harming established trees.
PERMIT MODIFICATION	**REASON**
Planting Area I not completed. Portions of Areas II, III and IV not planted.	Overflow planting areas were necessary to compensate for replacement of dead trees and creation of unplanted access pathways.
The tree density was decreased from 435 trees/acre to 285 trees/acre.	Planting area was increased but number of trees remained the same.
Some plant species were substituted.	Commercial availability was lacking.
Control wetland was redefined.	Species substitutions would have caused similarity index tests to fail.

1. Specific haul and access routes were designated in order to minimize impacts to beneficial native vegetation;

2. Permitted wetland creation acreage was relocated because it became imperative to create access roads inside Planting Areas II and IV in order to minimize disturbance to newly planted trees. These unplanted access roads also help to minimize inadvertent tree damage caused by airboats during high-water events;

3. Cypress trees from Area I were relocated above elevation 0.3 meters (1 foot) NGVD in specially created overflow areas because water levels remained too high to successfully install trees at the south end of Area I; and

4. A permanent grid system was installed and numbered in order to expedite construction and post-construction activities.

Adaptation of design features to site conditions and restrictions:

For large mitigation sites with a hydrologic history of extreme fluctuation in water levels, provisions (beyond those of standard landscape design) for watering during both the plant establishment period and potential drought periods should be considered during design. On the Seminole Expressway Mitigation Site, with past hydrographic data showing that water levels could vary as much as 1.5 meters (5 feet), pre-construction planning for watering trees would have greatly aided the construction team. Almost as serious an issue as flooding, drought conditions parched the site twice during plant installation. Prolonged droughts resulted in higher than anticipated tree mortality. Without plans for site irrigation, the contractor was compelled to watch and wait as trees died back to below contract-specified heights. Drought preparedness could have prevented the replanting of approximately 15,000 trees. Fire never consumed the Lake Jesup site during these droughts, however it was a constant threat.

Field construction personnel employed various measures and spent considerable effort to resolve the need for irrigation during drought periods. When the first drought persisted well into the contractor's tree warranty period, the contractor teamed up with the construction engineer and designer to improvise a field irrigation system consisting of thousands of meters of flexible hose and four portable hydraulic pumps. Irrigation was accomplished primarily by sheetflow. Irrigation water was drawn from the site canal interior to the levee, but eventually this canal ran dry.

Other potential avenues for providing water which could have been considered during project planning include the use of on-site artesian wells (plugged during Phase I of construction) and agricultural pumping equipment.

Analysis of tradeoffs between standard design components and alternative components:

For wetland mitigation projects, the development of appropriate specifications for plant material is imperative. For the Seminole Expressway Mitigation Site, and many other mitigation projects, Florida Grade No. 1 trees are the standard construction components specified. This grade of tree mandates specific tree form as well as tree vigor and health requirements. Availability of trees became a constructibility issue for this project because the total contract number of trees exhausted the commercial availability of Florida Grade No. 1 trees for all contract species. Thousands of trees were rejected prior to planting because the roots were either diseased, pot-bound, or not adequately developed. Because plant health is more important than plant form (aesthetics) for mitigation sites, field personnel resolved this issue in part by concentrating inspection efforts on the specifications for root development and tree height only. Except for bald cypress and red maple, demand could not be met, even with these reduced standards.

DISCUSSION

Constructibility and design reviews conducted at the early stages of a construction project are critical to controlling the schedule and budget of a project during the construction phase. Conducting adequate reviews of this type offers the opportunity to improve the design by identifying omissions, ambiguities and inadequacies; this aids in preventing problems during the actual construction, and ultimately the project benefits through shortened construction duration. In addition, proper review can contribute to lower operating and maintenance costs, and encourages the consideration of more economical designs.

Reviews performed by a well-coordinated group of personnel (ecological, engineering, construction and operation) can greatly aid in resolving some of these issues before the project goes to construction, thereby reducing in-field modifications. Surveys and studies conducted by the USACE for their own projects (U.S. Army Construction Engineering Research Laboratory, 1983, and Mogren, 1986) identified three major causes of contract modifications: design deficiencies, user-requested changes, and unexpected site conditions. The USACE determined that 56 percent of the modifications were to correct design deficiencies. In addition, a publication prepared by the Construction Industry Institute (1986) suggests that savings from 6 to 23 percent of the original estimate can be achieved through proper constructibility review. Although these statistics were prepared based upon civil engineering projects, the construction experience of the authors suggests that to expect similar savings for wetland mitigation sites would not be unrealistic.

