Proceedings of the twentieth Annual Conference on Wetlands Restoration and Creation, May 1993. 1993

Tampa, Florida: Hillsborough Community College, 1993

https://digital.library.wisc.edu/1711.dl/W5TE47WWIK7JR8X

Proceedings of The 20th Annual Conference on Wetlands Restoration and Creation

May, 1993

Sponsored by
Hillsborough Community College
Institute of Florida Studies

Hillsborough Community College

Proceedings of
The Twentieth Annual Conference on Wetlands Restoration and Creation

May, 1993

Sponsored by
Hillsborough Community College
Institute of Florida Studies

Frederick J. Webb, Jr.
Editor

HILLSBOROUGH COMMUNITY COLLEGE
1206 North Park Road
Plant City, Florida 33566

This publication should be cited as:

Webb, F. J., Jr., Editor. 1993 Proceedings of the Twentieth Annual Conference on Wetlands Restoration and Creation. Hillsborough Community College, Tampa, Florida.

Table of Contents

TABLE OF CONTENTS
Continued

Introduction

The Annual Conference on Wetlands 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. The conference is designed to be of particular benefit to governmental agencies, planning organizations, colleges and universities, corporations, and environmental groups with an interest in wetlands. These proceedings are a compilation of papers and addresses presented at the Twentieth Annual Conference.

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 acknowledgement for contributing to the conference and assisting in the preparation of the proceedings for publication: Elaine Baskin, Sanjeev Choudhry, Lydia Dehoyos, Donna Foley Janet Giles, Charles Mason, 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 staff of **LEWIS ENVIRONMENTAL SERVICES** and the **FLORIDA DEPARTMENT OF ENVIRONMENTAL REGULATION** for arranging and conducting very successful field trips to wetland restoration sites.

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

To all these people, thank you.

INNOVATIVE USE OF DREDGED MATERIAL IN TEXAS BENEFITS ENDANGERED WHOOPING CRANES

Charles E. Belaire
BELAIRE CONSULTING, INC.
P.O. Box 741
Rockport, Texas 78381

David Templet
Manager, Environmental Engineering
MITCHELL ENERGY CORPORATION
P.O. Box 4000
The Woodlands, Texas 77387-4000

ABSTRACT

During 1991 and 1992, Mitchell Energy Corporation, The Woodlands, Texas, constructed a whooping crane habitat from dredged material at the Aransas National Refuge. Belaire Consulting, Inc., Rockport, Texas, performed design, construction management, planting, and monitoring efforts for the project. The habitat was designed in cooperation with federal and state resource agencies in an effort to restore lost habitat, to protect existing habitat from erosion, to serve as a demonstration project, and to provide a beneficial use for dredged material. Approximately 65,000 cubic yards of hydraulically-dredged sand and silt were pumped to a 13.5-acre leveed area. The elevation of the discharged material was controlled to duplicate elevations in coastal marshes that are heavily utilized by the endangered whooping crane (Grus americana). Approximately 75% of the site was stabilized at elevations suitable for Spartina alterniflora and the remaining 25% of elevations were established to support intermediate-to-high marsh species. Prior to planting, the exposed levees surrounding the fill area were covered with interlocking concrete mats. Other erosion control measures, including floating wave barriers, erosion blankets, and concrete sacks were also installed. Fill areas within the project's levees were sprigged in September, 1991, and March-April, 1992. Survival of the planted species exceeded 90% sixty days after planting. Coverage of Spartina alterniflora and mixed marsh planting areas exceeded 80% and 90%, respectively, one year after planting. This project was a successful demonstration that dredged can be used to create whooping crane habitat.

INTRODUCTION

In 1990 Mitchell Energy Corporation (MEC), The Woodlands, Texas, decided to re-enter an oil and gas field in Mesquite Bay, Texas. Water depths in the bay and within previously dredged channels were insufficient to allow drilling rig access to existing wells. To obtain access to the field, MEC was required to maintenance dredge an existing channel and drilling basin.

Prior to commencement of dredging, MEC sought and obtained authorizations from the U.S. Army Corps of Engineers, Galveston District, and the Texas General Land Office. During the process of obtaining these authorizations, MEC evaluated more than ten disposal site options. After detailed environmental assessments, in-depth discussions with the U.S. Fish and Wildlife Service, National Marine Fisheries Service, Environmental Protection Agency, Texas Parks and Wildlife Department, and personnel of the Aransas National Wildlife Refuge (ANWR), and economic and engineering evaluations, all parties involved agreed that the preferred means of dredged material disposal was the construction of a whooping crane habitat from dredged material placed adjacent to the ANWR.

This 13.36-hectare (33-acre) island disposal site/whooping crane habitat was authorized at a location which would provide a variety of direct and indirect benefits to the endangered whooping crane and to the local bay ecology. The project would: (1) offer erosion protection to Bludworth Island, a severely eroded island which protects high-use whooping crane feeding areas from open bay wave action; (2) restores salt marshes which had been lost to erosion; (3) cause minimal impact to high energy shallow bay bottom; and (4) provide a demonstration project for the beneficial use of dredged material. Federal and state authorizations for this project were granted in April, 1991.

In late spring, 1991, MEC began construction of a 5.46-ha (13.5-acre) first phase of the demonstration project. Dredging and disposal for this first phase was completed in late summer, 1991. Planting of the 5.46 ha (13.5 acres) was completed in April, 1992.

Other salt marsh habitat has been created elsewhere in the United States. The U.S. Army Corps of Engineers (Webb, et al., 1974) established salt marsh as well as upland vegetation on dredged material placed in Galveston Bay, Texas. The Corps has also constructed approximately 15 acres of salt marsh habitat from dredged material in the Chesapeake Bay system (Earhart and Garbisch, 1983). The Port of San Diego used dredged material to construct approximately 60 acres of subtidal and intertidal habitats (Andrecht and Firle, 1987). The MEC project described herein appears to be unique in that the dredged material was used to construct habitat for a specific endangered species.

Government authorizations required that MEC's constructed habitat be monitored for transplant survival sixty days after initial planting and for vegetation coverage six months, one year, two years, and three years after initial planting. The results of the survival and coverage surveys, as well as the relative effectiveness of various erosion-control measures are presented in this report.

STUDY AREA AND SITE

Mesquite Bay is a primary bay situated between San Antonio and Aransas Bays on the Central Texas Coast. Mesquite Bay is connected to the Gulf of Mexico by Cedar Bayou, a comparatively small Gulf pass which separates Matagorda Island from San Jose Island. Mesquite Bay lies between the ANWR and the Gulf of Mexico (Figure 1).

The ANWR is an approximately 22,258.5-ha (55,000-acre) refuge established in 1937 for the primary purpose of protecting and enhancing the population of the endangered whooping crane. The wintering crane population at the ANWR has grown from fewer than 20 individuals in the early 1940's to approximately 140 individuals in recent years. Crane habitat at the refuge has decreased over the years in large part due to erosion along the Gulf Intracoastal Waterway (GIWW) (Stehn, 1991). Volunteer efforts have helped stabilize some of the GIWW shoreline through the placement of concrete sacks on the edges of marshes which border the GIWW. The Corps of Engineers has announced plans to stabilize additional areas of this shoreline by installation of concrete erosion control mats.

Bludworth Island lies between the GIWW and Mesquite Bay. The source of dredged material for this habitat creation project was the Wynne Channel and a drilling basin adjoining that channel. The Wynne Channel lies at the northeast end of Bludworth Island (Figure 1). The 5.46-ha (13.5-acre) habitat was constructed in Mesquite Bay within approximately 60.94 m (200-ft) of the Bludworth Island shoreline (Figure 2). The remaining 7.89-ha (19.5 acres) of the habitat is authorized for construction to the east-northeast of the 5.46-ha (13.5-acre) site. This additional acreage would adjoin, or be placed in close proximity to, the first phase of the project.

Bay bottom elevations at the location of the 5.46-ha (13.5-acre) habitat typically ranged from approximately -15.23 cm (-0.5 ft) MLT to +15.24 cm (+0.5 ft) MLT (Corps of Engineers datum). Normal tide level in the project vicinity is approximately 67.06 cm (+2.2 ft) MLT. Daily tidal fluctuations are generally 6.10 cm (0.2 ft) to 9.14 cm (0.3 ft).

MATERIALS AND METHODS

The demonstration project described in this report was funded by MEC. Belaire Consulting, Inc. (BCI) assisted MEC with permit acquisition, data collection, environmental assessments, site selection, project design, construction management, planting and monitoring. Kingfisher Marine Services, Port Lavaca, Texas, served as the general contractor. Design criteria, construction methods, and monitoring techniques were developed through extensive coordination among MEC, Belaire Consulting, Inc., the Corps of Engineers, Texas General Land Office, U.S. Fish and Wildlife Service, National Marine Fisheries Service, Environmental Protection Agency, and the Texas Parks and Wildlife Department. The materials and methods employed during the projects are described below.

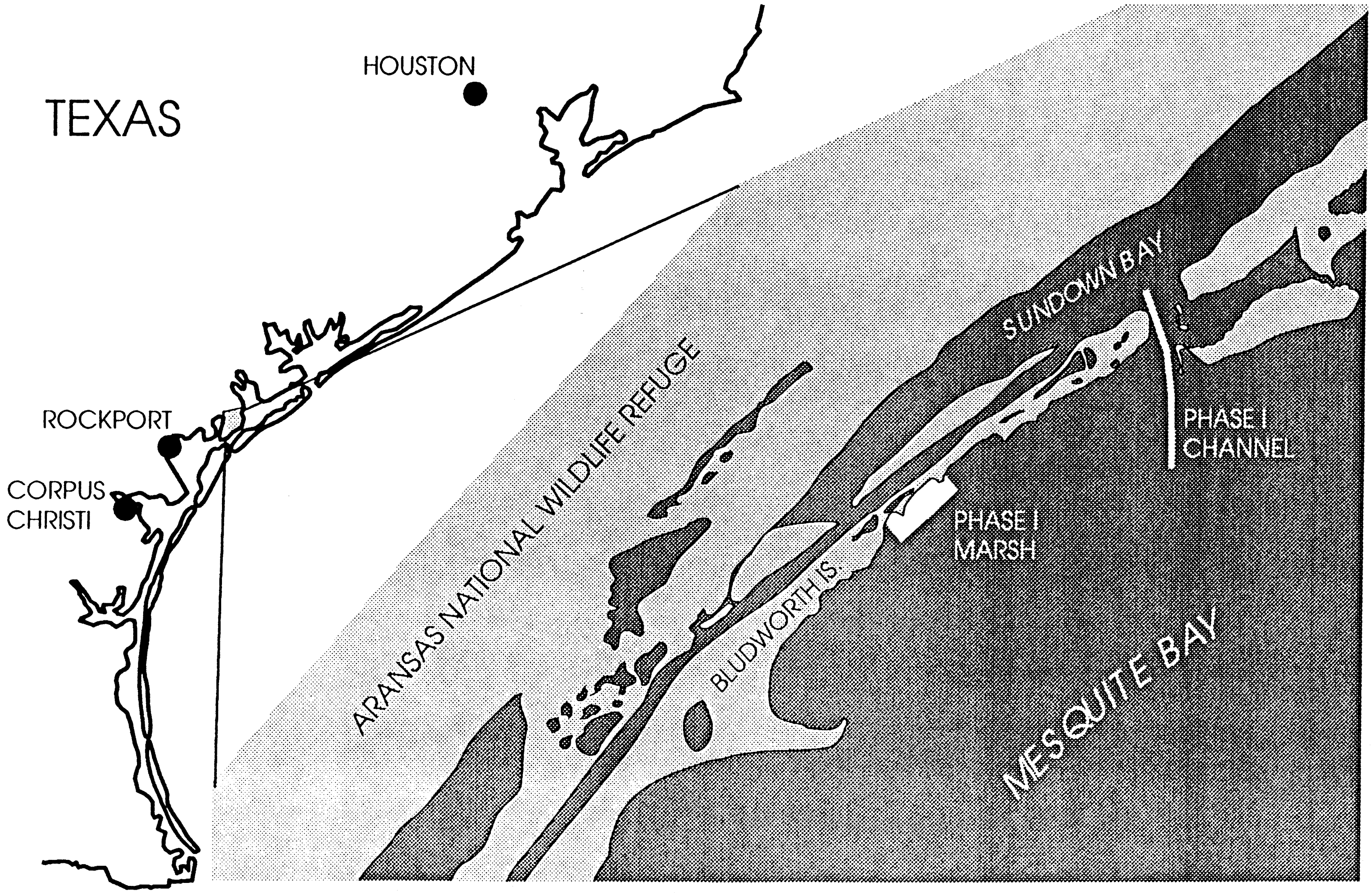
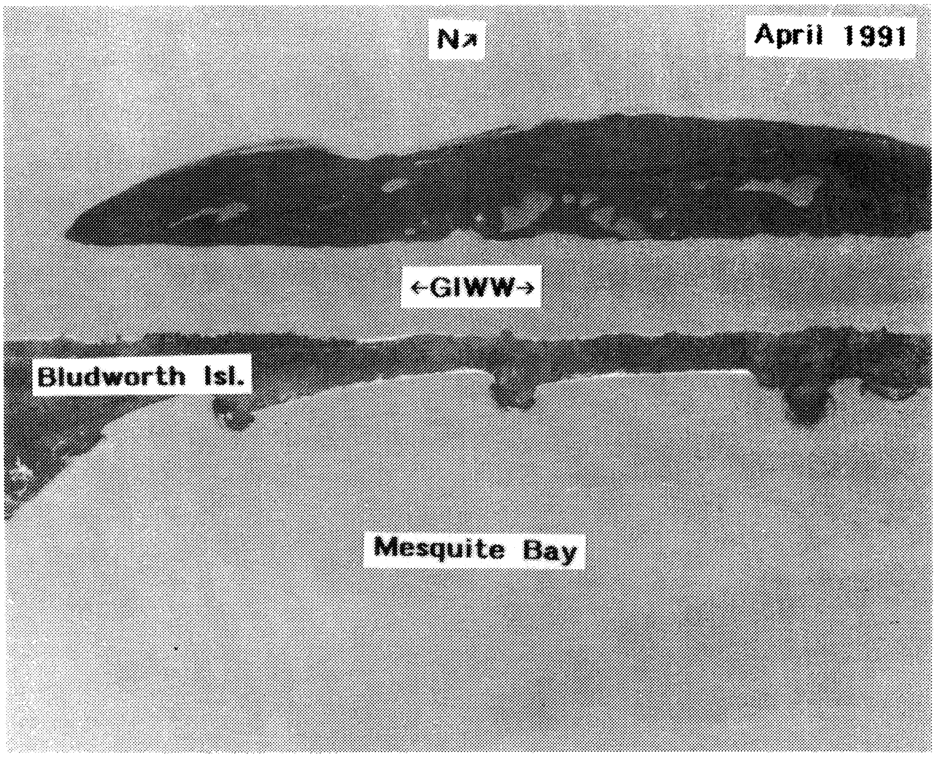


Figure 1. Location map.



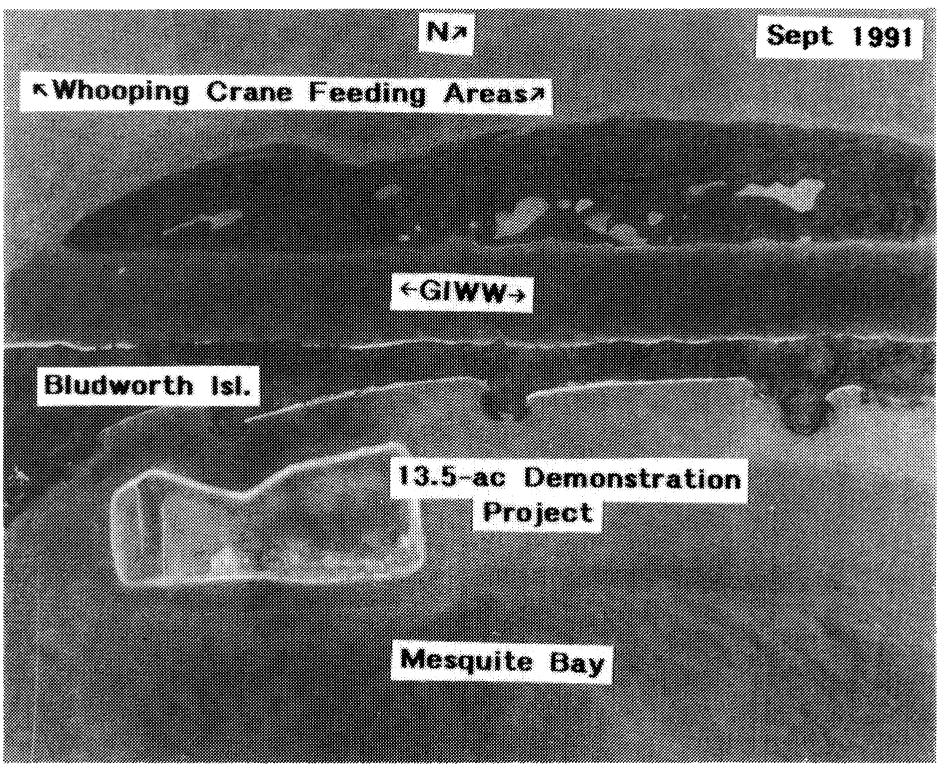


Figure 2. Aerial photographs of pre- and post-construction study site.

Design and Construction

Levees. Levee construction was performed in June, 1991. Levees were constructed form bay bottom materials within the perimeter of the 5.46-ha (13.5-acre) site. Predominantly sand material was available for the landward and side levees. Sandy silt material was the primary material used for constructing the seaward levee. The levees were built to a peak of 1.83 m (+6 ft) MLT for the dredging phase of construction, then lowered to 1.22 m (+4 ft) MLT after dredging was completed. This 1.22 m (+4 ft) MLT elevation is the maximum levee height permitted by the ANWR. Levee slopes varied form 4:1 to 7:1 depending upon the material type.

Dredge and Fill. Hydraulic dredging began in late June, 1991, and was completed in mid-July, 1991. Sandy material from the 609.34 ha (2,000 ft) of Wynne Channel nearest the GIWW was dredged first. This sandy material was pumped immediately behind the project's seaward levee to provide for stability and long-term resistance to wave erosion forces. Silty material from the remainder of the Wynne Channel and the drilling basin was discharged into the interior of the site.

To determine desirable fill elevations for this demonstration project, BCI worked with the ANWR staff to identify a high-use whooping crane feeding habitat in the refuge. BCI estimated elevations in this model marsh area and used these data in selecting elevations within the constructed habitat. A Corps of Engineers' tide staff at GIWW marker No. 7 was used as a benchmark to estimate all project elevations.

The maximum fill elevation permitted by the ANWR was 1.22 m (+4 ft) MLT. Elevations above 1.22 m (+4 ft) MLT would allow the establishment of shrubs and other vegetation types which might attract coyotes and other whooping crane predators. Approximately 75% of the site was filled to between 30.48 cm (+1.0 ft) MLT and 76.2 cm (+2.5 ft) MLT, elevations suitable for transplanting Spartina alterniflora. The remaining 25% of the site was filled to between elevations 76.2 cm (+2.5 ft) MLT and 1.22 m (+4 ft) MLT. These elevations are suitable for transplanting a variety of higher salt marsh species.

Erosion Control. Several erosion control measures were employed to protect the constructed habitat from wave action. On the seaward and side levees, concrete mats were installed from the top of the levee to the toe of the levee slope. The levee was graded to a 3:1 slope prior to placement of the mats (Figure 3). The 2.44 m (8 ft) x 4.87 m (16 ft) mats were comprised of interlocking concrete blocks interconnected by non-corrosive metal cable. Prior to placing the mats, the levee slope was covered with manufactured filter fabric. Each mat was placed on the filter fabric then interlocked to the two mats to either side. All mats were anchored at the top of the levee with 1.22 m (4 ft) screw-type metal anchors. Mats were installed in August, 1991.

To stabilize the soil adjacent to the armored levees, erosion blankets were installed on a 4.57 m (15 ft) wide strip immediately behind the top of the concrete mats (Figure 3). The 1.22 m (4 ft) wide x 30.46 m (100 ft) long erosion blankets were comprised of a 2.54 cm (1-inch) thick layer of biodegradable wood shavings bound by 1-inch mesh photodegradable plastic, were held in place by 18-inch long metal staples and 4-feet long wooden survey stakes placed at approximately .914 m (3 ft) centers. The erosion blankets were installed in September, 1991.