Due to the inherent nature of wetland mitigation construction, certainly not all potential construction difficulties can be resolved pre-construction. Changes in site conditions from time of design to time of construction can play a significant role in determining the amount of in-field modification which will be required, and these changes can not be anticipated. Even so, designers of mitigation sites should consider the following three ideas for large-scale projects:

1. Establish appropriate plant specifications which can be met by commercial growers without sacrificing plant vigor;

2. Provide for permanent, unplanted, access pathways into extremely large planting areas which can be used by both construction and post-construction personnel; and

3. Provide for plant watering which is beyond standard landscape design, and/or on-site water level control, for sites where extreme water level fluctuations are anticipated.

For large-scale projects, attention to these three details during project planning and design may greatly enhance the construction effort.

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INVASIVE NON-INDIGENOUS SPECIES IN FLORIDA'S PUBLIC LANDS

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The report to the Florida Legislature on the creation of the Florida Department of Environmental Protection (FEAP) contained a mandate stating, "By January 1, 1995, DEP shall make recommendations to the Legislature for the establishment of a comprehensive program for the research into and control of exotic plants and animals that are invasive to public lands". In January of 1995, FDEP delivered a report edited by Schmitz and Brown (1994) to the Legislature that was designed to be the foundation upon which that comprehensive program is to be built.

A total of 60 individuals helped author this report. Their professional backgrounds ranged from employees of state, federal, and county government agencies and university professors, to staff members of non-profit organizations like The Nature Conservancy. None of these individuals were paid. They all volunteered their valuable time and their expertise, because most of them felt it was time that Florida should begin seriously to evaluate invasions of non-indigenous species that plague what remains of our state's natural areas. Other than a smaller, less comprehensive report on non-indigenous species invasions for the State of Hawaii (The Nature Conservancy and Natural Resources Defense Council, 1992), this was the first time an entire state has evaluated the ecological consequences, management, and present governmental approaches to non-indigenous plants, insects, fishes, amphibians and reptiles, mammals, birds, and freshwater and marine invertebrates. The following text briefly summarizes the results of the FDEP report.

WHY IS FLORIDA BEING INVADED?

Non-indigenous, exotic, alien, introduced, and non-native species are all terms used to describe organisms not historically found in an area that did not arrive by natural means, and they persist, thrive, and may displace or harm native species. Within the United States, Hawaii appears to have the worst problems with biological invasions; Florida second, and California third. It is commonplace of invasion biology that two types of sites seem to be particularly likely to be devastated by non-native species: islands and disturbed landscapes. Florida fulfills both criteria. Much of the state has become a patchwork of habitats owing to human activities and remaining natural areas are fragmented remnants of once formerly large, contiguous ecosystems that stretched from coast to coasts and north to south. The southern third of Florida is essentially a habitat island, surrounded on three sides by water and on the fourth by frost. It is no coincidence that Hawaii and Florida have the worst problems with non-indigenous species invasions.

In addition, Florida's warm tropical climate and extensive network of human-made canals and waterways is believed to have helped facilitate the spread of non-native organisms throughout a good portion of the state. Combining Florida's physical characteristics with a large out-of-state tourist economy, large aquarium fish and exotic plant nursery industries (the majority of all live propagated plant material and aquarium fish are first imported into Florida for later distribution throughout the United States), it is not hard to conclude why Florida is being invaded.

In 1958, the late prominent ecologist, Charles Elton, described these biological invasions as ecological explosions (Elton, 1958). Elton stated, "I use the word 'explosion' deliberately because it means the bursting out from control of forces that were previously held in restraint by other means." Although these invasions were recognized as a serious environmental problem years ago, past research efforts regarding biological invasions throughout the world have been sparse. Much of the available information regarding the ecological impact of the non-native plants and animals in Florida is anecdotal.

PLANTS

Successful invading plant species in Florida generally have traits such as high population growth rates, short life cycles, high allocation of resources to reproduction, good dispersal mechanisms, and a flexible use of a variety of habitats. Many of these invading plant species once had, or still have, traits considered desirable to humans. For example, 39% of the Exotic Pest Plant Council's 1993 list of most invasive species were still commercially available for sale and in continual spread in 1994.

It has been suggested that the floating South American water lettuce (Pistia stratiotes) was probably the first introduction of an invasive non-indigenous plant species into Florida, and maybe even into North America (Schmitz, et al., 1993). Its establishment in Florida has been linked to the arrival of the early Spanish settlers in the Old City of St. Augustine.

Another floating South American invader, waterhyacinth (Eichhornia crassipes), arrived much later, sometime in the 1880's. During the 1950's, it covered more than 48,500 hectares of Florida's waterways. Because it has been managed and controlled for over 100 years in Florida, because of its interference with navigation, waterhyacinth is one of the few invasive species for what there is published scientific evidence about its ecological impact. Waterhyacinth's ability to form complete cover, or an aquatic plant canopy, over a waterbody can lead to low dissolved oxygen levels, higher water temperatures, greater water loss through evapotranspiration, higher sediment loading, lower fish production in infected waterbodies, smothered beds of native submersed vegetation, and destroyed uprooted native emergent plants (Schmitz, et al., 1993). Because of a statewide coordinated approach by a lead agency and the management philosophy of maintenance control (maintaining widespread invasive alien plant species at their lowest practical population levels), waterhyacinth no longer disrupts Florida's waterways (only 680 hectares in 1993).

Waterhyacinth, considered by some to be the world's worst weed, has been superseded in Florida as the most invasive non-native aquatic plant species. Hydrilla (Hydrilla verticillata), introduced in the early 1950's, spread rapidly throughout Florida by fragments of the plant attached to boats and their trailers. It now occupies more than 39,000 hectares of Florida's waterways (Table 1).

Table 1. Estimated hectares of Florida's most widespread non-indigenous plant species in public lands and waterways (Source: Schmitz and Brown, 1994). Note: One hectare infested can vary from one individual plant to densely-packed monospecific stands.

SPECIES	COMMON NAME	YEAR SURVEYED	HECTARES
Causarina spp.	Australian Pine	1993	151,246
Eichhornia crassipes	Waterhyacinth	1993	680
Hydrilla verticillata	Hydrilla	1995	39,485*
Lygodium microphyllum	Climbing Fern	1993	10,434
Melaleuca quinquenervia	Melaleuca	1993	197,827
Panicum repens	Torpedo Grass	1992	7,100
Schinus terebinthifolius	Brazilian Pepper	1993	284,708

*Estimate provided by J. Schardt, FDEP, April, 1995.

A lack of state funding to manage hydrilla adequately is linked to its incredible spread in the last few years. Because hydrilla interferes with navigation, there have been numerous published studies on how this species effects Florida's waterbodies. Like waterhyacinth, its key environmental impact is its ability to produce a dense mat, or canopy, at the water's surface. Consequently, existing bottom inhabitants are shaded out and water chemistry can be substantially altered. For example, when hydrilla biomass peaked during the fall in a small eutrophic Central Florida lake, dissolved oxygen concentrations measured at the bottom were generally less than 2.0 ppm and probably related to reduced underwater light penetration and water circulation (Schmitz and Osborne, 1984).

Another widespread introduced species is the Australian melaleuca tree (Melaleuca quinquenervia). Widely planted for ornamental landscape purposes and as fence rows in Southern Florida since the early part of this century, it has spread throughout South Florida's wetlands and vast areas of the remaining Everglades. Melaleuca forests have been linked to changes in normal fire ecology regimes, a total displacement of native vegetation, possibly higher evaportranpiration rates, the creation of "artificial" tree islands, and is generally considered to be poor habitat for wildlife.

Recently, there has been controversy about the wildlife value of melaleuca forests in South Florida. An ongoing study contracted by Dade City (Everglades Research Group, 1995) found 89 species and more than 1,700 individuals during a one-year period in a wetland melaleuca forest with greater than 75% coverage. Based on their data, it would appear that wetlands invaded by melaleuca contain a relatively high diversity and abundance of wildlife. A closer examination of this unpublished data reveals:

1. The number of species of amphibians, reptiles, birds, and mammals found within this wetland melaleuca forest were almost half wetland independent species, or species more typical of upland environments.

2. The number of non-native wildlife species found within this dense melaleuca forest comprised about 30% of the total individuals collected (as opposed to <4% non-natives found in adjacent uninvaded systems).

3. Water levels were up and unusually high within this wetland melaleuca forest. Most of the increase in numbers of species in this dense wetland melaleuca forest consisted of aquatic species (fish) simply swimming into this cover type. As water levels recede, it is believed that these species will either swim back out or will perish (Everglades Research Group, 1995).