To break wave action which overtopped the levee armor, concrete sacks were installed at the top of the concrete mats (Figure 3). The sacks were placed one, two, or three high, depending upon the wave exposure at the point of installation. The sacks were secured by .318-cm (3/8-inch) steel, rebar driven vertically through the bags. After installation, the bags were watered and the concrete allowed to harden. The concrete sacks were installed in March, 1992.

From July, 1991, through April, 1993, a floating wave barrier was maintained approximately 30.46 m (100 ft) - 45.70 m (150 ft) off the seaward levee and the northerly side levee. The wave barrier was made of floating 30.48-cm (12-inch) diameter pipe fastened to piling clusters. The purpose of this barrier was to dampen waves prior to their impacting the armored levee.

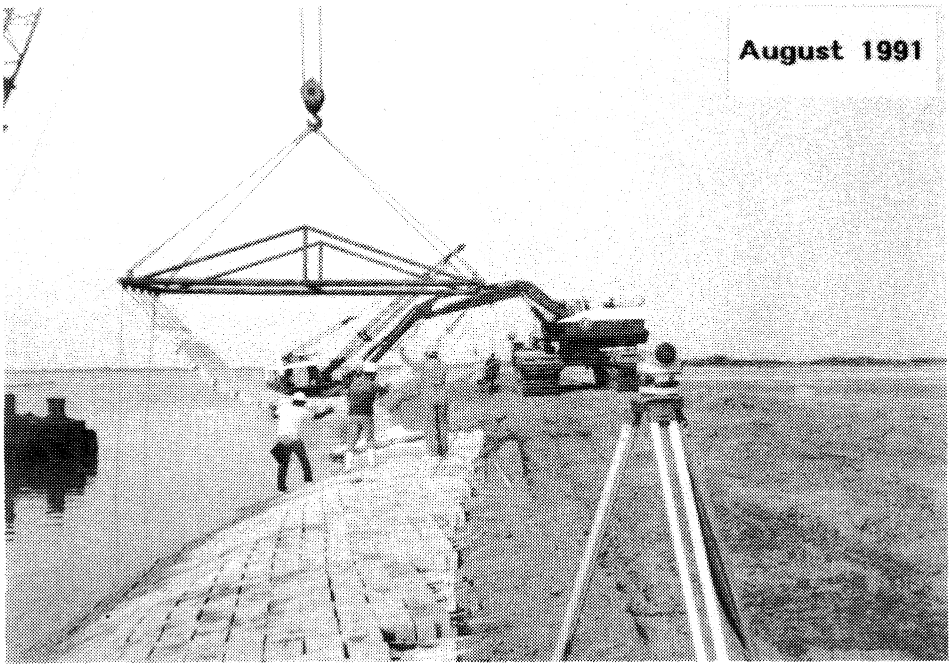


Figure 3. Erosion control measures.

Circulation. After dredging was completed, temporary tide exchange pipes were placed in the landward levee. These pipes were equipped with dropgates on the ends of the pipes that were within the perimeter of the levee. The gates were adjustable to allow control of water levels from approximately 30.48 cm (+1 ft) MLT to 115.82 cm (+3.8 ft) MLT. After crusting of the fill material was completed in March, 1992, the tide exchange pipes were removed. Two circulation channels with 30.48 cm (12 ft) bottom widths were constructed at the locations of the tide exchange pipes to provide for long-term water exchange.

Dewatering/Crusting. Between July, 1991, and March, 1992, water was removed to allow consolidation of the hydraulically-dredged material. Tide exchange gates were adjusted as necessary to allow water within the 5.46-ha (13.5-acre) area to exit the site. The gates were also adjusted as needed to block intrusion of tidal waters. Water was pumped from the site during low tide periods.

Transplanting

Approximately .61-ha (1.5 acres) of mixed marsh was planted behind the armored levee in September, 1991. Approximately 2.02 ha (5.0 acres) of Spartina alterniflora and 1.42 ha (3.5 acres) of mixed marsh were planted in March and April, 1992. Approximately 1.42-ha (3.5 acres) of the 5.46-ha (13.5-acre) site were left as open water. Open water areas were typically at elevations of 30.48 cm (+1.0 ft) MLT.

Mixed marsh species included Batis maritima, Borrichia frutescens, Distichlis spicata, Eustoma exaltatum, Lycium carolinianum, Paspalum vaginatum, Salicornia virginica, Sesuvium portulacastrum, Spartina patens, Sporobolus virginicus, and Sueda linearis. Spartina patens and Paspalum vaginatum were predominantly planted in close proximity (within 15.24-22.85 m [50-75 ft]) to the armored levee. These species were selected for placement in areas subject to wave action because they were the two mixed marsh species, which demonstrated an ability to withstand erosive forces along the banks of the GIWW and on the shoreline of Bludworth Island. The other mixed marsh species were selected because they tend to dominate vegetation coverage in the whooping crane feeding areas.

Transplants were taken from natural marshes along the Mesquite Bay shoreline of Bludworth Island and from the banks of the nearby GIWW. State of Texas General Land Office guidelines were followed. No more than one 6-inch diameter plug was taken from .84 m^2 (9 ft^2) of marsh. Upon removal of a plug, the soil was washed from the roots of plants and the plants were submerged immediately in containers filled with bay water. Plant material was then transported to the project site and transplanted within 24 hrs.

One sprig or plug of marsh vegetation was planted on .914-m (3-ft) centers within the planting area. Sprigs or plugs were placed in holes made with hoes, shovels, or other hand tools. Soil was then replaced and compacted around the transplants. Emergent transplants were irrigated with bay water immediately after planting and at a minimum of twice weekly for six weeks after initial planting. If significant rainfall events occurred, no irrigation was provided.

Emergent transplants were fertilized quarterly during the nine-month period following planting. A "13-13-13 all purpose" fertilizer was utilized.

Monitoring

Transplants were observed sixty days, six months, and one year after initial planting. The sixty-day monitoring effort evaluated transplant survival. The six-month and one-year monitoring estimated vegetation coverage. Methods utilized for vegetation monitoring conform with those routinely accepted by the State of Texas General Land Office.

Transplant survival was estimated by sampling within quadrats systematically spaced throughout the planted area. The quadrats were 6.1 m (20 ft) x 6.1 m (20 ft) squares. The number of live plugs or distinct sprigs were counted within each quadrat. The species of vegetation were noted and recorded. Percent survival was calculated by dividing the total number of plugs originally planted into the number of plugs counted during the monitoring survey.

Vegetation coverage was estimated by sampling 20 transects systematically spaced throughout the planted area. Each transect completely traversed the planted portion of the habitat. At 3.04-m (10-ft) intervals along each transect, three areas, each 15.24 cm (6-in) in diameter, were observed. One 15.24-cm (6-inch) diameter observation area was on the transect line, the second was 45.72 cm (1.5 ft) to the right of the line and the third was located 45.72 cm (1.5 ft) to the left of the transect line. The presence or absence of vegetation and the species present were noted and recorded for each 15.24-cm (6-inch) observation area. Coverage was estimated by dividing the total number of 6-inch observation areas into the number of observation areas which contained vegetation.

Erosion control measures and other constructed components of the project were inspected at a minimum of once a month from September, 1991 through April, 1993. Any deficiencies or needed repairs were noted and corrected as warranted.

RESULTS

Construction

Dredge and fill activities resulted in elevations which seemed appropriate for the establishment of the vegetation and open-water acreages called for in the project plans. The levee armor, through April, 1993, has shown no signs of failure. The erosion blankets provided some resistance to erosion, but soil loss up to one foot was experienced in some areas. The erosion blankets performed best when accompanied by the concrete sacks. The concrete sacks placed upon the top of the levee armor have broken,overtopping waves sufficiently to allow dense vegetation growth on adjacent ground. Only two of the approximately 2,000 concrete sacks were displaced by wave action as of April 1993. The floating pipe wave barrier performed as expected in dampening wave action from the open bay. This barrier's dampening effect reduced the force of waves upon the levees and adjacent ground. This floating barrier has, however, required periodic maintenance to repair leaks.

The circulation channels have shown no signs of siltation. Soundings of these channels in April, 1993 indicate that they were, in fact, larger than when originally constructed.

Vegetation Monitoring

Results of survival and coverage monitoring are summarized in Table 1. Survival was 98% for the Spartina alterniflora and 95% for the mixed marsh areas. Coverage for the Spartina alterniflora and mixed marshes were 99% and 83%, respectively, one year after planting. State of Texas General Land Office and U.S. Army Corps of Engineers' authorizations required 50% survival after 60 days and 70% coverage two years after planting.

Table 1. Vegetation survival and coverage data.

Marsh Type	60-Day Survival	6-Month Coverage	1-Year Coverage
Spartina alterniflora	(485/497) 98%	(233/311) 75%	(336/341) 99%
Mixed Marsh	(1115/1176) 95%	(222/466) 48%	(371/448) 83%

The species composition for the mixed marsh is summarized in Table 2. Only the data from the annual monitoring surveys are included in this table. Spartina patens, Paspalum vaginatum, Batis maritima, and Distichlis spicata are the most frequently occurring mixed marsh species. A total of 12 species of vegetation were observed during the annual monitoring surveys. Figure 4 shows photographs of the project during planting and one year after planting.

Crane Utilization

Since September, 1991 three sightings of whooping cranes have been reported to the ANWR (Stehn, 1993). A tour guide reported one sighting in the winter of 1992. An oilfield service company and Belaire Consulting, Inc. personnel each reported a sighting during the early spring of 1993. Numerous other species of birds, including herons, pelicans, terns, gulls, skimmers, ducks, and geese have been routinely observed utilizing the constructed habitat.

Table 2. Vegetation composition one year from initial transplanting.

Species	Frequency of Occurrence in Samples	Percent of Cover
Spartina alterniflora	336	48%
Batis maritima	64	9%
Borrichia frutescens	13	2%
Distichlis spicata	55	8%
Eustoma exaltatum	4	less than 1%
Lycium carolinianum	20	3%
Paspalum vaginatum	68	10%
Salicornia virginica	34	4%
Sesuvium portulacastrum	3	less than 1%
Spartina patens	85	12%
Sporobolus virginicus	24	3%
Sueda linearis	1	less than 1%

April 1993

Figure 4. Study site before and after transplanting.

CONCLUSIONS

Based on the evaluations described above, the 5.46-ha (13.5-acre) demonstration project at the ANWR can be termed as successful. The performance of the construction components, erosion control measures, transplanting efforts, and habitat utilization have generally met or exceeded expectations.

One modification which would decrease the likelihood and severity of erosion for similar future projects would be an increase in permanent levee height. An increase in height to +6 ft MLT at the crest of the levee with a steep drop to +4 ft MLT behind the levee would offer greater protection from wave forces and would provide minimal area for crane predator habitat.

The short-term goals of this project were definitely accomplished: erosion protection has been established for Bludworth Island; eroded wetlands have been restored in the project area; and a demonstration project has been constructed to form whooping crane habitat from dredged material. Only continued monitoring and perhaps maintenance of this habitat will determine its long-term success. This project has, and should continue to serve as, a reference point for planners of future coastal projects.

LITERATURE CITED

Andrecht, K. L., and T. E. Firle. 1987. Development of a coastal salt marsh in South San Diego Bay. Association of Environmental Professionals, Annual Conference. San Diego, California.

Earhart, H. G., and E. W. Garbisch, Jr. 1983. Habitat development utilizing dredged material at Barren Island Dorchester County, Maryland. Wetlands 3:108-119.

Stehn, T. 1993. Personal communication. Aransas National Wildlife Refuge. Austwell, Texas.

Webb, J. W., J. D. Dodd, B. W. Cain, W. R. Leavens, L. R. Hossner, C. Lindau, R. R. Stickney, and H. Williamson. 1978. Habitat development field investigations, Bolivar Peninsula marsh and upland habitat development site, Galveston Bay, Texas. Appendix D: Propagation of vascular plants and post-propagation monitoring of botanical, soil, aquatic biota, and wildlife resources. Technical report D-78-15. NTIS no: AD-AO63 781. Waterways Experiment Station, U.S. Army Corps of Engineers. Vicksburg, Mississippi. 521 pp.

CONSTRUCTED WETLAND POTENTIAL FOR PERFORMING WETLAND FUNCTIONS DEPENDENT ON ANAEROBIC SOIL CONDITIONS

R. B. Atkinson and J. Cairns, Jr.
Respectively, Assistant to the Director and Director
University Center for
Environmental and Hazardous Materials Studies
VIRGINIA POLYTECHNIC INSTITUTE AND STATE UNIVERSITY
Blacksburg, Virginia 24061-0415

P. R . Benzing
214 Biosciences Building
School of the Environment
DUKE UNIVERSITY
Durham, North Carolina 27708

ABSTRACT

Replacement of plant community structure in constructed wetlands has been a primary goal of past mitigation efforts; however, current goals include replacement of wetland functions and ecological services. By virtue of reducing soil conditions and position within landscapes, wetlands often improve water quality. Sites in this study were wetlands constructed as compensation for palustrine forested wetlands filled by highway construction in the coastal plane of Virginia. Soil respiration rate within laboratory microcosms, and constructed wetland redox potential, vegetation weighted average, soil organic matter, hydrology, and elevation were measured. Additional sites were surveyed in order to determine: (1) percent of constructed wetlands that are hydrologically connected to the impacted system; (2) prevalence of factors that could preclude functional performance over time, e.g., erosion; and (3) other site characteristics that could limit functional replacement. Microcosm studies conducted four years after construction indicated that reducing conditions are more readily established in microcosms with soil cores from reference sites than compensation sites. In situ redox potential data appeared to reflect both the vegetation weighted average (r^2 = 0.91) and the frequency and/or duration of saturation, since wetter areas exhibited lower redox potential (r^2 = -0.87). Potential for functional performance was adversely impacted by limited connection to surface hydrology in 69.2% of the sites, by potential siltation (61.5%), and by proximity to highways (46.2%). These results suggest that wetland functions and ecological services dependent on anaerobic conditions may not be performed by large portions of constructed wetlands.

INTRODUCTION

Wetlands were filled in the United States for many years with minimal regulation, and only a fraction of the original acreage remains (Dahl, 1990). Most wetland protection is currently afforded by Section 404 of the 1977 Clean Water Act, which requires a permit for deposition of fill material in a wetland. The National Environmental Policy Act (CFR Part 1508.20 [1-5]) allows permits to require substitution or replacement of lost wetlands with constructed systems when no alternatives are available. Mitigation by wetland construction is increasing in Virginia (A. Allen-Grimes, personal communication) and throughout the southeastern United States (R. Deitz, personal communication).

Many wetland mitigation projects have used replacement of similar vegetative composition as the primary goal. However, the phrase "in-kind mitigation" has evolved to include the replacement of functional, as well as structural, characteristics of the wetlands that were lost; concern over lost wetland functions is growing (e.g., Race and Christie, 1982; Quammen, 1988; Brinson and Lee, 1989).

Sather and Smith (1984) include water quality enhancement as one of five major values of wetlands. Brinson, et al. (1984) assessed nutrient assimilative capacity in an alluvial floodplain swamp and found considerable nitrate and ammonium removal. Gambrell and Patrick (1978) suggest that wetlands may accumulate or degrade organic pesticides and accumulate heavy metals. As wetland functions of value to society (or "ecological services") (Cairns, in press) become recognized, wetland protection and creation can be enhanced.

Hydrology and organic matter influence development of anaerobic soil conditions. Once flooded, diffusion of oxygen to sediment slows dramatically. The microbial community consumes organic matter and oxygen, thereby increasing carbon dioxide (and sometimes methane) concentrations and lowering oxidation-reduction (redox) conditions in soil (Pearsall, 1938; Ponnamperuma, 1972; Wharton and Brinson, 1979). Once anaerobic conditions become established, organic matter may accumulate and help maintain the reducing environment (Day, 1989). Soil organic matter content is also important since water quality enhancement functions dependent upon anaerobic conditions may not be performed if carbon availability is no sufficient (Faulkner and Richardson, 1989). Municipalities are increasingly constructing wetlands to treat wastewater (Hammer, 1989; Choate, et al., 1990). Watson, et al. (1989) found that anaerobic soil was very important in constructed wetlands designed for wastewater treatment and that vegetation was only responsible for about 10% of the nitrogen removal.

This study was conducted to assess the potential of a wetland constructed for mitigation to develop anaerobic soil conditions, which are necessary in order for wetlands to perform ecological services related to water quality.

SITE DESCRIPTIONS

Study sites included 13 previously constructed compensation sites in the coastal plane of Virginia. Detailed monitoring was conducted at four sites including Route 106 in Charles City County, Fourmile Creek--Route 295 in Henrico County, Powhite Parkway in Richmond, and Wagner Road in Petersburg. Reference wetlands were nontidal forested wetlands that were adjacent to each compensation wetland. All reference wetlands were dominated by obligate wetland species (species that occur in wetlands >99% of the time [Reed, 1988]). Uplands adjacent to each constructed wetland were selected based on the presence of mature hardwood forests with mixed composition and an undisturbed topsoil.

The Wagner Road constructed wetland was selected for further study. This constructed wetland site is located in Petersburg, Virginia, at 37_0 11'44" north latitude and 77_0 20'26" west longitude (Figure 1). The 1.53-ha study site is located in the upper coastal plane and was constructed in summer, 1987. Conditions at the

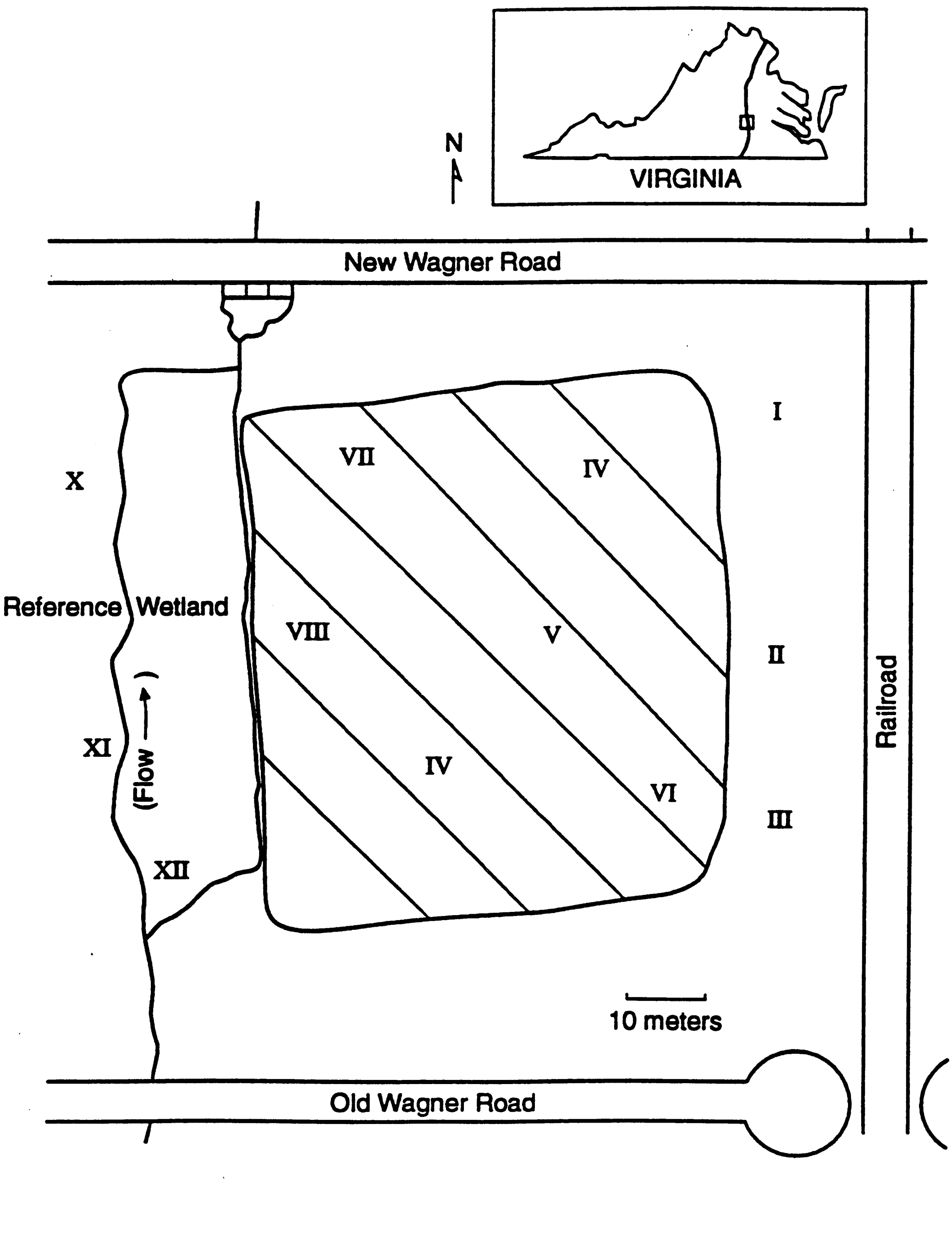


Figure 1. Location of the Wagner Road constructed wetland study site and reference wetland in Petersburg, Virginia. Roman numerals represent well locations.

site were representative of the 13 nontidal wetland compensation sites surveyed in the Virginia coastal plain. An adjacent, forested wetland in Second Swamp of the Blackwater Swamp headwaters was used as a reference site. A recently completed study of the hydrology and vegetation of the Wagner Road constructed wetland site indicated that hydrology was much more variable, both spatially and temporally, than the reference wetland (Atkinson, 1991). Vegetation in the constructed wetland site is dominated by colonizing herbaceous species (57) that range from obligate wetland species to obligate upland species. A middle-aged stand of Fraxinus pennsylvanica, Nyssa sylvatica, and Taxodium distichum characterizes the reference site vegetation, and species range from obligate wetland to facultative species.

METHODS

The study was composed of two laboratory and one field experiment. The Student-Newman-Keuls test for significance was used for statistical analyses (Zar, 1984). In addition, qualitative review of potential and ongoing erosion and siltation and hydrologic connectivity to adjacent streams and rivers was conducted by repeated site visits, review of mitigation site plans, and 1:24,000 scale 7.5-minute topographic quadrangle maps.

Experiment No. 1.

The effect of soil amendments on soil respiration in microcosms during short-term flooded conditions was quantified over a 3-week period to predict the effect of applying soil during compensation site construction. Nine soil cores (0.30 L) were taken from upland and wetland areas adjacent to each of the four compensation sites, including adjacent upland O horizons or topsoil (top 10 cm; 3 cores), adjacent upland A horizons or mineral soils (10-20 cm; 3 cores), and adjacent forested wetland O horizons (top 10 cm; 3 cores).

Microcosms consisting of 1-L Mason jars received a soil core and sufficient water to leave 0.30 L of gas head space. The jars were sealed with a rubber stopper fitted with two sampling ports. After 3 weeks, gas samples were taken and injected into a gas chromatograph fitted with a thermal conductivity detector and a Supelco Hayesep Q 80/100 column, and carbon dioxide and methane concentrations were calculated. After gas sampling was completed, soil core organic matter content was calculated based on ash-free dry weights of subsamples.

Experiment No. 2.

The effect of location within the Wagner Road constructed wetland on soil respiration during exposure to short-term flooded conditions was quantified over an 8-week period. Field sampling locations were selected, based on mean moisture estimates for each plot and plot vegetation weighted average (WA). Moisture estimates were taken monthly in summer, 1989 at each plot and converted to a proposed semi-quantitative moisture modifier scheme (Atkinson, 1991; Table 1).

Vegetation WA was calculated for each plot, using the following formula (Jongman et al., 1987):

$$WA=(y_1u_1 + y_2u_2 + ... + y_mu_m)/100$$

where

$y_1, y_2, ..., y_m$ = the relative percent cover estimates for each species in the plot, and

$u_1, u_2, ..., u_m$ = the indicator values of each species.

Species indicator values were assigned using probability of occurrence in wetlands based on a regional plant list prepared by Reed (1988). The plant list employs an index that ranks species on a 1 to 5 integer scale from obligate wetland to obligate upland (Table 2). Therefore, plot WA must also range from 1 to 5. Colonizing vegetation WA was used to distinguish "upland" (WA>2) and "wetland" (WA<2) portions of the constructed wetland, based on reference wetland WA, which was consistently below 2. Analysis of the colonizing vegetation WA at the Wagner Road constructed wetland site indicated that both WA<2.0 ("wetland" portions) (65.4%) and WA>2.0 ("upland") portions (34.6%) were present within this constructed wetland (Atkinson, et al., in press). Soil cores were placed in 1-L microcosms as in Experiment 1, and were taken from an adjacent upland O horizon (4 cores), an adjacent reference wetland O horizon (4 cores), and from specific plots within the constructed wetland (28 cores). Soil cores from within the constructed wetland were based on vegetation WA of plots and included two "wetland" portions (14 plots), a vegetated "upland" portion (10 plots), and an unvegetated "upland" portion of the constructed wetland (4 plots).

Table 1. Water regime modifiers for mean growing season surface moisture from Cowardin, et al. (1979) and the purposed system.

	Proposed Soil Moisture Modifiers	
Cowardin Modifier	Modifier	Moisture Range/Description
permanently flooded, intermittently exposed, semipermanently flooded	8	standing water >20 cm deep, 100% coverage
seasonally flooded	7	standing water 11-20 cm deep, 100% coverage
	6	standing water (all depths) >50% to 100% coverage
saturated, intermittently flooded	5	soil saturated (> field capacity) or standing water (all depths) 0-50% coverage
temporarily flooded, intermittently flooded	4	soil very moist, near field capacity
not wetland	3	soil dry to moist, below field capacity
	2	soil dry
	1	soil very dry, often blocky

Table 2. Wetland indicator categories based on plant species frequency of occurence in wetlands and the corresponding indicator values for use in WA calculations.

Wetland Indicator Category	Frequency in Wetlands (%)	Indicator Value
Obligate wetland	>99	1
Facultative wetland	67 - 99	2
Facultative	34 - 66	3
Facultative upland	1 - 33	4
Upland	<1	5

Nine piezometers, six in the constructed wetland and three in the reference wetland, were installed and monitored in 1989 and 1991. The reference wetland is classified as "seasonally flooded" using the scheme set forth by Cowardin, et al. (1979).

Elevations at the Wagner Road constructed wetland site were measured at 95 plots along seven parallel transects using a laser light level.

Experiment No. 3

The extent to which redox potential might by lowered throughout a constructed wetland and a reference wetland following a major precipitation event was assessed at the Wagner Road constructed wetland site. Platinum redox electrodes (40 were constructed following the welded redox electrode design of Faulkner, et al. (1989). Redox probes were used to measure oxidation-reduction potentials 24 hours after a major precipitation event at all plots sampled in Experiment 2. Electrodes inserted to a depth of 10 cm, were allowed to stabilize, and were read with a portable voltmeter after insertion of a calomel reference electrode. Mean redox potential of three replicate probes was analyzed for correlation with vegetation WA, moisture estimate, and organic matter content for each plot.

RESULTS

Review of 13 constructed wetlands in Virginia demonstrated that the potential for functional performance was adversely impacted by limited connection to surface hydrology in 69.2% of the sites, by actual or potential erosion and siltation (61.5%), and by proximity to highways (46.2%). The results of the three experiments are reported below.

Experiment No. 1.

Three of four sites produced significantly less carbon dioxide in microcosms containing upland topsoil than in microcosms containing adjacent wetland soil (Figure 2). Methane production was detectable in 11 of 12 reference wetland microcosms, but no methane production was detected in upland topsoil or upland mineral soil microcosms.

Mean organic matter content (21.39%) was variable among reference wetland sites (SD $\pm$18.83); however, reference wetland organic matter was significantly higher than both upland topsoils and mineral soils for all four sites. Mean organic matter content in upland topsoil and mineral soil was 14.39% (SD $\pm$3.68) and 2.98% (SD $\pm$2.13), respectively.

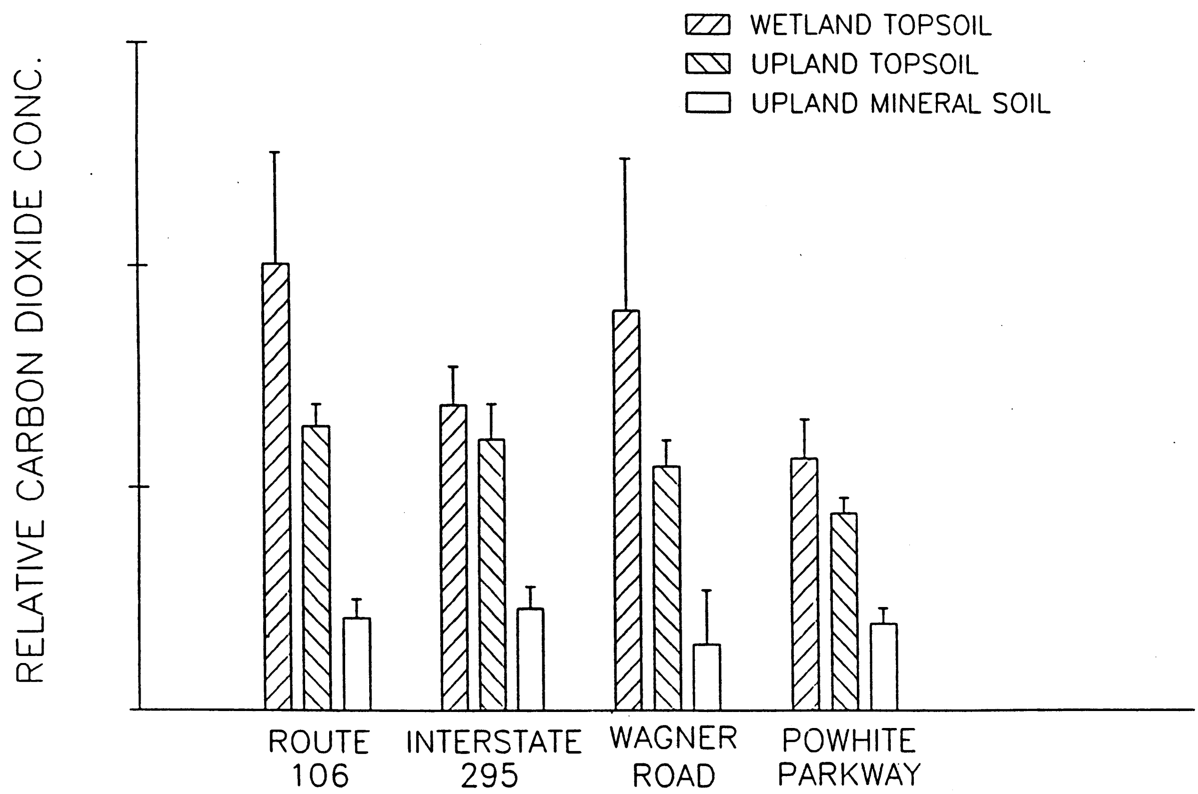


Figure 2. Relative carbon dioxide concentrations measured for three soil types collected from each of four constructed wetland study sites in Southeast Virginia. (Bars represent one standard deviation.)

Experiment No. 2.

Carbon dioxide production in soils from the Wagner Road constructed wetland was significantly lower ($p < 0.05$) than soils from the reference wetland for weeks 2 and 3. Soils from "wetland" plots of the Wagner Road constructed wetland consistently produced more carbon dioxide than soils from "upland" plots, but differences were generally not significant. Soils from bare areas within the constructed wetland consistently produced the lowest amount of carbon dioxide and were significantly different from carbon dioxide production in soils from vegetated "wetland" portions of the constructed wetland. Carbon dioxide production in adjacent upland soil microcosms was lower than the reference wetland each week, although usually not with statistically significance (Figure 3).

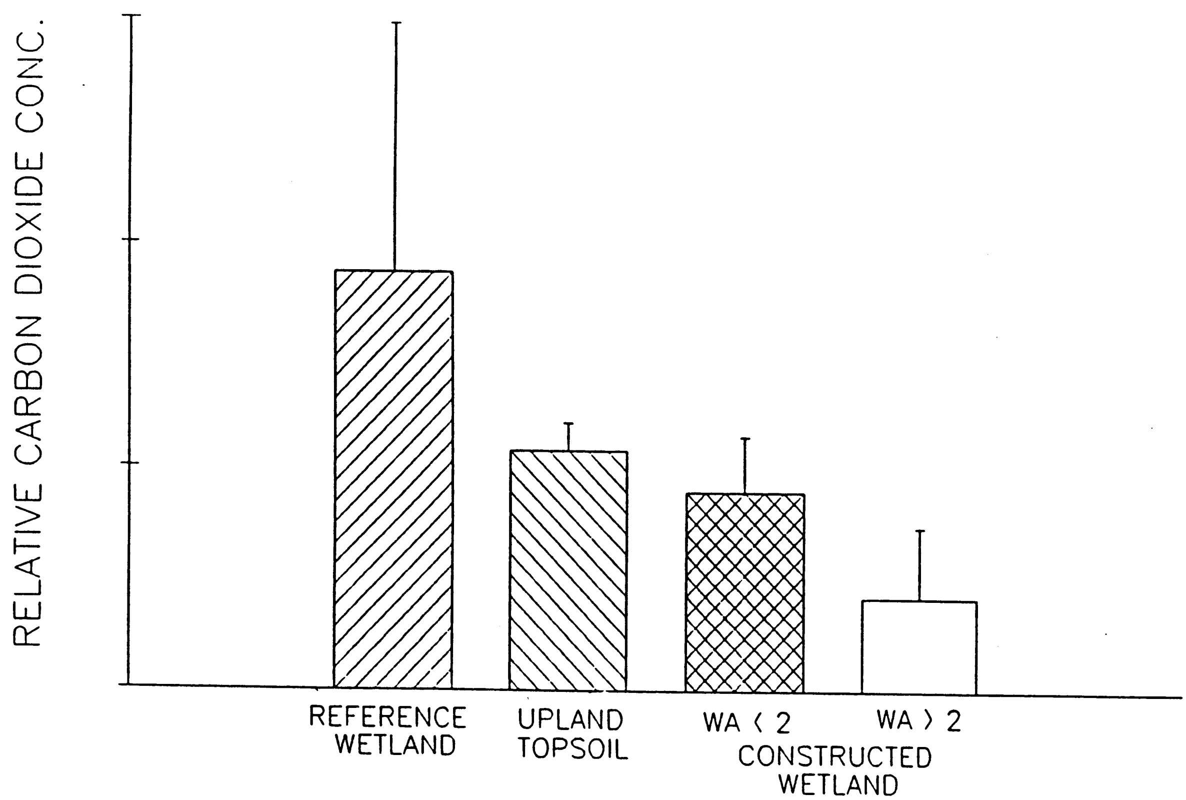


Figure 3. Relative carbon dioxide concentrations measured for soils taken from four locations at the Wagner Road study site in Petersburg, Virginia. (Bars represent one standard deviation.)

By week 3, only the reference wetland contained detectable concentrations of methane, but methane production could not be quantified in any microcosms until weeks 6 through 8. Although only statistically significant in week 7, methane production in the reference wetland and the upland topsoil was consistently greater than constructed wetland methane production throughout the study. At no time during the experiment was methane detected in microcosms containing soil from bare areas of the constructed wetland.

There was significantly ($p < 0.05$) higher percent organic matter in the reference wetland than in all other locations in Experiment 2. Mean percent organic matter was higher in upland topsoils than in constructed wetlands, but not significantly (Figure 4).

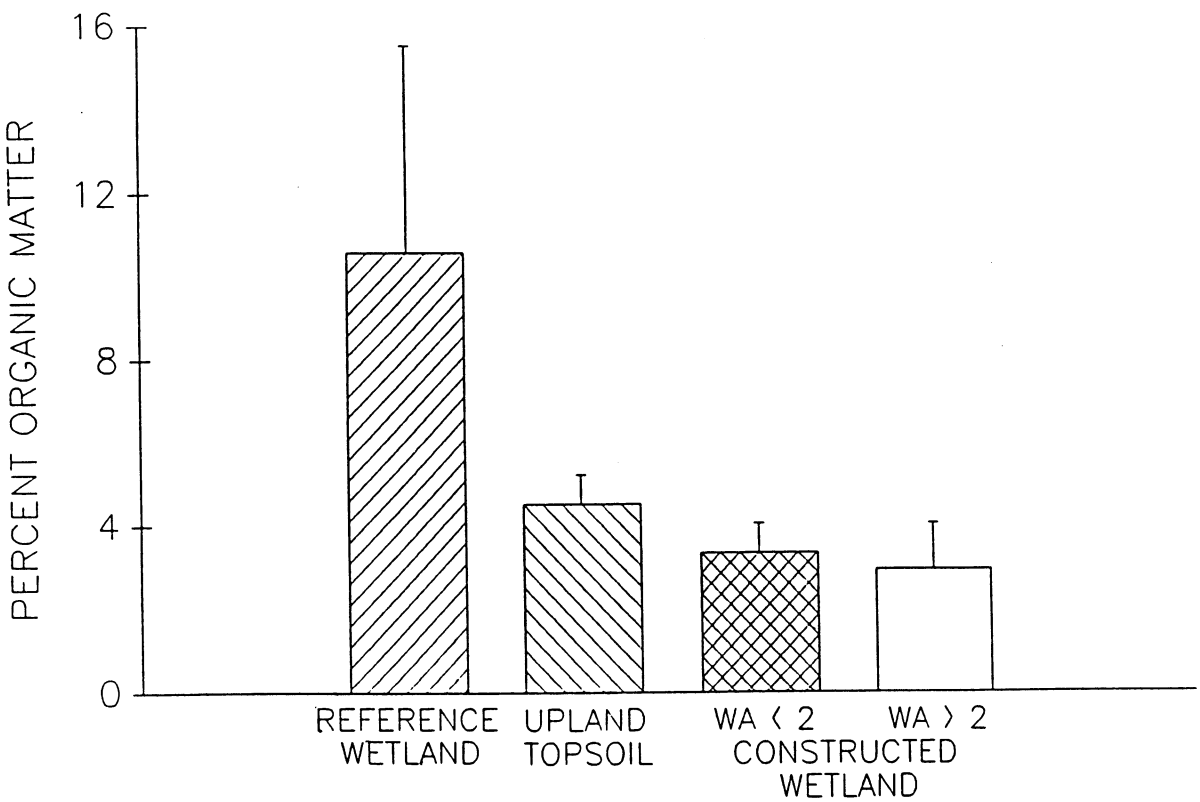


Figure 4. Percent organic matter measured for soils taken from four locations at the Wagner Road study site in Petersburg, Virginia. (Bars represent one standard deviation.)

An elevational range of 0.631 m was measured within the Wagner Road constructed wetland site, and the standard deviation of elevation measurements was ± 0.126. A small ridge crosses the site diagonally. Three of six wells in the constructed wetland demonstrated mean flooding duration shorter than the mean for the reference wetland (Figure 5). Mean moisture estimates ranged from 1.6 ("very dry" to "dry") to 6.4 ("0-10 cm deep" to "11-20 cm deep"). Mean moisture estimate was 3.91 ("dry to moist" to "very moist"), and standard deviation was ± 1.37 (Atkinson, et al., in press).

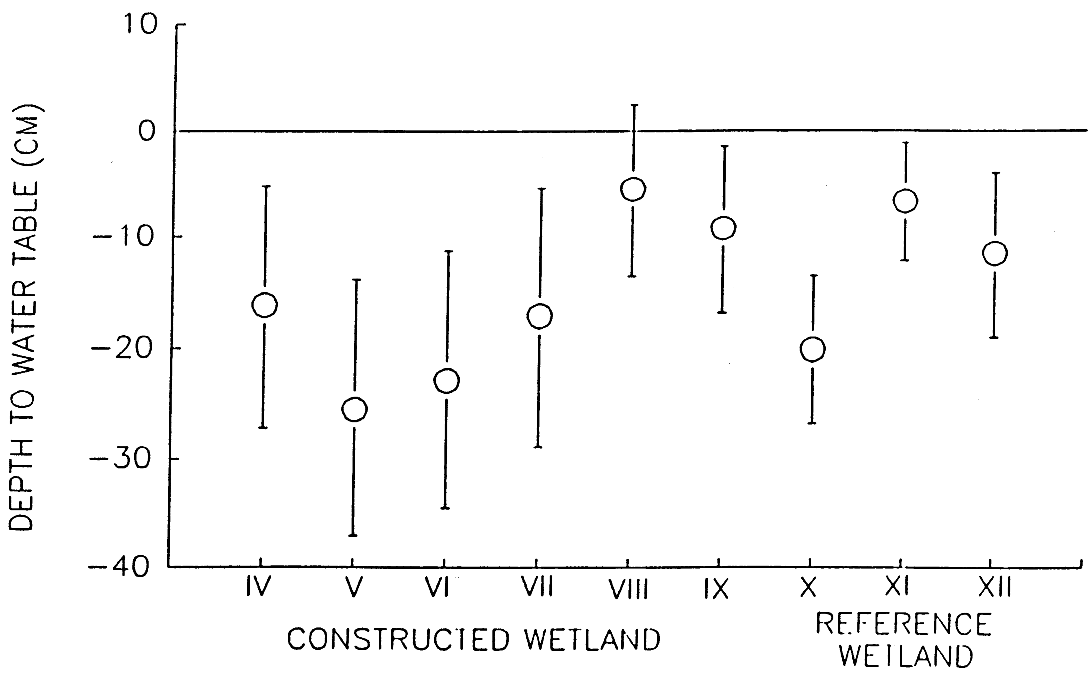


Figure 5. Depth to water table for summer, 1989, in constructed wetland wells (IV-IX) and reference wetland wells (X-XII). Wells I-III were located in adjacent uplands, and no water was detected.