4. Finally, the number of species found within this particular melaleuca forest was not a good measure of the general ecological impacts of these monospecific stands because of an adjoining 10,500-acre, uninvaded sawgrass community that most likely was the source of many of these species.

The major ecological problem with areas invaded with melaleuca appears to be the lack of a usable forage source for herbivores. These is little evidence of any grazing of melaleuca trees by either native or introduced herbivores. The almost total displacement of native understory by mature melaleuca forests only compounds the problem.

Brazilian pepper (Schinus terebinthifolius) in Florida's most widespread invasive non-indigenous plant species and occupies more than 288,000 hectares (Table 1). Although primarily an invader of disturbed habitats, this species has invaded and formed large, dense forests in undisturbed areas adjacent to mangroves along the southwestern portion of the Everglades National Park. Environmental changes linked to Brazilian pepper invasion include the almost total displacement of native understory, negative impacts on native bird communities, and possible interference with natural mangrove detritus ecology.

Other widespread non-indigenous plant species found in Florida include climbing fern (Lygodium microphyllum) and torpedo grass (Panicum repens) (Table 1).

INSECTS

Entomologists classify all species of foreign origin as immigrant species. They do not distinguish between insect species that arrived on their own volition, or those that arrived with human help. Approximately 1,000 immigrant insect species are established in Florida. These immigrants cause millions of dollars of damage to agriculture, horticulture, and the human-made structures in Florida each year. Many are new arrivals. For example, the citrus leafminer (Phyllocnistis citrella) may have hitchhiked a ride on plants imported to replace those destroyed by hurricane Andrew. Other new arrivals may eventually directly impact public health. The Asian tiger mosquito (Aedes albopictus), first found in a scrap tire dump in Jacksonville in 1986, has spread throughout most of Florida. This species is a major vector of several human diseases. African honeybee (Apis mellifera scutellata) arrival is expected to result in rare, but horrible, stinging deaths to both animals and humans.

The ecosystem impact of these new insect arrivals is poorly known. Fire ants (Solenopsis invicta) have invaded all of Florida. However, most published information regarding their impact is on agroecosystems. Sometimes members of one taxonomic group of non-native organisms increase the impact of another group of non-indigenous species. For example, more than 60 non-native ficus tree species are used for landscaping purposes in Southern Florida. For years, they could not set viable seed because each species is pollinated only by its own species of agonoid wasps. Three of their species of wasps have appeared in Florida during the last two decades. These ficus species are now setting viable seed and seedlings can be found in Dade County's tropical hardwood hammocks.

FISH

There are approximately 28 non-indigenous fish species, permanently or temporarily established, in the open waters of Florida. None of these non-indigenous fish species are linked to extinction of native species. However, this does not mean they constitute no threat to native taxa. Many of the non-indigenous fish species are predators and others are known to compete with native fishes for spawning sites and similar resources. For example, Blue talapia (Oreochromis aureus) originally imported as a potential aquatic weed control agent in the 1960's, has become dominant in some Central Florida lakes and rivers and may be impacting native fish spawning sites. The Mayan cichlid (Cichlasoma urophthalmus), recently found in South Florida, is a voracious predator and may prove to be one of the most detrimental introductions to date. It can tolerate salinities up to 40 ppt. Conversely, some fish introductions have not had the dire ecological impacts that once were predicted. The walking catfish (Clarias batrachus) is a good example.

AMPHIBIANS AND REPTILES

There are approximately 27 species of non-indigenous amphibians and reptiles established in Florida. The majority of them are considered to have little consequence, as either competitors, predators, or pests for native biota, humans, and domestic pets. If there are consequences, they may fall into four categories: competition, predation on native species, unpalatablity to native predators, and possible hybridization between closely-related native and non-indigenous species.

It has been suggested that non-indigenous species may alter complex food webs. For example, Cuban tree frogs (Osteopilus septentrionalis), giant toads (Bufo marinus), tokay geckos (Gekko gecko), and boa constrictors (Constrictor), all established in Florida, prey upon native species and may ultimately impact the number of prey species available to native predators. On the other hand, it has also been suggested that established and long-lived boa constrictors could actually benefit native reptiles and amphibians by replacing missing top carnivores within Florida's environment, thereby controlling mid-sized mammalian predators like house cats, opossum, and raccoons, all of which feed upon native herptofauna.

MAMMALS AND BIRDS

At least 10 non-indigenous mammal species are known, or suspected, of having established populations in Florida. Like amphibians and reptiles, little is known about their overall ecological impacts. Consequences, such as habitat destruction by the rooting behavior of feral hogs (Sus scrofa), have been reported by land managers throughout Florida. This creates ideal conditions for non-native plants to establish and flourish. Other impacts by non-indigenous mammal species ranged from predation on native species, vectoring of diseases and parasites that affect native mammals (as well as humans and domestic animals), possible increased hybridization between closely-related native and introduced species, and possible competition for food or other resources by non-native species.

Eleven non-indigenous bird species have established breeding populations in Florida. Many of these established species are escapes from the pet trade. No known ecological impacts have been documented. However, possible harm to native bird populations could occur as a result of diseases introduced by non-indigenous species.

FRESHWATER AND MARINE INVERTEBRATES

Presently, only 2% of Florida's 1,200 aquatic freshwater invertebrates are considered to be of foreign origin. Probably the best known non-indigenous freshwater invertebrate found in Florida's waterways is the Asian clam (Corbicula fluminea). It was introduced during the 1920's by oriental laborers, possibly for human consumption. It has a reproductive advantage over native mussels because it does not require a fish host to support its veliger stage. The species has detrimentally altered stream bottom habitat and reduced ambient phytoplankton populations.

Although not yet found in Florida's waterways, the zebra mussel (Dreissena polymorpha) is believed to pose a threat to panhandle waterways. Like the Asian clam, zebra mussels do not require a fish host to ensure survival of the veliger stage. Prolific spread of this species in the last ten years since its introduction into the United States and its associated high bottom densities, has caused concern that severe ecological and economic impacts could result if zebra mussels become established in North Florida.

The number of marine invertebrate species found in Florida's coastal waters is not known. It is difficult to tell what ecological impact, if any, they have. Many of these non-indigenous species are believed to be cryptic; that is, introductions were made as fouling organisms on ship bottoms, commencing in the early 16th century by early Spanish colonization attempts in the Florida region. Since there were no available species lists at the time of these colonization attempts, it is difficult to ascertain what is native or non-native.

SUMMATION OF ECOSYSTEM IMPACTS AND A LOOK TO THE FUTURE

Although most of what we know is based on anecdotal evidence, it appears that Florida's remaining natural areas are changing from naturally-evolved ecosystems to systems that contain dense monospecific stands of non-indigenous vegetation. Plants form the foundation of ecosystems. The displacement of native diverse vegetation into monospecific non-indigenous landscape is detrimental to maintaining diverse assemblages of native wildlife. Other effects of introduced wildlife are unclear. However, anecdotal evidence suggests a good cause for concern.

What does the future hold? Other established non-indigenous plant species such as tropical soda apple (Solanum viarum), cogan grass (Imperata cylindrica), carrotwood (Cupaniopsis anacardioides), Chinese tallow (Sapium sebiferum), and catclaw mimosa (Mimosa pigra) may eventually become the new "melaleucas" in Florida. For example, recent research indicates the Everglades wetland system may be particularly vulnerable to catclaw mimosa invasion because of this plant's ability to grow in soils with poor fertility levels that are similar to those soils found in the Everglades. Catclaw mimosa is presently found in five small distinct locations in South Florida.

Among non-indigenous animal species, the mildly-venomous brown tree snake (Boiga irregularis), a native of the Solomon Islands, Papua New Guinea (and Australia), has devastated the ecology, economy, and quality of life on the Island of Guam. The species has caused power outages, extinctions of native bird and lizard species, and invasions of homes and hotels (where snakes search for food), often biting sleeping victims. Densities on Guam have ranged from 25,900 to 64,750 per square kilometer. When live prey is not available, this snake has been observed feeding on human garbage. If this snake became established and produced high population densities in Florida (the species does well in disturbed, hot, humid climates, especially those with high rainfall), it could have significant economic impact of Florida's tourism industry, as well as native wildlife populations. Obviously, a great many tourists would not travel to Florida, avoiding possible encounters with this snake.

In conclusion, the invasion of non-indigenous species in Florida's public lands is a major conservation problem, perhaps second only to habitat destruction and loss. Although much of the available information is anecdotal and extensive research is needed, what we already know should be cause for great concern.

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