Experiment No. 3.

Field redox measurements were taken in plots throughout the Wagner Road constructed wetland for 24 hours after a significant precipitation event. Redox potentials throughout most of the constructed wetland were significantly higher (less reducing) than redox potentials in the reference wetland and the typically flooded "wetland" portions of the constructed wetland. Redox potential measurements in the constructed wetland were positively correlated with vegetation WA (r^2 = 0.91) and negatively correlated with moisture estimates (r^2 = -0.87). Redox potentials for unvegetated "upland" portions of the constructed wetland were indicative of oxygen content near saturation. Unfortunately, persistent dry conditions in the "upland" portions of the constructed wetland limited site-wide sampling to only one day following a heavy rain.

DISCUSSION

A frequently used procedure for freshwater wetland construction involves excavation to the approximate elevation of the water table and then leaving the exposed mineral soil without amendments. Implied assumptions are: (a) apprapriate hydrologic conditions will be established; (b) hydric soils will develop and hydrophytic vegetation will become established; and (c) wetland functions and ecosystem services will be replaced.

The results from Experiment 1 suggest that, if a low organic matter mineral soil was left without amendments, then soil respiration would be significantly lower than in the reference wetland. Although not field validated here, the results further suggest that soil respiration could be enhanced by spreading wetland soil over a site. In addition, spreading upland topsoil over a site may also lead to considerably more soil respiration than leaving an upland mineral soil unamended. However, methane production in Experiment 1 was limited to the wetland soil, which indicated that a more negative oxidation-reduction system might be established if wetland soils were spread instead of upland topsoil. Methane production may have eventually occurred in microcosms containing upland topsoils, but results of this study indicate that natural wetland soil becomes reduced more rapidly following inundation.

In Experiment 2, within site sampling of the Wagner Road constructed wetland in the 4 years following construction supports the predictions based on Experiment 1. Microcosms containing the Wagner Road constructed wetland soil exhibited significantly lower soil respiration than the reference wetland soil. Significant differences in soil respiration were also found at two plant communities within the Wagner Road constructed wetland (Figure 3).

The results of a delineation of the Wagner Road constructed wetland suggest that portions of the constructed wetland site may be, or may be becoming, wetland, while other areas may remain upland. These "wetland" portions of the site occur at lower elevations sustain longer periods of inundation, have more hydrophytic vegetation (lower WA and higher percent cover) than "upland" portions of the constructed wetland, and demonstrate consistently greater soil respiration (Figure 3). However, organic matter content differences among plots within the constructed wetland are not significant (Figure 4), perhaps due to the relatively young age of the site (4 years at the time of sampling).

The practice of spreading wetland soil over constructed wetlands was not widely employed in Virginia at the time of this study. Organic matter content, microbial species, and wetland vegetation propagules may be established by this practice. While soil amendments may increase the likelihood that some functions will be performed sooner, inappropriately high elevations could lead to rapid decomposition of organic matter and higher redox potentials. Alternating aerobic and anaerobic conditions facilitate organic matter decomposition (Reddy and Patrick, 1975). Kadlec (1989) states that organic matter-rich peat may take millenia to decay, but exposure to the atmosphere would greatly accelerate the process. Organic matter accumulation may not occur over time in the higher elevation portions of the site with alternating aerobic and anaerobic conditions. Therefore, elevation and hydrology must be taken into consideration in planning for soil applications.

In Experiment 3, field redox potentials were taken at the same soil core locations sampled in Experiment 2. Experiments 1 and 2 characterized soil respiration under flooded conditions; therefore, Experiment 3 was to be conducted under flooded conditions following a heavy precipitation event. After the entire constructed wetland had been inundated for 24 hours, reference wetland redox potential was found to be significantly lower than redox potential in most of the Wagner Road constructed wetland. Redox potential seemed to reflect the frequency and/or duration of saturation, since areas with wetter moisture estimates exhibited lower redox potential (r^2 = -0.87).

Constructed wetland redox potentials were strongly correlated with both vegetation WA and moistures estimates. These results tend to support the assertion based on Experiment 2: soil respiration differs among "wetland" and "upland" portions within the Wagner Road constructed wetland, and that frequency and/or duration of saturation may be an important factor.

CONCLUSIONS

This study showed that elevation, hydrology, and soil organic matter influenced carbon dioxide production and redox potential. Longer flooding periods may be required for lower redox potential to develop in the constructed wetland than the adjacent reference wetland, perhaps resulting from lower organic matter content and lower soil respiration in the former. A large portion (34.6% of the Wagner Road constructed wetland) seldom exhibited saturated soil conditions. If these hydrologic conditions persist, much of the Wagner Road constructed wetland may lack the flooding frequency and/or duration necessary for organic matter accumulation and for development of strongly reducing conditions.

Soil amendments are suggested by the data from all three experiments. Demucked soils from the area receiving fill material should be spread over the wetland under construction. The additional organic matter and microbial community established by this procedure could increase the likelihood of successful replacement of water quality functions. In addition, the spread of wetland soil may lead to higher macrophyte productivity and increased organic matter content. The timing of soil amendments may also influence success. If wetland construction is performed concurrently with the permitted wetland filling, wetland soils could be spread immediately over the sites. Timing construction in this fashion would reduce the effects of stockpiling wetland soil and would provide a longer period for wetland establishment.

Ecological services dependent upon reducing conditions are probably not performed by most of the Wagner Road constructed wetland when compared to an adjacent reference wetland. Over time, anaerobic conditions may predominate in wetter portions at lower elevations within the constructed wetland. While organic matter may be expected to accumulate in these "wetland" portions that have high cover by hydrophytic plant species, decompositional processes may keep pace with accumulation rates in portions of the constructed wetland site that are currently characterized as "upland." Such areas may not provide functions dependent on wetland soil, hydrology, and vegetation if current hydrologic conditions persist.

Many ecological services of wetlands are provided because of the hydrologic connectivity to a river or stream. The fact that many constructed wetlands do not exhibit the same connectivity found among reference wetlands suggests that these sites lack the opportunity to perform many water-quality enhancement services. Erosion and siltation may also limit the long-term performance of many ecological services. Finally, the practice of constructing wetlands beside highways, in interstate loops, or near any development may alter the ecological services provided. Such constructed wetlands may provide treatment of runoff instead of the intended compensation for wetland loss.

ACKNOWLEDGMENTS

We would like to express appreciation for the field assistance of Dave Barber, James Perry, and Carl Hershner, Sr. Manuscript review was provided by James Perry. The editorial assistance of Darla Donald is gratefully appreciated. Financial support from the Virginia Academy of Science and the E. I. duPont de Nemours Education Foundation is acknowledged.

LITERATURE CITED

Atkinson, R. B. 1991. An Analysis of Palustrine Forested Wetland Compensation in Virginia. Ph.D. Dissertation, Department of Biology, Virginia Polytechnic Institute and State University, Blacksburg, Virginia. 126 pp.

Atkinson, R. B., J. E. Perry, E. P. Smith, and J. Cairns, Jr. In press. Use of created wetland delineation and weighted averages as a component of assessment. Wetlands.

Brinson, M. M. and L. C. Lee. 1989. In-kind mitigation for wetland loss: Statement of ecological issues and evaluation of examples. Pages 1069-1085 in Freshwater Wetlands and Wildlife, R. R. Sharitz and J. W. Gibbons (Editors). U.S. Department of Energy, Washington, D.C.

Brinson, M. M., H. D. Bradshaw, and E. S. Kane. 1984. Nutrient assimilative capacity of an alluvial floodplain swamp. J. Appl. Ecol. 21:1041-1057.

Cairns, J., Jr. In press. Maintaining per capita ecosystem services while the human population grows to ten billion: A major global challenge for biologists. Assoc. Southeast. Biol. Bull.

Choate, K. D., J. T. Watson, and G. R. Steiner. 1990. Demonstration of Constructed Wetlands for Treatment of Municipal Wastewaters, Monitoring Report for the period: March, 1988 to October, 1989. TVA/WR/WQ--90/11, Tennessee Valley Authority, Chattanooga, Tennessee. 107 pp.

Cowardin, L. M., V. Carter, F. C. Golet, and E. T. LaRoe. 1979. Classifications of Wetlands and Deepwater Habitats of the United States. U.S. Fish and Wildlife Service, Washington, D.C. 103 pp.

Dahl, T. E. 1990. Wetland Losses in the United States. U.S. Fish and Wildlife Service, Washington, D.C. 21 pp.

Day, F. P., Jr. 1989. Limits on decomposition in the periodically flooded, non-riverine Dismal Swamp. Pages 153-166 in Freshwater Wetlands and Wildlife, R. R. Sharitz and J. W. Gibbons (Editors). U.S. Department of Energy, Washington, D.C.

Faulkner, S. P. and J. C. Richardson. 1989. Physical and chemical characteristics of freshwater wetland soils. Pages 41-72 in Constructed Wetlands for Wastewater Treatment, D. A. Hammer (Editor). Lewis Publishers, Chelsea, Michigan.

Faulkner, S. P., W. H. Patrick, Jr., and R. P. Gambrell. 1989. Field techniques for measuring wetland soil parameters. Soil Sci. Soc. Am. J. 53:883-890.

Gambrell, R. P. and W. H. Patrick, Jr. 1978. Chemical and biological properties of anaerobic soils and sediments. Pages 375-423 in Plant Life in Anaerobic Environments, D. D. Hook and R. M. M. Crawford (Editors). Ann Arbor Science, Ann Arbor, Michigan.

Hammer, D. A. 1989. Constructed Wetlands for Wastewater Treatment. Lewis Publishers, Chelsea, Michigan. 831 pp.

Jongman, R. H., C. F. T. ter Braak, and O. F. R. van Tongeren. 1987. Pages 83-86 in Data Analysis in Community and Landscape Ecology. Pudoc Wageningen, Weningen, The Netherlands.

Kadlec, R. H. 1989. Dynamics of inorganic and organic materials in wetlands ecosystems. Pages 459-468 in Constructed Wetlands for Wastewater Treatment, D. A. Hammer (Editor). Lewis Publishers, Chelsea, Michigan.

Pearsall, W. H. 1938. The soil complex in relation to plant communities. J. Ecol. 26:108-209.

Ponnamperuma, F. N. 1972. The chemistry of submerged soils. Adv. Agron. 24:29-92.

Quammen, M. L. 1988. Measuring success in wetlands mitigation. Pages 242-245 in Proceedings of the National Wetlands Symposium: Mitigation of Impacts and Losses, J. A. Kusler, M. L. Quammen, and G. Brooks (Editors). Association of State Wetland Managers, Berne, New York.

Race, M. S. and D. R. Christie. 1982. Coastal zone development: mitigation, marsh creation, and decision making. Environ. Manage. 6:317-328.

Reddy, K. R. and W. H. Patrick, Jr. 1975. Effect of alternate aerobic and anaerobic conditions on redox potential, organic matter decomposition and nitrogen loss in a flooded soil. Soil Biol. Biochem. 7:87-94.

Reed, P. B., Jr. 1988. National List of Plant Species that Occur in Wetlands: 1988, Virginia. NERC-88/18.46, U.S. Fish and Wildlife Service, St. Petersburg, Florida. 115 pp.

Sather, J. H. and R. D. Smith. 1984. An Overview of Major Wetland Functions. FWS/OBS- 84/18, U.S. Fish and Wildlife Service, Washington, D.C. 68 pp.

Watson, J. T., S. C. Reed, R. H. Kadlec, R. L. Knight, and A. E. Whitehouse. 1989. Performance expectations and loading rates for constructed wetlands. Pages 319-351 in Constructed Wetlands for Wastewater Treatment, D. A. Hammer (Editor). Lewis Publishers, Chelsea, Michigan.

Wharton, C. H. and M. M. Brinson. 1979. Characteristics of southeastern river systems. Pages 32-40 in Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems, R. R. Johnson and J. F. McCormick (Technical Coordinators). GTR-WO-12, U.S. Forest Service, Washington, D.C.

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

TREATMENT OF STORMWATER BY A NATIVE HERBACEOUS MARSH

David W. Carr
Betty Rushton
SOUTHWEST FLORIDA WATER MANAGEMENT DISTRICT
2379 Broad Street (U.S. 41 South)
Brooksville, Florida 34609-6899

ABSTRACT

A native herbaceous marsh was studied for its efficiency to treat stormwater and observe any changes in vegetation caused by stormwater input. The marsh successfully reduced pollutant loads at its discharge for inorganic nitrogen, zinc, cadmium, and total suspended solids and was unsuccessful with organic nitrogen, phosphorus and iron. An unmeasured groundwater input was faulted for all loads not reduced, except organic nitrogen which was probably caused by marsh vegetation. Rain that fell directly onto the marsh represented 57.9% of the total input volume.

A designed increase in slope to the marsh edge, construction of a basin and a landscaped lawn allowed the nuisance Typha latifolia and Ludwigia peruviana to invade. The first of two detailed vegetation analyses showed Nymphoides aquatica, Ludwigia peruviana and Utricularia species to be of spatial interest along the stormwater flow path.

INTRODUCTION

The use of certain jurisdictional wetlands for the treatment of stormwater was approved by the Florida legislature in 1984 (Chapter 17-25, Florida Statutes) and by 1988, the Southwest Florida Water Management District (SWFWMD) included wetland treatment as a stormwater management practice. In March 1991, SWFWMD implemented a two and one-half year project to study a natural marsh used to treat stormwater (four months of data collection remains).

A 1.2-hectare marsh was selected at a corporate office park in Tampa, FLorida (Figure 1) which receives runoff from a 6.2-ha drainage basin. The marsh was categorized as a flag marsh (Kushlan, 1990) in that the center was dominated by a dense stand of pikerelweed (Pontedaria cordata) loosely surrounded by pools of water-lily (Nymphaea odorata and Nymphoides aquatica). Maidencane (Panicum hemitomon) grew along the edge of the marsh and invaded the water-lily pools. Primrose willow (Ludwigia peruviana) and wax myrtle (Myrica cerifera) inhabited the marshes' outermost edge to the south. The adjacent uplands were a hardwood-conifer mixed forest.

Two objectives of the study were: 1) to determine the effectiveness of the system to treat stormwater (i.e. to reduce pollutant loads from the inflows to the outflow) 2) to measure any ecological changes on a natural marsh caused by stormwater input.

Wetland Location Map

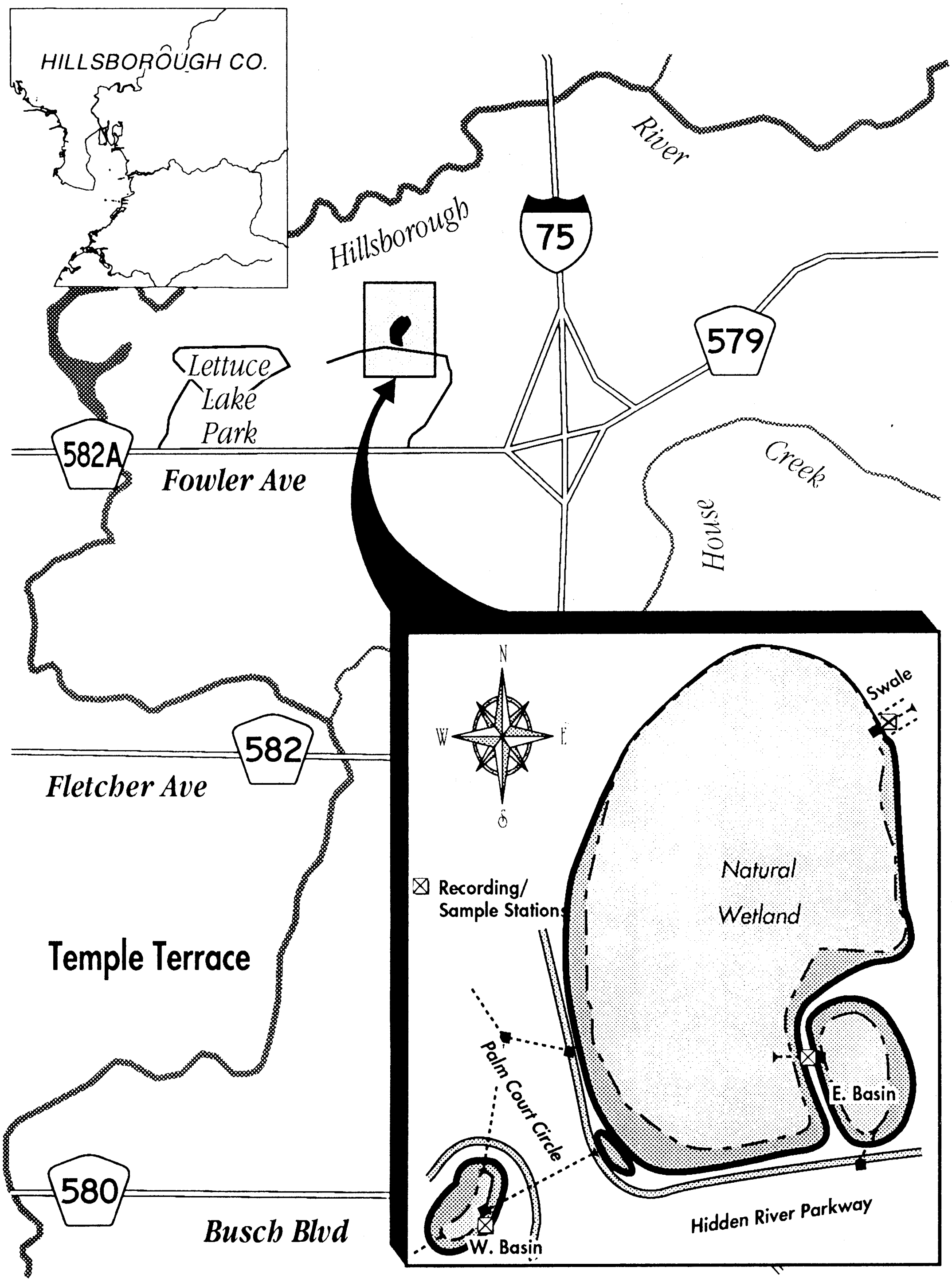


Figure 1. Location map of the marsh showing recording stations.

The specific research objectives were: a) to calculate a pollutant mass loading budget using the data collected May-August, 1991; and, b) to interpret aerial infrared photographs for vegetative cover (1973 and 1992) and perform a detailed vegetation analysis July, 1992 to observe vegetative changes/trends within the marsh due to stormwater input. An additional detailed analysis and infrared photography of the vegetation are scheduled near the end of the study (July and August 1993).

MATERIALS AND METHODS

Vegetation Analysis

Seventy-eight, one-meter-square quadrats were analyzed for percent cover in July, 1992. Of these, seventy-two quadrats were selected along six transects (Figure 2). The six remaining quadrats (13-16, 28 and 29) were selected randomly at the end of transects 1 and 2 by tossing the quadrat frame over the investigators right shoulder. Prior to entering the marsh, transect lines were selected in an attempt to obtain equal representation from the various communities along the depth gradient. Stakes were driven every five or ten meters along each transect. These stakes represented a distinctive corner of each quadrat.

A one-meter-square quadrat frame was placed over each plot by using the methods described above; vegetation that may have been trapped under the frame was restored to its original position. Two observers made field identifications of the plant species present in each plot and together determined the percent cover each species occupied. Representative samples of each species were archived and field identifications later verified using Dressler, et al., (1987); Godfrey and Wooten (1979).

To determine possible spatial trends within the marsh due to stormwater management, the quadrat percent cover data results were grouped into north (transects 1, 3, 6 and quadrats 62-70) and south (transects 2, 4, 7 and quadrats 54-61) arrangement.

A color infrared aerial photograph of the marsh and its adjacent uplands was taken in August, 1992. For a pre-stormwater impact comparison, a January, 1973 black and white aerial photograph of the marsh was obtained. Photo-interpreted vegetation maps were made and verified with quadrat data. Vegetation maps were recorded on a G.I.S. system (Geographic Information System).

Hydrology and Water Quality

Stormwater runoff enters the marsh at its south end from two pretreatment basins (East and West) and discharges through a single outflow at the north end. Instrument structures were installed at each pre-treatment basin's point of discharge into the marsh and at the outflow (Figure 1). At each station, instruments continuously recorded hydrology and rainfall data at 15-minute intervals.

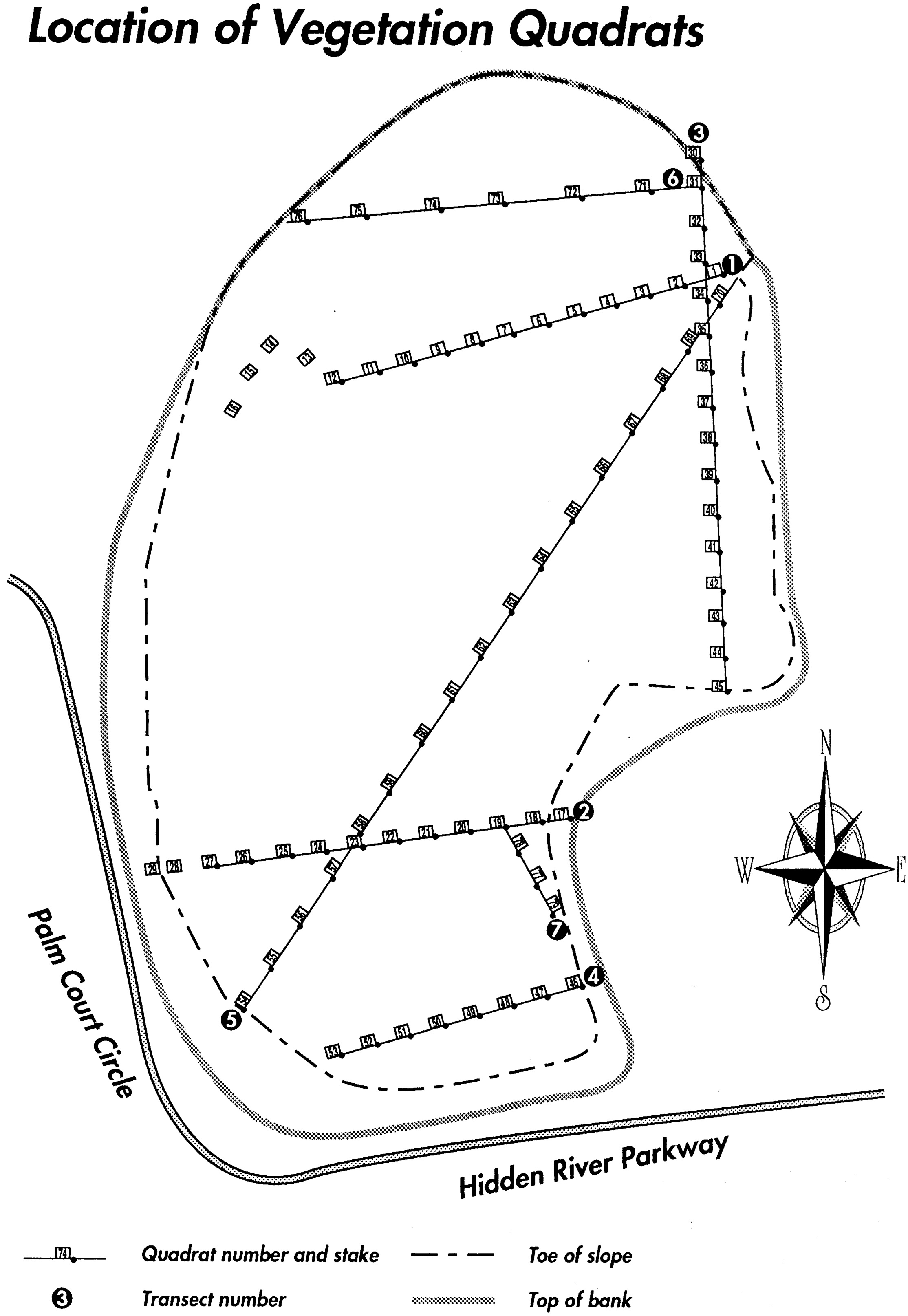


Figure 2. Location of transects and quadrats for the detailed vegetation analysis.

Load efficiency was based on measurements from 18 storm events taken during the summer of 1991 (USEPA, 1983).

Load efficiency (%) = (SOL in - SOL out/SOL in)*100

where: SOL is the sum of loads in cubic feet for the summer of 1991. Loads were calculated by multiplying the constituent concentration by volume and converting to cubic feet. SOL includes both basins, as well as rainfall volume loads. Rainfall loads were calculated as all the rain that fell directly on the marsh.

Composite flow weighted surface water quality samples were taken by automatic refirgerated samplers at each station. Rainfall water quality samples were collected at the outflow by an automatic-sensing wet/dry precipitation collector. Refrigerated samples were collected, preserved and transported to the SWFWMD lab for analysis, usually within twelve hours after storm events.

All water quality samples were analyzed for metals (cadmium, chromium, copper, lead, manganese, nickel, iron, and zinc), nutrients (ammonia, total nitrogen, total phosphorus, ortho phosphorus and organic nitrogen), hardness, and total suspended solids (depending on amount of sample). In addition, total organic carbon was analyzed from grab samples.

Field Parameter Analayis

Field parameters were obtained from multi-parameter sensor units. The parameter of primary interest to stormwater treatment, pH and dissolved oxygen, will be discussed. Continuous readings were recorded for sixteen distinct one-week intervals at three locations. For spatial analysis, field parameters were recorded at nine additional locations throughout the marsh on October 12, 1992.

RESULTS

Vegetation Analysis

The 1992 detailed vegetation analysis revealed 34 plant species within the 78 quadrats (Table 1). Those categories with the highest percent cover were Panicum hermitomon (21.82%), open water (17.65%), Pontederia cordata (16.27%), litter (13.83%), and fragrant white water-lily (Nymphaea odorata, 6.31%).

Table 1. 1992 Detailed Vegetation Analysis Results.
Wetland percent cover as a whole and grouped into north and south.

GENUS SPECIES	PERCENT COVER	PERCENT COVER NORTH	PERCENT COVER SOUTH
Panicum hemitomon	21.82	20.62	23.65
Open water	17.65	16.68	19.13
Pontederia cordata	16.27	17.00	15.16
Litter	13.83	14.19	13.29
Nymphaea odorata	6.31	6.53	5.97
Sagittaria lancifolia	3.21	1.49	5.81
Polygonum hydropiperoides	2.04	2.45	1.42
Eupatorium capillifolium	1.65	1.40	2.03
Andropogon virginicus	1.60	1.81	1.29
Open water w/Litter	1.47	2.45	0.00
Erechtites hieracifolia	1.33	0.81	2.13
Cladium jamaicense	1.28	2.13	0.00
Eleocharis vivipara	0.90	1.17	0.48
Leerasia hexandra	0.88	1.47	0.00
Rhynchospora inundata	0.81	1.34	0.00
Ludwigia suffruticosa	0.81	0.70	0.97
Hydrocotyle umbellata	0.78	0.87	0.65
Amphicarpum muhlenbergianum	0.77	1.28	0.00
Rhexia mariana	0.71	0.11	1.61
Nymphoides aquatica	0.69	1.11	0.06
Juncus effusus	0.64	1.06	0.00
Woodwardia virginica	0.51	0.00	1.29
Proserpinaca pectinata	0.41	0.64	0.06
Quercus laurifolia	0.38	0.64	0.00
Utricularia fibrosa	0.38	0.32	0.48
Lachnanthes caroliniana	0.38	0.00	0.97
Utricularia unknown	0.32	0.00	0.81

Table 1. (continued)

Ludwigia peruviana	0.32	0.00	0.81
Algae	0.32	0.00	0.81
Bacopa unknown	0.26	0.43	0.00
Utricularia purpurea	0.26	0.32	0.16
Polygonum hirsutum	0.19	0.32	0.00
Centella asiatica	0.18	0.11	0.29
Rhynchospora microcephala	0.15	0.26	0.00
Mikania scandens	0.13	0.00	0.32
Hypericum hypericoides	0.13	0.21	0.00
Baccharis halimifolia	0.01	0.00	0.03
Field error	0.19	0.11	0.32

The marsh received greater concentrations of stormwater from both basins at its south end. Stormwater received treatment as it flowed to the north and discharged at the outflow structure (Figure 1). Trends detected along the flow gradient from south to north are as follows (Table 1):

1. Fragrant water-lily (Nymphaea odorata) and floating-heart (Nymphoides aquatica) two water-lily species observed. N. aquatica was more prevalent in the north end of the marsh (1.11%) than in the south (0.06%). N. odorata was observed to be evenly distributed throughout the marsh with 6.53% in the north and 5.97% in the south.

2. Species found in the north end of the marsh which had zero percent cover in the south end (>1.0%) were: sawgrass (Cladium jamaicense, 2.13%), southern cutgrass (Leerasia hexandra, 1.47%), horned-rush (Rhynchospora inundata, 1.34%), blue maidencane (Amphicarpum muhlenbergianum, 1.28%) and soft rush (Juncus effusus, 1.06%).

3. Some species found in the south end of the marsh which had zero percent cover in the north end were: Virginia chain fern (Woodwardia virginica, 1.29%), red-root (Lachnanthes caroliniana, 0.97%) and primrose willow (Ludwigia peruviana, 0.81%).

4. The bladderworts (Utricularia fibrosa, purpurea and unknown) were observed at the north end with 0.32, 0.32, and 0.00 percent covers were observed at the south end with 0.48, 0.16, and 0.81% covers, respectively.

5. Although not within a quadrat, a few common cattail (Typha latifolia) were observed near quadrat #54 at the time of the survey.

The August 1992 photographic interpretation of the marsh (Figure 3) revealed the north edge of the marsh had been left natural with a 5% slope to the uplands. While the south end, which receives the greatest influence from stormwater, has had its edge altered. Design of the stormwater system has increased the south slope to an average 25.5% slope and a landscaped lawn was maintained. It is along the south edge that the nuisance species L. peruviana has invaded. The west basin has become choked with cattail (Typha latifolia) which has begun to invade the marshes of southwestern edge. The east basin remained unvegetated despite the lack of herbicide application probably because it is relatively deep with mowed grass margins.

Ground truthing the 1992 aerials enabled a species-specific level of detail in the interpretation. The inability to ground truth the 1973 aerial limited its interpretation to much broader classifications (Figure 4) (FDOT, 1985). The road and the stormwater system design were constructed in two phases during the late 1980's. As a result, the pre-treatment basins were constructed and the southernmost wet prairie peninsular portion of the marsh was filled for a road. An east-central area adjacent to the marsh was excavated to mitigate for the loss of wetland (miscellaneous grasses, Figure 3).

Hydrology and Water Quality

A water budget for the summer of 1991 shows measured input volume from both basins and rainfall, as well as discharge volume on a monthly basis (Table 2). Evapotranspiration and percolation were not taken into consideration. The marsh did not discharge in May or June, but discharged continuously from July 9 to September 17. During this period of discharge, more volume left the marsh than entered it. For the water measured entering the marsh, the east basin contributed a mean 25.5%, the west basin 16.6%, and the volume of water that entered the marsh directly from rainfall was 57.9%. Groundwater was also a significant input that has not, as yet, been quantified. Surficial water table wells were installed in January, 1992 and since then, levels have been continuously recorded which may help in estimating groundwater inputs.

Table 2. 1991 WATER BUDGET.

MONTH	TOTAL IN Cubic feet	TOTAL OUT Cubic feet
MAY	172429	0
JUNE	198019	0
JULY	375176	420776
AUGUST	268948	354447
SEPTEMBER	21826	525
TOTAL	1036398	775748

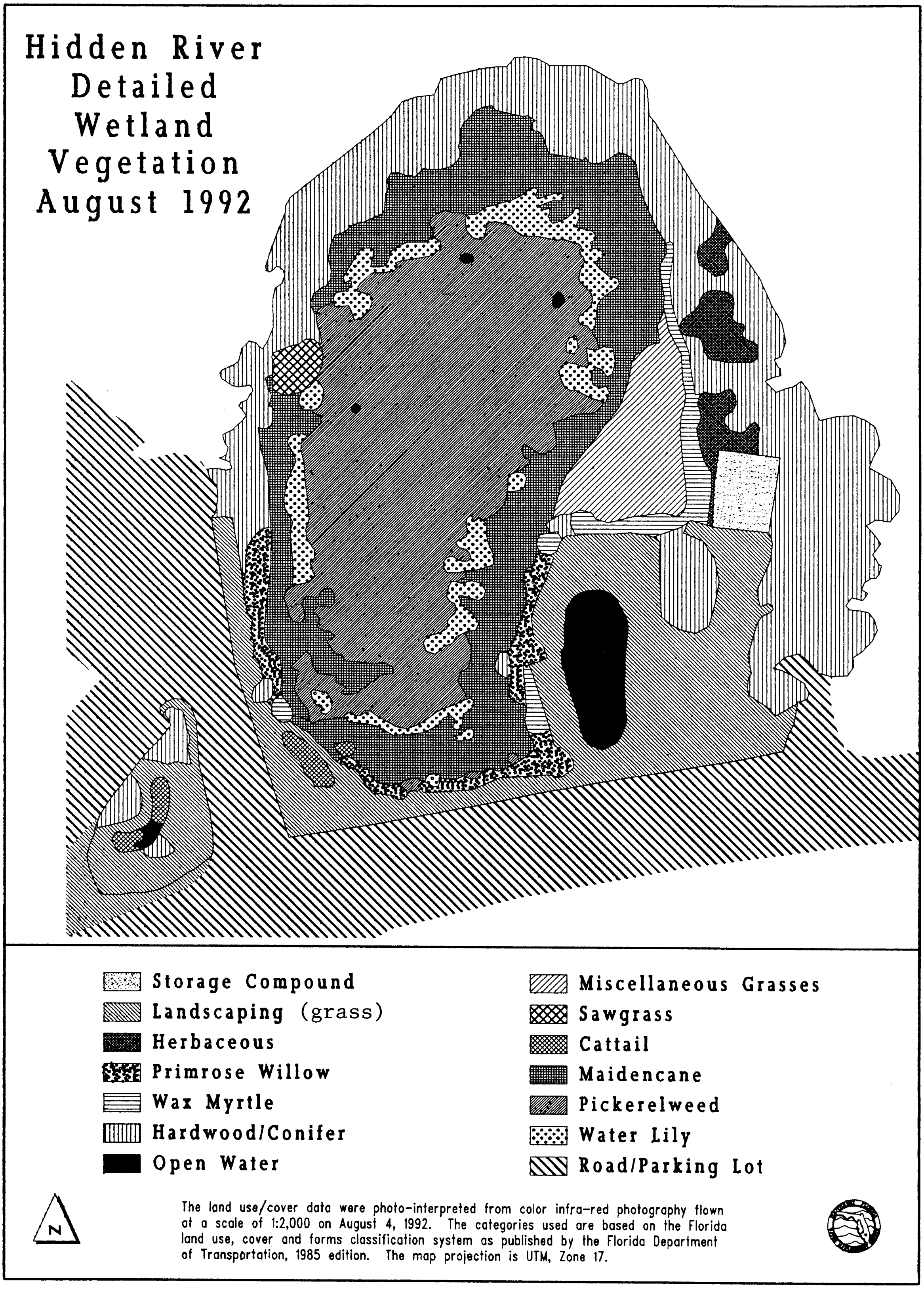


Figure 3. Photographic interpretation of an August, 1992 aerial for vegetative cover.

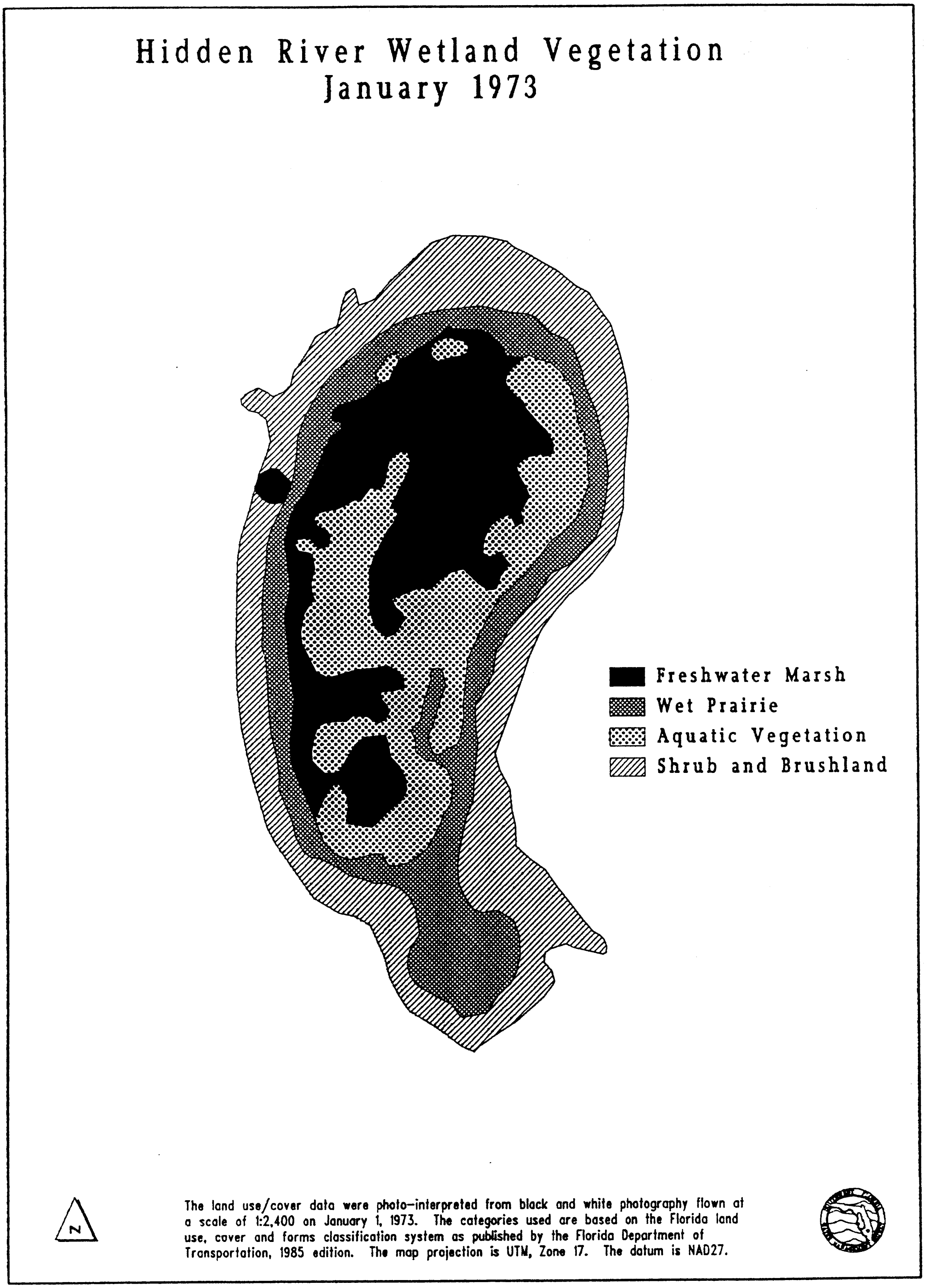


Figure 4. Photographic interpretation of a January, 1973 aerial for vegetative cover.

Among the inflows (east basin, west basin and rainfall), rainfall had the highest pollutant load for ammonia (929.08g), nitrate (NO_2) + nitrite (NO_3) (1744.04g) and cadmium (63.78g) (Table 3). Additionally, organic nitrogen (5467.08g), ortho phosphorus (316.04g), total phosphorus (1108.4g), iron (3141.25g) and total suspended solids (T.S.S.) (95.81kg) loads from the east basin and zinc (592.1g) loads from the west basin are the highest among the inflows.

Table 3. POLLUTANT LOAD EFFICIENCIES SUMMER OF 1991.

	East (gm)	West (gm)	Rain (gm)	Out (gm)	Ground Water Conc.	Load Efficiency %
AMMONIA-N	351.01	336.64	929.08	1183.38	9x	26.80
NO2+NO3	381.96	806.94	1744.04	557.31	same	81.00
ORGANIC-N	5467.08	1439.79	495.01	22883.34	2x	-209.16
ORTHO-P	316.04	143.75	70.73	851.55	6x	-60.51
TOTAL-P	1108.44	193.87	40.46	1641.80	4x	-22.27
ZINC	137.77	592.10	166.19	482.09	same	46.20
CADMIUM	36.02	36.16	63.78	107.27	same	21.10
IRON	3141.25	739.96	616.23	13146.17	4x	-192.30
T.S.S.	95.81	44.90	12.11	73.18	---	52.12

Note: T.S.S. is in Kg.

Pollutant load reduction at the outflow demonstrated positive load efficiency percentages for ammonia, NO_2 + NO_3, zinc and cadmium and T.S.S. Negative percent efficiencies were found for organic N., total P, ortho P, and iron.

Groundwater concentrations (mg/l) were found to be essentially the same as surface waters for N02 _ NO3, zinc, and cadmium (Table 3). Concentrations were 9, 2, 6, and 4 times greater in groundwater than surface waters for ammonia, organic N, ortho P, total P and iron, respectively.

Field Parameter Analysis

Dissolved oxygen (D.O.) concentrations observed in both basins were 3.74 mg/L and 7.8mg/L (west and east basins respectfully) and were substantially higher than the marsh (Figure 5). The D.O. levels in the marsh were measured below 1.25mg/L. The west basin and east basin recorded pH levels of 7.2 and 8.1 respectfively; these levels were higher than those observed in the marsh (5.2 - 6.0).

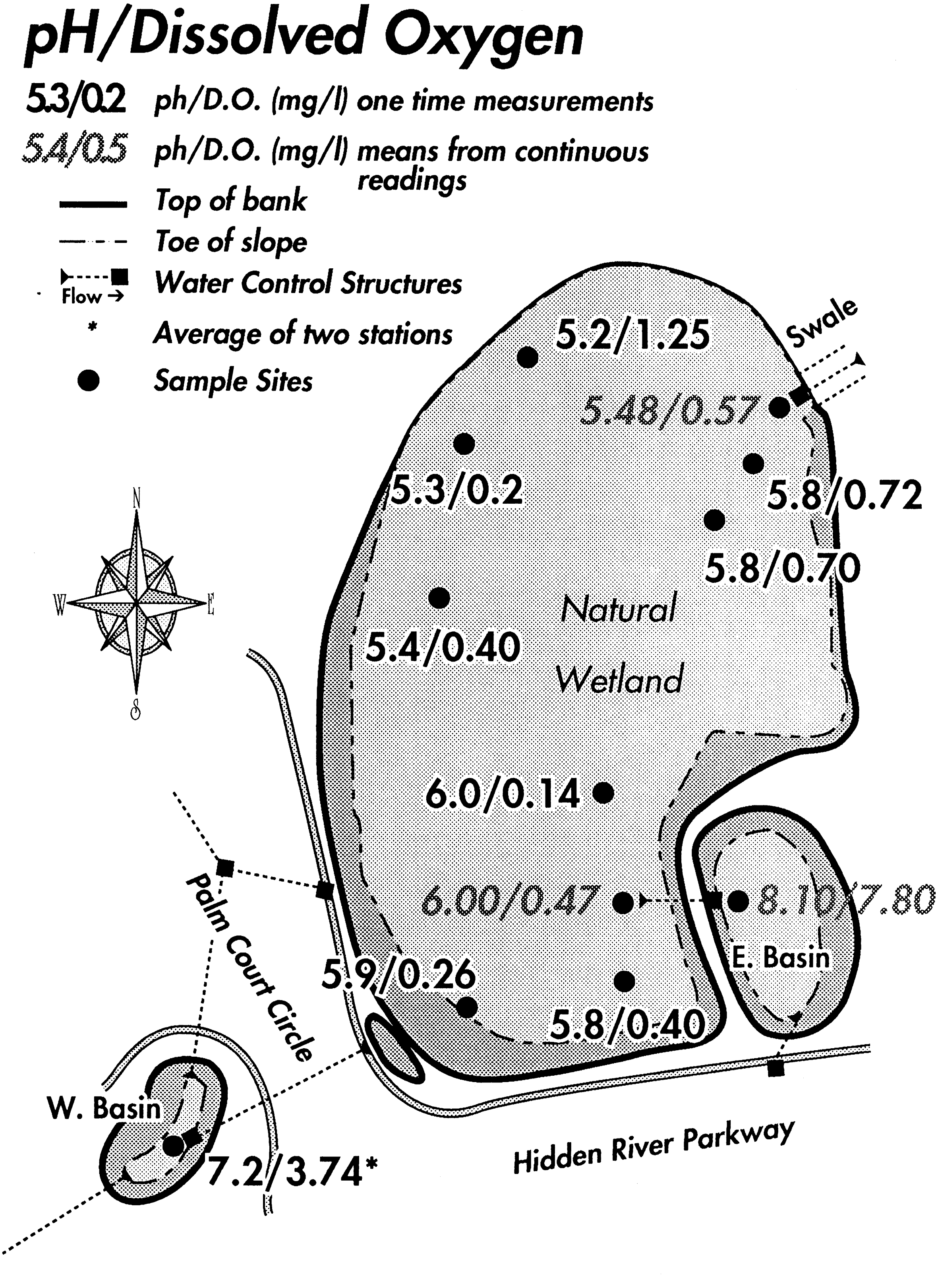


Figure 5. Measurments of pH and dissolved oxygen of both pre-treatment basins and throughout the marsh.

DISCUSSION

Vegetation Analysis

Historically, the 1973 aquatic vegetation (water-lily and open water) were more dominant than observed in 1992 (Figures 3 and 4). Two factors that may have contributed to the decrease of open water are: 1) influence form the treatment system (increased nutrient loading form stormwater and the fertilized landscape); and, 2) the outflow weir elevation which may be lower than the historical discharge elevation.

Alteration to the marshes' southern edge and construction of the west basin probably led to the establishment of two nuisance species. After the construction of west basin, T. latifolia established itself and has since invaded the southwestern corner of the marsh. The altered south edge of the marsh has seen an invasion of L. peruviana. Neither T. latifolia nor L. peruviana have been observed on the marshes' natural north edge.

The 1992, detailed vegetation analysis revealed N. aquatica, U. fibrosa, U. purpurea, and U. unknown are species of spatial interest. Upon completion of the 1993 analysis, comparison of the north and south percent coverage results may be more conclusive.

Hydrology and Water Quality

Pollutant removal efficiencies indicate the system is effective at removing ammonia, NO_2 + NO_3, zinc, T.S.S., and possibly cadmium. Negative efficiencies for phosphorus and iron may be the result of unmeasured groundwater input. Poor removal of organic N is probably caused by the heavy vegetation growth in the marsh.

Indications that the marsh experienced groundwater exchange were: 1) continuous discharge during the wet season; 2) zero discharge during the dry season; 3) more volume leaving than entered the marsh for two months during the wet season; and, 4) negative pollutant load efficiencies (especially iron). It is believed that once groundwater influence has been quantified, the efficiencies will become more positive.

Pollutant removal efficiencies indicate the systems' internal effectiveness. The quality of the water discharged from the system, however, indicates whether the system achieves its ultimate treatment goal - to meet State standards. Average pollutant concentrations (mean of 1991 summer storm events at the outflow) met Class III state water quality standards, with the exception of zinc and cadmium, much of which seem to be entering from rainfall. Zinc and cadmium frequently exceed standards from stormwater management systems (Kehoe, 1992; Rushton and Dye, 1993).

Field Parameter Analysis

The sedimentation basins have much higher levels of pH and dissolved oxygen than the marsh. Since low pH and dissolved oxygen are implicated in the release of phosphorus and some metals from the sediments, these parameters need to be considered in determining the effectiveness of the marsh for stormwater management.

ACKNOWLEDGEMENTS

Special thanks are due to Hidden River Corporate Park for their cooperation and use of the site, Leah Polomchak for her photo interpretations and Steve Saxon for his field work. This research was funded in part by USEPA 205J grant number WM434 administered through the Florida Department of Environmental Regulation.

LITERATURE CITED

Dressler, D. L., D. W. Hall, K. D. Perkins, and N. H. Williams. 1987. Identification Manual for Wetland Plant Species of Florida, University of Florida, Gainesville, FL.

Florida Department of Transportation (D.O.T.). 1985. Florida Land Use, Cover and Forms Classification System.

Godfrey, R. K., and J. W. Wooten. 1979. Aquatic and Wetland Plants of Southeastern United States; Monocotyledons. University of Georgia Press, Athens, GA.

Kehoe, M. J. 1992. Water-quality survey of twenty-four stormwater wet-detention ponds. Environmental Section, Southwest Florida Water Management District, Brooksville, FL.

Kushlan, J. A. 1990. Freshwater Marshes In: Ecosystems of Florida (Ronald L. Myers and John J. Ewel, Editors), University of Central Florida Press, Orlando, FL.

Rushton, B. T. and C. W. Dye. 1993. An In-depth Analysis of a Wet Detention Stormwater System. Environmental Section, Southwest Florida Water Management District, Brooksville, FL.

United States Environmental Protection Agency (USEPA). 1983. Results of the National Urban Runoff Program (NURP) Executive Summary; Volume I - Final Report; Volume II - Appendices; Volume III - Data Appendix. USEPA, Washington D.C.

AN EVALUATION OF THE EFFECTIVENESS OF WETLAND MITIGATION IN BROWARD COUNTY, FLORIDA PERMITTED BY THE FLORIDA DEPARTMENT OF ENVIRONMENTAL REGULATION

Vicki L. Cruse
NOVA UNIVERSITY
Dania, Florida

ABSTRACT

This study examined the effectiveness of the Florida Department of Environmental Protection (FDEP, Formerly Florida Department of Environmental Regulation) in regulating, through mitigation, the wetlands of Broward County, Florida. Utilizing agency permit files, wetland impacts and required mitigation were analyzed for the years 1983-1991. Field evaluations were conducted to determine the actual area of wetland impacts and completion of required mitigation as permitted by FDEP. Results indicate that freshwater wetlands containing both forested and marsh areas were permitted for the largest impact and were impacted the most by project activities. Mangroves comprised the largest area of required mitigation and freshwater forested areas composed the largest area of completed mitigation. There was no trend or consistency in mitigation ratios as analyzed on a yearly basis. The overall potential mitigation ratio for permits issued by FDEP between 1983 and 1991 in Broward County was 2.3:1; however, by excluding preservations and easement areas, the ratio becomes 0.9:1. The actual mitigation ratio for areas with completed project impacts and mitigation was 2:1; by excluding preservation and easement areas, the ratio becomes 0.7:1, indicating a loss of wetland area through the FDEP permitting process. Results of mitigation area evaluations show that permitted wetland impacts were incomplete at 24% of the sites, mitigation was not required in 5% of the permitted projects; mitigation was either incomplete or not attempted at 41% of the proposed mitigation areas; and permit requirements were met at 30% of the mitigation areas.

INTRODUCTION

With the upsurge in the environmental movement of the late 1980's and early 1990's, people are becoming more aware of their environment and the need to protect its resources. Some of the most valuable of these resources are wetlands. Wetlands provide flood and storm damage protection, erosion control, water supply and groundwater recharge, and habitats for many forms of fish and wildlife (Burke, et al, 1989). According to the U.S. Fish and Wildlife Service (Dahl, 1990), Florida once had an estimated 8.2 million ha (20.3 million acres) of wetlands covering approximately 54.2% of the state. Estimates of wetlands acreage in the 1980's reveal that Florida has approximately 4.5 million hectares (11 million acres), a reduction of 46% of the original wetland acreage.

Within the State of Florida a wide variety of agencies have wetland permitting authority. These include the U.S. Army Corps of Engineers (USCOE), the Florida Department of Environmental Protection (FDEP), the water management districts, and local governments. The focus of this study was FDEP. Since 1969, Florida statutes have required a permit or other form of permission from the FDEP, or its predecessor agencies, in order to excavate or fill certain wetlands. Mitigation was not required by the original statute, but mitigation has been required on an informal basis in permit negotiations since the mid-1970's without any guidelines or performance criteria.

The Henderson Wetlands Act (1984) requires the FDEP to consider mitigation proposed by the applicant as the Department decides whether to issue or deny a wetland resource (dredge and fill) permit. Mitigation is defined by FDEP as the creation, enhancement, or preservation of wetlands to offset the loss of wetland functions and values that may be damaged or destroyed by a permitted project.

For purposes of this study, wetland mitigation includes creation, enhancement, and preservation. Creation refers to construction of a wetland in an area that was not wetland in the recent past. Enhancement refers to increasing one or more of the functions of an existing wetland by modifying environmental parameters, but without changing wetland type. Preservation refers to the setting-aside of wetlands in their existing condition. Another commonly used term is restoration, which refers to the reestablishment of a wetland in an area where it historically existed but, which now performs no or few wetland functions. However, FDEP does not use the term "restoration", rather utilizing the term "enhancement" instead.

In order to ensure that wetland losses are adequately compensated, permitting agencies generally set ratios which aid in the determination of the amount of mitigation required to "replace" the wetland area lost. The Florida Department of Environmental Regulation's Chapter 17-312 regulated Dredge and Fill Activities within the State. It is the Department's policy to use a guideline of two hectares (ha) created for each hectare adversely impacted by proposed dredging or filling. According to FDEP, this guideline is for preliminary planning purposes only, and the actual extent of wetland creation may differ depending upon a variety of factors.

MATERIALS AND METHODS

The FDEP, Southeast District Office, in West Palm Beach, Florida was contacted to assist with this study. All complete permit files for projects in Broward County (Figure 1), which had not been archived, were reviewed. According to FDEP personnel, the archived files are those projects which are completely closed and which generally did not require mitigation. A total of 428 files having permits issued between 1983 and 1991 for impacts and mitigation within Broward County, were reviewed. All files with permits requiring mitigation (n = 80) were selected for study.

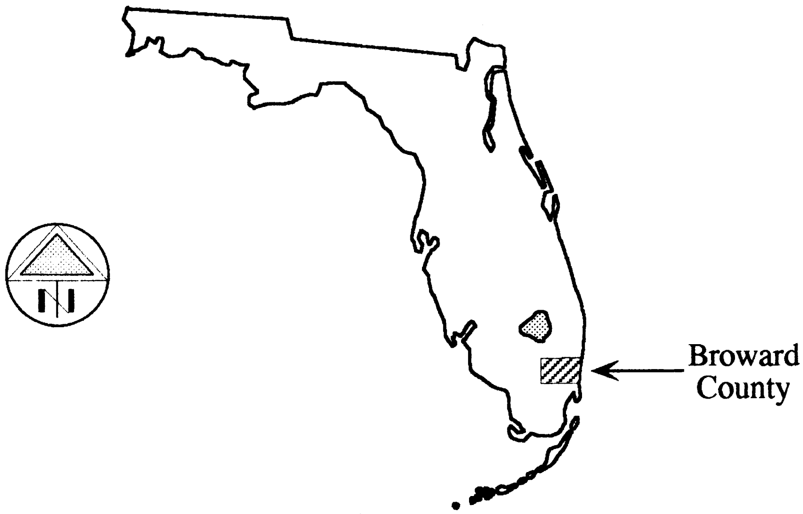


Figure 1: Study area in southeastern Florida.

Upon completion of the Permit File Review, the individual permitted project sites and mitigation areas were visited. The information regarding project and mitigation completion obtained from the file was verified by the field visit. If the mitigation area was attempted or completed, it was analyzed for compliance with the permit conditions at this time.

Preservation areas were field-verified and checked for compliance with the specific conditions included in the permit, if any, but were not evaluated for compliance because they were existing, natural wetlands and were not considered mitigation areas. In addition, one project that was planted one month after this study began was not evaluated. All other mitigation areas were completely evaluated for compliance.

RESULTS

A total of 428 FDER files containing permits issued in Broward Count were reviewed. Eighty permits issued between 1983 and 1991 requiring mitigation were isolated and project information was recorded. From these permits, a total of 82 potential mitigation areas, located in Broward County, were identified.

All of the data regarding impacted and mitigated wetlands has been placed into the following four categories which will be used throughout this discussion:

Potential Impacted Wetland: reflects the total area of wetlands to be impacted by permitted projects.

Actual Impacted Wetland: represents the area of wetlands that was impacted as a result of permitted project activities.

Potential Mitigated Wetland: reflects the total area of wetlands to be created, enhanced, or preserved as a result of permitted projects.

Actual Mitigated Wetland: represents the area of mitigation completed through creation, enhancement, or preservation as a result of permitted project activities.

Impacted and Mitigated Wetland Types

Impacted and mitigated wetland types were obtained during the permit file review from either the permit or the biological assessment conducted by FDEP personnel. A total of nine different wetland types were identified as impacted and eight different types of wetlands were mitigated, including easement and preservation areas. Freshwater wetlands, containing both forested and marsh area, comprised the largest area of wetlands potentially impacted by permitted projects (34.6 ha [85.4 acres]) (Table 1), while mangrove wetlands were the most common type of wetland permitted for potential mitigation (24.5 ha [60.4 acres]) (Table 2).

Freshwater forested and marsh areas were also the most common wetland type that was actually impacted by permitted projects (34.2 ha [84.4 acres]). Freshwater forested wetlands were the most common type of wetland actually created after project impacts were complete (23.8 ha [58.8 acres]). This is in contrast to the most common type of wetland permitted for mitigation-mangrove wetlands.

Figure 2 shows the area of impacted and mitigated wetlands by type of wetland, and is helpful in determining the types fo wetlands impacted and required as mitigation by FDEP. For instance, although freshwater forested wetlands are rarely impacted (in fact the area was so small that it does not show in the graph) this type of mitigation is common. The opposite trend is shown for freshwater forested/marsh wetlands where the area of impact is greater than that of mitigation. By far, the largest area of wetland gain is shown in easements and preservation areas. This is misleading, because one project alone included a preservation area of 76.5 ha (189.0 acres). Areas of littoral zone, seagrass, saltmarsh, and miscellaneous trees are represented as "Other" because their individual areas were too small to be shown on the graph.

Table 1. Area of impacted wetland types in Broward County.

IMPACTED	POTENTIAL		POTENTIAL	ACTUAL		ACTUAL
WETLAND TYPE	ha	acres	MISC.	ha	acres	MISC.
FW* Forested	0.1	0.30	3 trees	0.1	0.3	3 trees
FW Forested/Marsh	34.6	85.4		34.2	84.4	
FW Marsh	9.3	23.0		8.9	21.9	
Littoral Zone	1.9	4.6	1.1 km (3700 ft)	1.8	4.5	
Littoral Zone/Open Water	6.4	15.9		2.5	6.3	
Mangrove	14.4	35.6	142 trees	10.5	26.0	113 trees
Miscellaneous Trees**	-	-	27 trees	-	-	20 trees
Open Water/Mangrove	2.2	5.4		2.2	5.4	
Seagrass	0.01	0.02		0.01	0.02	
Total	68.9	170.2		60.2	148.8	

Table 2. Area of mitigated wetland types in Broward County.

MITIGATED	POTENTIAL		POTENTIAL	ACTUAL		ACTUAL
WETLAND TYPE	ha	acres	MISC.	ha	acres	MISC.
Easement/Preservation	92.5	228.5		80.0	197.6	
FW* Forested	23.8	58.8		23.8	58.8	
FW Forested/Marsh	8.8	21.7	40 trees	0.7	1.7	40 trees
FW Marsh	5.4	13.4		3.4	8.4	
Littoral Zone	2.5	6.1	1.1 km (3700 ft)	0.5	1.3	0.4 km (1250 ft)
Mangrove	24.5	60.4	1040 seedlings	15.1	37.3	591 seedlings
Miscellaneous Trees**	0.1	0.2	269 trees	0.1	0.2	56 trees
Saltmarsh	0.3	0.8		0.1	0.2	
TOTAL	157.9	390.0		123.7	305.5	

The information comes from FDER permit files and field evaluations. Wetland types are expressed in area and/or miscellaneous measurements. Potential refers to data appearing in the FDER permit. Actual refers to the actual area or miscellaneous measurements of completed mitigation.

FW* = Freshwater

** Includes isolated trees such as Annona glabra and Acer rubrum, does not include isolated mangroves.

Impacts and Mitigation By Year

Data from the permit files were analyzed to determine the potential and actual wetland impacts and mitigation for each year between 1983 and 1991. FDEP permits issued in 1986 total the largest potential impact of any year (22.7 ha [56 acres]). This was also the same year which saw the largest actual impact on wetlands. The most mitigation was required in 1987 (29.2 ha [72.2 acres]). 1989 reflects the largest actual mitigation area, but includes the previously mentioned project requiring 76.5 ha (189.0 acres) designated as easement/preservation. Without this project, 1989 is no longer the year with the most required mitigation. Despite the fact that permits issued in 1987 required the most mitigation, the most required mitigation was completed in 1986 (19.2 ha [47.4 acres]).

Figure 3 shows the area of wetland impacted and mitigated, both potential and actual, for each year of permit issuance. Easement and preservation areas, including the previously mentioned project from 1989, have been excluded from this figure to more accurately represent the results. Several important results are shown in this figure. If the potential impacted wetland and actual impacted wetland bars are of equal height (1983-1987), this indicates that all permitted wetland impacts were completed. If the bar representing actual impacted wetland is shorter, then all permitted impacts have not been completed, as shown in 1988-1991.

This comparison similarly holds for mitigated wetlands. If the bars representing potential mitigated wetland and actual mitigated wetland are the same height, then all permitted mitigation has been completed in that particular year (e.g., 1983 and 1984). Generally, however, the bar representing actual mitigated wetland is shorter, meaning that all of the permitted mitigation has not been completed, as shown in years 1985 through 1991.

The relationship between total area permitted for impacts and proposed for mitigation is also evident by comparing the bars representing potential impacted wetland and potential mitigated wetland. If the potential impacted wetland bar is higher than the potential mitigated wetland bar, then a potential loss in wetland area occurred during that year. This is shown in 1985, 1986, 1988, and 1990. A potential gain in wetland area occurs when the potential mitigated wetland bar is higher than the potential impacted wetland bar. This occurred in 1984, 1987, 1989, and 1991. All of the bars are equal in 1983, representing no loss or gain in wetland area.

By comparing the potential area of impacted wetland (70.3 ha [173.5 acres]) to the potential area of mitigation required (157.2 ha [388.8 acres]), there is a gain in wetland area of 89.9 ha (215.5 acres). However, by excluding easement and preservation areas from the total, the potential mitigation required is 64.7 ha (160.3 acres). These results indicate a loss of 5.6 ha (13.2 acres).

Similar results were found when analyzing actual wetland areas, both impacted and mitigated, using the same technique of comparing bars in Figure 3 as discussed above for potential impacts and mitigation. The actual area of impacted wetlands is 61.6 ha (151.8 acres) and the actual area of completed mitigation is 123.6 ha (305.3 acres), resulting in a gain of 62.0 ha (153.5 acres). However, excluding easement and preservation areas yields and actual area of completed mitigation of 43.6 ha (107.7 acres), revealing a loss in wetland area of 18.0 ha (44.1 acres) in Broward County.

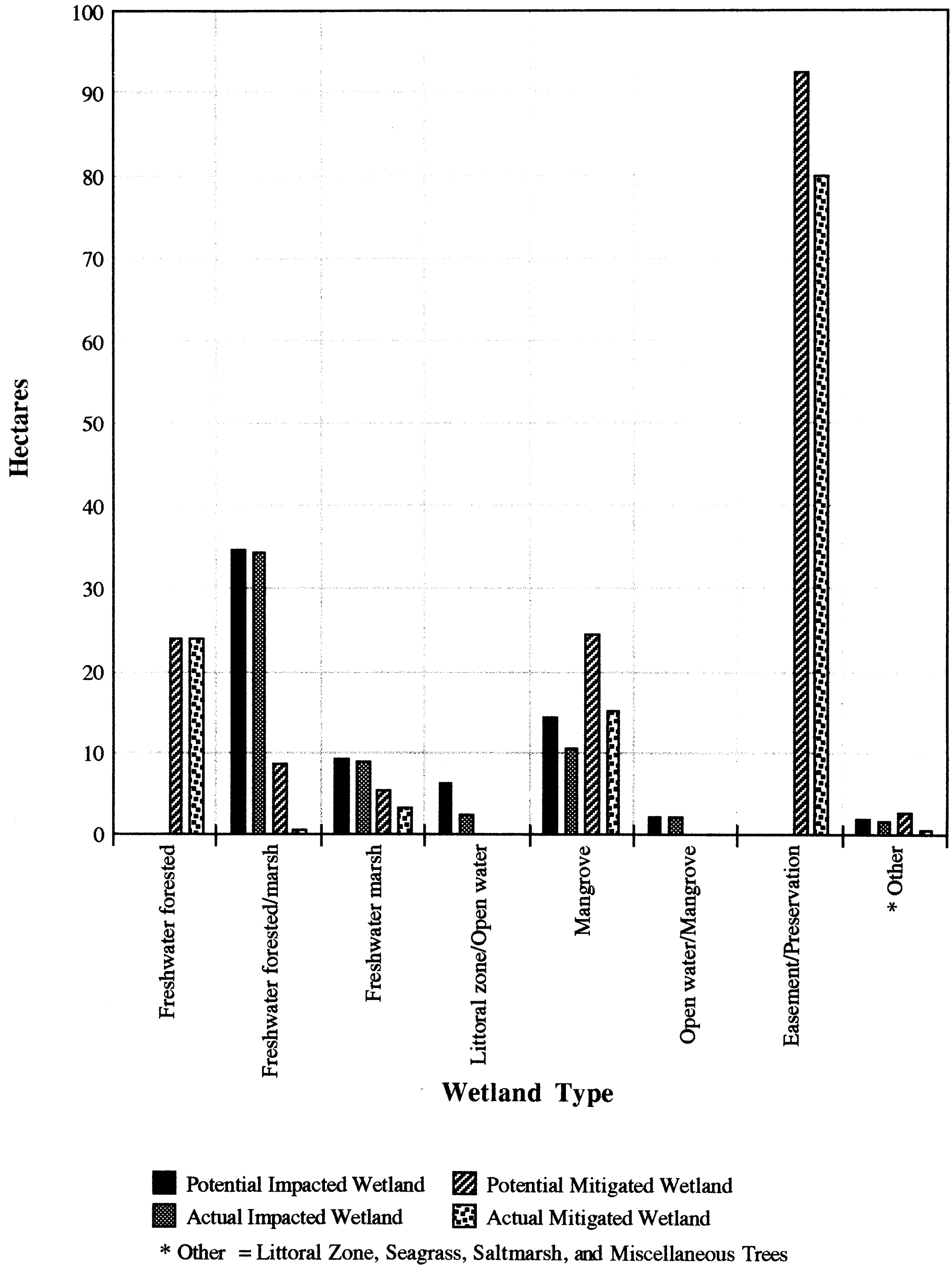


Figure 2. Area of impacted and mitigated wetlands by wetland type in Broward County, Florida 1983 - 1991.

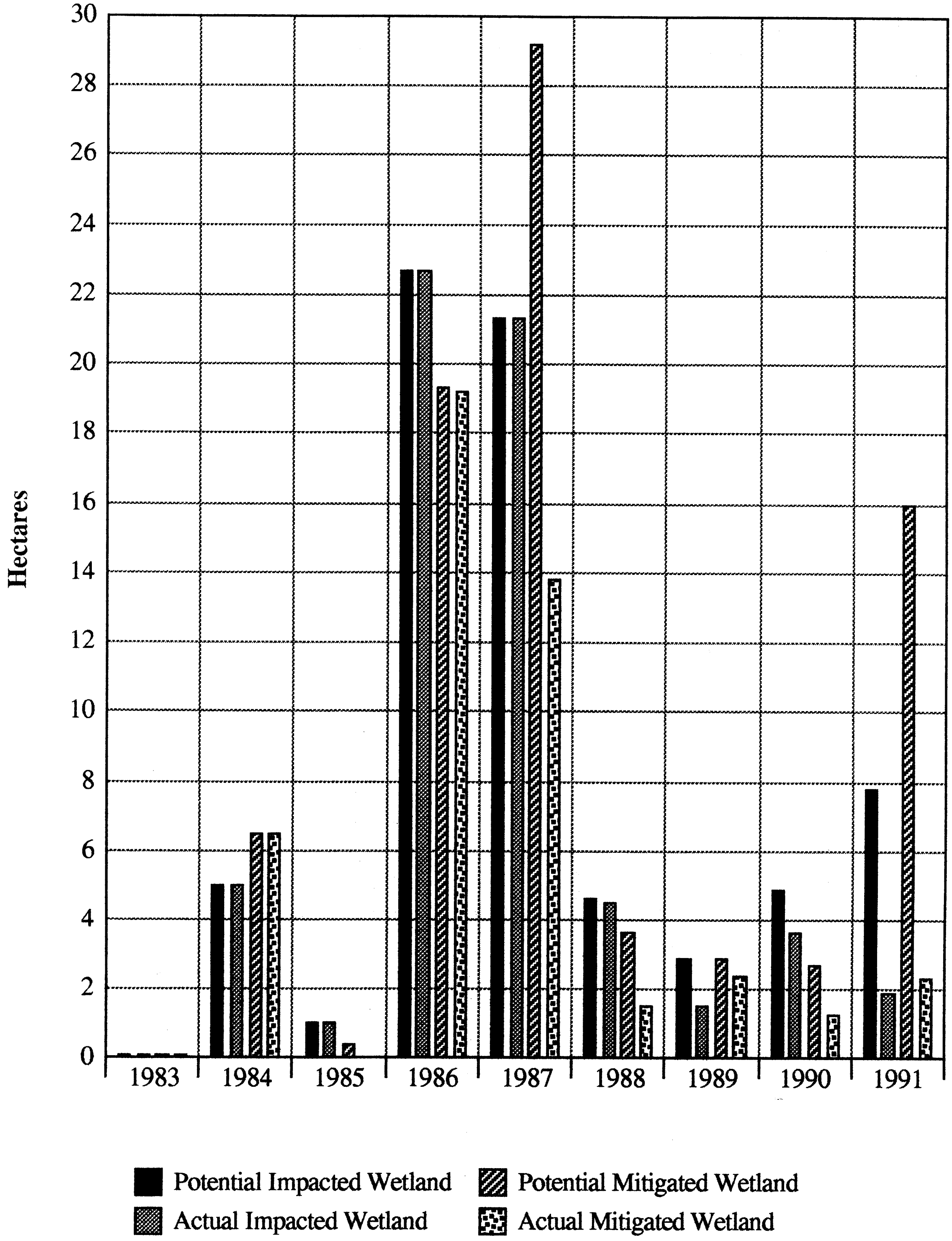


Figure 3. Area of impacted and mitigated wetlands by year in Broward County, Florida 1983 - 1991.

Figure 4 shows the potential and actual gains or losses in wetland area on a yearly basis, excluding easement and preservation areas. Both figures indicate an overall loss in wetland area, but as shown in the data above, the loss is greater in actual wetland area (18.0 ha (44.1 acres)) than in potential wetland area (5.6 ha (13.2 acres)).

Mitigation Ratios

Mitigation ratios were calculated, where possible, for all permitted projects, whether or not the impacts or mitigation was completed. These ratios will be termed potential mitigation ratios because they were calculated using the potential impacted wetland area and the potential mitigated wetland area. Ratios were calculated separately for wetlands represented both in terms of area and numbers of trees. It was not possible to calculated ratios for permitted projects where the wetland impact was represented as an area (ha, for instance) and the mitigation was represented by the number of trees or seedlings required. In addition, ratios could not be calculated for projects where either the wetland impact or required mitigation was zero.

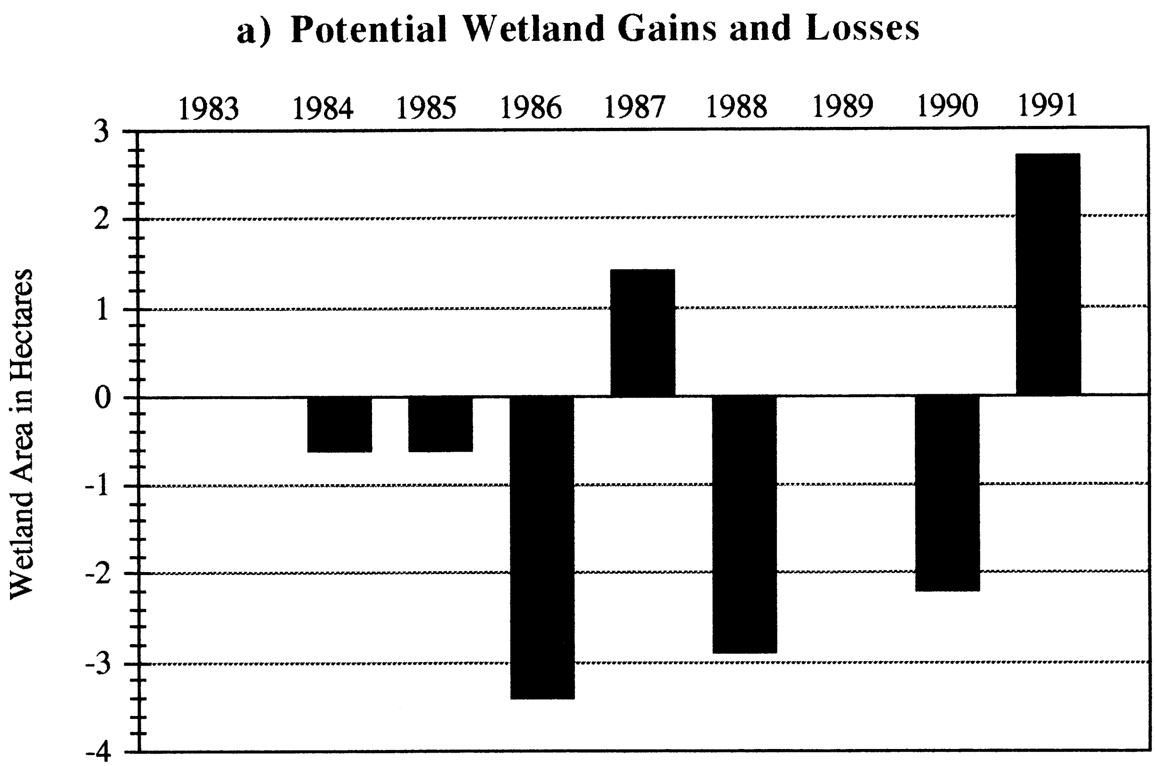
Figure 5 represents the average mitigation ratios calculated for each year of permit issuance. It is evident that there is no consistency or trend in mitigation ratios by year, either in permitted projects based on area or tree numbers. No permits issued in 1983 and 1984 require mitigation in the form of a specific number of trees, therefore, these years are not represented in the lower graph. A permit requiring mitigation in the form of trees was issued in 1985, but the mitigation was required for a project with no wetland impacts, therefore, a ratio could not be calculated.

In addition to mitigation ratios calculated by year of permit issuance, ratios were calculated based on potential impacted and mitigated wetland type. This resulted in the determination of gains and losses in wetland types based on area. Table 3 shows mitigation ratios of impacted and mitigated wetland types for permitted wetlands. Through FDEP permitting, Broward County has gained additional area in freshwater forested, littoral zone, mangrove, and saltmarsh wetlands and has lost area in freshwater forested/marsh, freshwater marsh, and seagrass wetlands. Littoral zone/open water and mangrove/open water wetlands were not included in this comparison because they contain an open water area with an undetermined area of wetland.

By combining all the data, both from FDEP permit files and verification in the field, mitigation ratios for Broward County were calculated. The overall ratio of potential mitigated:impacted wetlands, as reflected in Table 3, in Broward County is 2.3:1 for all permitted projects, whether or not the project impacts or mitigation were completed. By excluding easement and preservation areas, the potential ratio becomes 0.9:1. This is consistent with the data indicating wetland loss previously discussed.

The mitigation ratio was calculated for actual impacts and mitigation, with and without easement and preservation areas. The mitigation ratio of actual mitigated:actual impacted wetlands, including easement and preservation areas is

2:1. When excluding easement and preservation areas, the actual ratio becomes 0.7:1. This result is consistent with that shown for actual wetland area, indicating a loss in wetland area for Broward County, and also indicates that required mitigation is not being completed by all permit applicants.



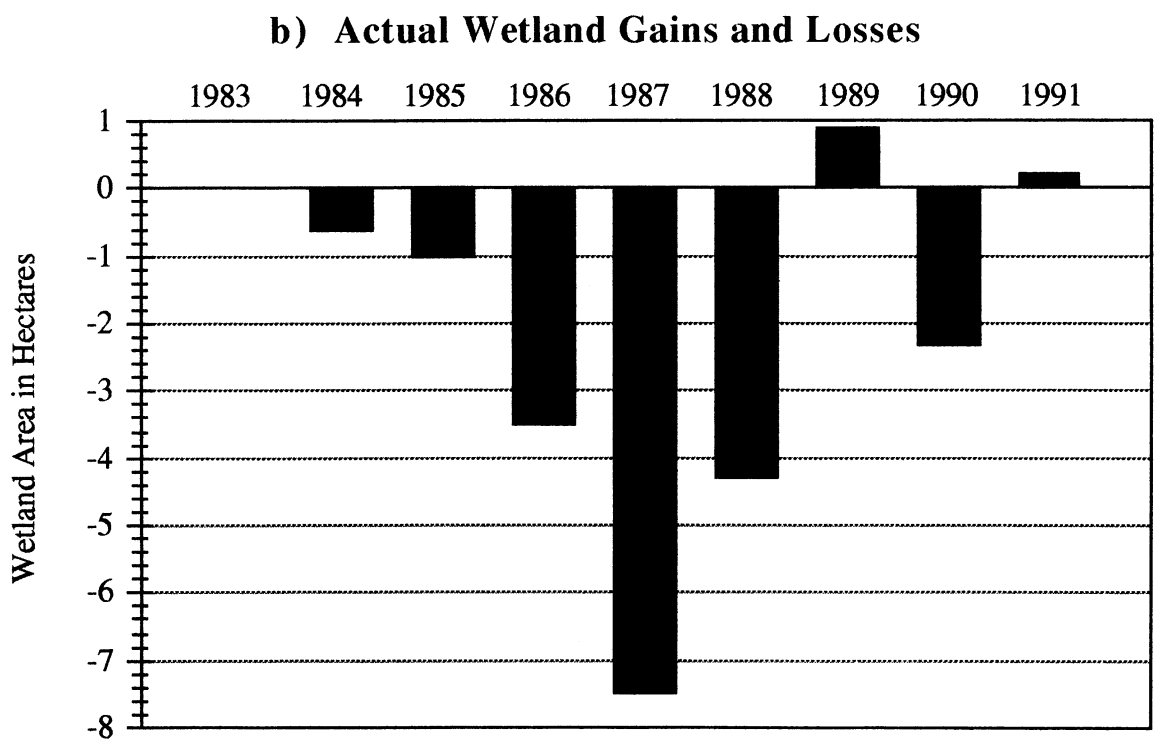
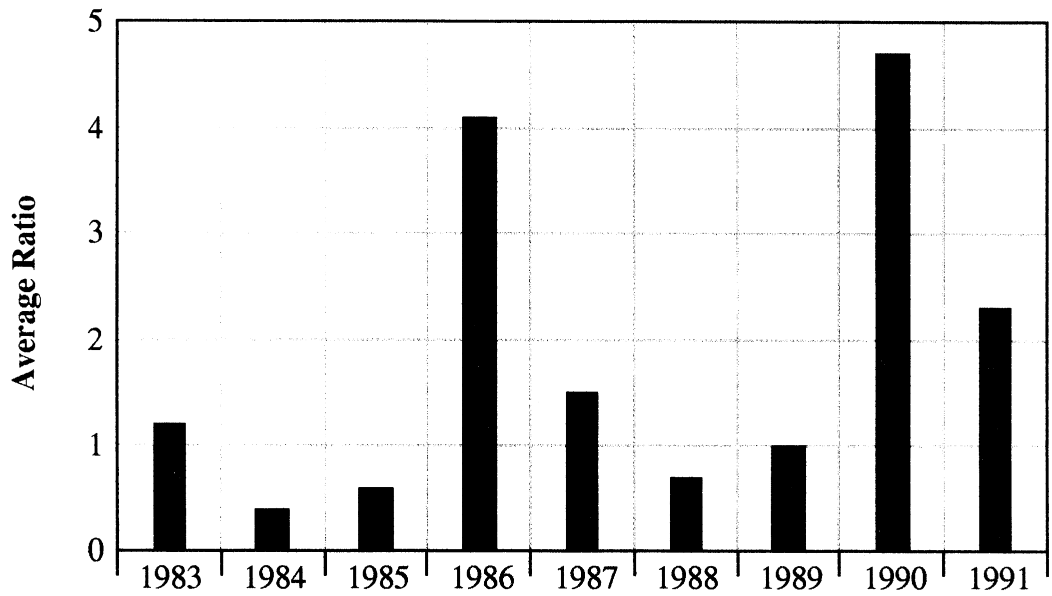


Figure 4. Wetland gains and losses in Broward County, Florida 1983 - 1991.
* Potential losses (a) totaled 5.6 ha while actual losses (b) totaled 18.0 ha.

Impacts and Mitigation Measured as an Area



Impacts and Mitigation Measured in Tree Numbers

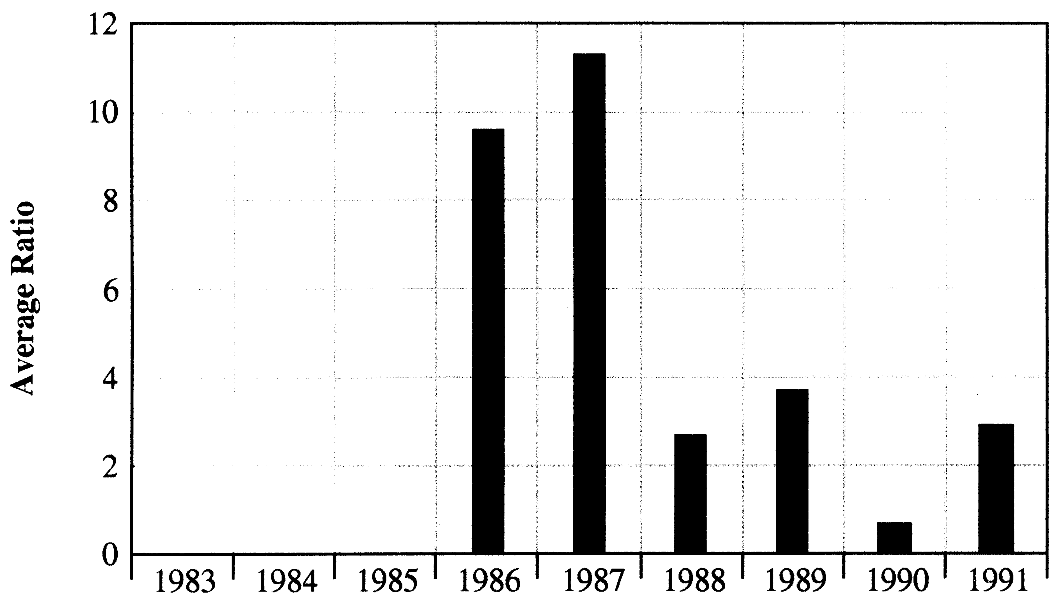


Figure 5. Average potential mitigation ratios in Broward County, Florida 1983 - 1991.

Table 3. Ratios of potential mitigated and impacted wetland types for Broward County, 1983 - 1991.

IMPACTED WETLAND TYPE	POTENTIAL ha	MITIGATED WETLAND TYPE	POTENTIAL ha	MITIGATED: IMPACTED RATIO
FW* Forested	0.1	FW* Forested	23.8	238:1
FW Forested/Marsh	34.6	FW Forested/Marsh	8.8	0.3:1
FW Marsh	9.3	FW Marsh	5.4	0.6:1
Littoral Zone	1.9	Littoral Zone	2.5	1.3:1
Mangrove	14.4	Mangrove	24.5	1.7:1
Miscellaneous Trees**	-	Miscellaneous Trees**	0.1	-
Littoral Zone/Open Water	6.4	Saltmarsh	0.1	-
Open Water/Mangrove	2.2	Easement/Preservation	92.5	-
Seagrass	0.01			-
Total	68.9	Total	157.7	2.3:1

Area and wetland types were obtained from FDER permit files.

FW* = Freshwater

** Includes isolated trees such as Annona glabra and Acer rubrum, does not include isolated mangroves.

Mitigation Status

After all of the mitigation area evaluations were completed, permitted projects were placed into one of six categories. In two instances, a project fit into more than one category. The categories reflecting mitigation status are as follows:

1- Meets Permit Requirements: The mitigation area evaluated meets or exeeds all permit requirements.

2- Meets Requirements, Not Evaluated: The mitigation was verified for completion, but was not evaluated. Represents easement and preserve areas and projects requiring only trees for mitigation.

3- Impacts Not Complete The project impacts for which the permit was obtained have not been completed.

4- Impacts Complete, Mitigation Not Attmepted. The project impacts for which the permit was obtained have been completed but the mitigation required has not been attempted.

5- Impacts Complete, Mitigation Incomplete: The project impacts for which the permit was obtained have been obtained have been completed, but the mitigation required is incomplete and does not meet permit requirements.

6- Mitigation Not Required: A permit was obtained for project impacts, but mitigation was not required.

A total of 80 permits were reviewed resulting in 82 separate mitigation areas. Permitted wetland impacts were incomplete at 20 (24%) sites, mitigation was not required in 4 (5%) of the permitted projects, mitigation was incomplete in 19 (23%) of the mitigation areas, mitigation was not attempted at 15 (18%) of the proposed mitigation areas, and permit requirements were met at 24 (30%) of the mitigation areas. Figure 6 graphically represents the number of permits which fell into each category by year.

Figure 7 reflects the percentage of permitted projects which have completed project impacts and have mitigation areas which meet permit requirements out of a total number of projects and have completed permitted project impacts. With the exception of 1983, in which only one permit was issued and mitigation was completed, the highest percentage of mitigation (67%) that met permit requirements was in 1986. The overall percentage of projects permitted between 1983 and 1991 which have completed project impacts and have mitigation meeting permit requirements is 40%.

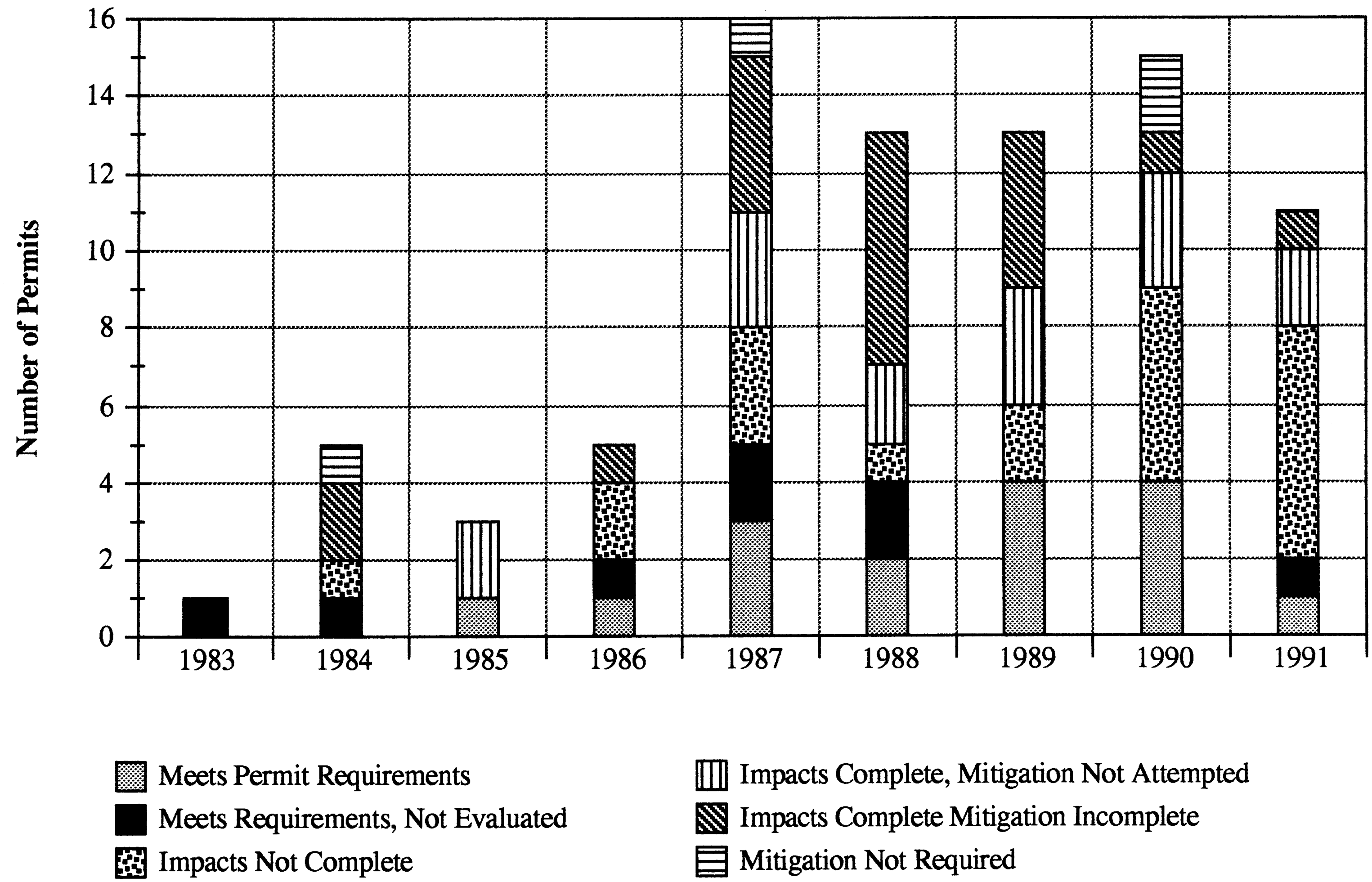


Figure 6. Status of permits requiring mitigation issued by FDER in Broward County, Florida 1983 - 1991.

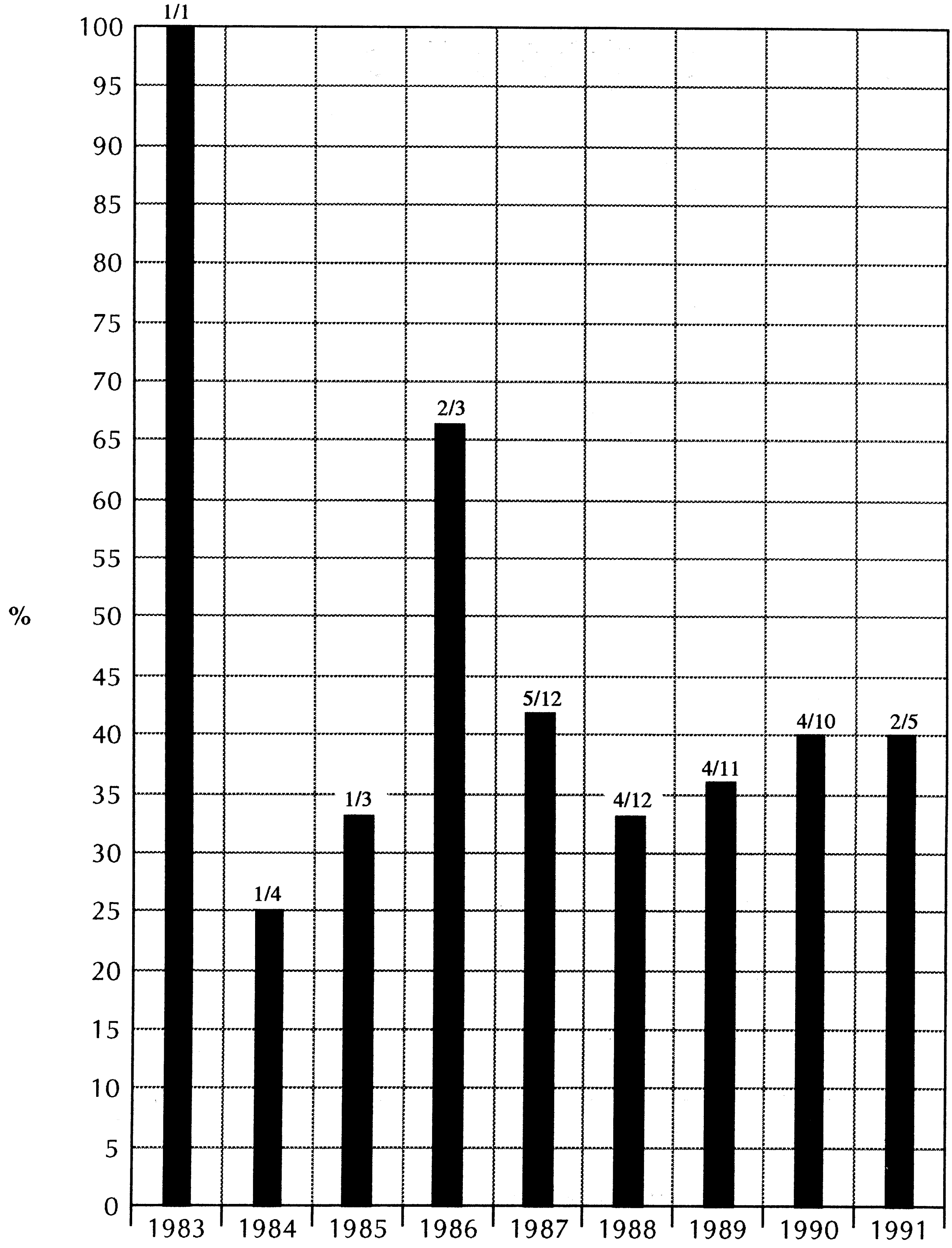


Figure 7. Percent of projects with completed impacts and mitigation meeting permit requirements.

Discussion and Conclusions

The original objectives of the permit file review and field study were to assess the effectiveness of the FDEP in achieving no net loss of wetlands in Broward County, Florida, and to develop and use mitigation area evaluation criteria which could be utilized by FDEP and other permitting agencies in assessing the quality of the mitigation areas.

It must be noted, however, that the two largest migration areas in Broward County were not used in this study. One of the permits was issued for saltwater impacts in 1985 and mitigation, and the other in 1989 for freshwater. The permit for the saltwater project was issued to joint applicants. The jurisdictional and mitigation acreage was distributed between the applicants. A total for the project was never calculated by FDEP. In addition, the applicants were in the process of clarifying the permitted mitigation, as well as modifying the proposed mitigation. The freshwater permit was also a large, complex project. The file correspondence from the applicant combines all jurisdictional wetland, permitted by all agencies (FDEP, ONRP, USCOE), into one total. As a result of the conditions of these projects, the FDEP jurisdictional area and mitigation could not be determined, and therefore, these projects could not be evaluated for this study.

The data analyses utilized for this study involved the calculation of areas and ratios with, and without, the inclusion of easement and preservation area data. These areas do not represent a gain in wetland area for the County because they are jurisdictional wetlands which, most likely, would not be allowed to be impacted by permitting agencies. The permit applicant has chosen to deed (easement) or set aside (preservation) these areas as a portion of the total mitigation requirement. This usually occurs when the applicant has proposed an area of creation or enhancement which does not quite meet the proposed requirement of the permitting agency. The easement and preservations areas may be accepted in lieu of additional creation or enhancement, but are not accepted alone as mitigation.

Wetland Types

By comparing impacted and mitigated wetland types, information can be generated regarding the loss or gain of particular types of wetlands. Broward County shows a loss in freshwater forested/marsh, freshwater marsh, and seagrass wetlands and a gain in littoral zone, freshwater forested, saltmarsh, and mangrove wetlands. This indicates that, as a result of the FDEP permitting process, more impacts are occurring in freshwater forested/marsh, freshwater marsh, and seagrass areas than are being created or enhanced as mitigation. On the other hand, the FDEP permitting process has resulted in a larger area of wetlands through the creation or enhancement of littoral zone, freshwater forested, saltmarsh, and mangrove areas than have been impacted by issued permits.

Mitigation Ratios

It is evident from the data comparing average mitigation ratio by year (Figure

5), that there is no evident trend in this ratio, nor does there appear to be any consistency between years. Other researchers have observed this same inconsistency and expressed their concern (Eliot, 1985, and Kruczynski, 1990). Most permitting agencies have guidelines by which the amount of mitigation required for the proposed project impacts is determined. FDEP has a guideline of 2:1, mitigated wetland area:impacted wetland area. Unfortunately, there appears to be no consistency in the application of this ratio, either between years or types of wetlands (Table 3).

Due to the diverse nature of wetlands, ratios cannot be established which will accommodate every type of wetland and proposed mitigation. In order to achieve more consistency within permitting agencies, those agencies must become more aware of their past permitting activity and the ratios they have established. This past activity would then be used when reviewing applications for new permits.

Dredge and fill permitting by FDER in Broward County, between 1983 and 1991, has resulted in a mitigation ratio of 2.3:1 (157.9:68.9 ha). This exceeds the 2:1 guideline established in FDEP dredge and fill permitting rules (Chapter 17-312). However, with the exclusion of easement and preservation areas from this calculation, the County shows a slight loss in wetland area, with a mitigation ratio of 0.9:1 (65.4:68.9 ha). These ratios were calculated for permitted projects, whether or not the impacts and mitigation were completed.

The mitigation ratios mentioned above were calculated using the potential impacted and mitigated wetland areas, which do not take into account impacts and mitigation not completed. Calculating a ratio which includes only actual impacted and mitigated wetland areas results in a ratio of 2.0:1. Excluding easement and preservation areas, the actual ratio becomes 0.7:1 which indicates Broward County is actually losing wetland area. This is in comparison to the potential ratio of 0.9:1 which indicates that there is a failure of applicants to complete the required mitigation and a failure on the part of FDEP to verify compliance with the permits issued.

Mitigation Status

As a result of verification of permitted project impacts and mitigation evaluations, it was noted that 40% of permitted projects which have completed project impacts, have mitigation which meet or exceed permit requirements. The remaining 60% of the mitigation areas have either not been attempted or do not meet permit requirements. The results provided here (40% of projects meeting permit requirements) exceed those found by FDEP (6%) in its own study (FDEP, 1991). This is not surprising because in some counties FDEP may be either the only permitting agency or the only permitting agency requiring mitigation. The lack of staff may not allow for verifying compliance with the FDEP permits issued, but in areas where mitigation was also required by another agency the mitigation has twice the change of being checked for compliance.

Wetland Functions

One important aspect of the wetland permitting process is the assessment of wetland functions and the application of those functions as goals of the proposed mitigation area. It is the opinion of the author that there is currently no objective, scientific method for assessing the quality and functions of wetlands; therefore, it is difficult to require that specific functions be incorporated into mitigation areas. The analysis undertaken in this study addresses wetland area and does not make a judgement regarding the functions of the wetlands impacted or required for mitigation through FDEP.

As a regulatory agency whose purpose is to regulate wetlands through mitigation, there must be an awareness of wetland function. For instance, during the biological assessment if a coastal wetland area proposed to be impacted is noted as being a nursery habitat for fish, then this same function should be a goal of the proposed mitigation. In addition, assuming the coastal area was of low quality, then the required mitigation may occupy less area than the original wetland. If the mitigation area provides good quality nursery habitat, this may result in a loss of wetland area but an increase in wetland function. This analogy addresses the need for wetland function recognition.

It is not likely that the permits examined for this study took wetland functions into account. The importance of the incorporation of the functions of the impacted wetland into the proposed mitigation areas has only recently become an issue in the permitting agencies. Fortunately, the assessment of wetland functions and the application of those functions as goals of mitigation are being addressed. Since this study began, FDEP in West Palm Beach has begun including a specific condition to their permits which states, "In order to offset the adverse impacts of the permitted project, the created area is expected to perform the following functions: ...". The condition then lists functions noted during the biological assessment or those which would be expected of they type of mitigation proposed. While these proposed functions may lack some objectivity, the awareness of FDEP and its step forward in the regulatory framework is an important one which should allow for more successful and functional mitigation areas.

Recommendations

In order for FDEP, or any wetland permitting agency, to assure the success of their program, they must have the support of the administrators of the agency through resources committed to the wetland permitting effort, including equipment, staff, and salaries which will encourage personnel to stay with the agency instead of moving to private jobs, which generally pay more. Currently, this does not appear to be a priority.

To assure the success of the mitigation required through FDEP's permitting activity, they must have a capable staff. The factor leading to the low percentage of mitigation areas meeting permit requirements is the lack of staff to complete the inspections necessary to assure compliance with the permits issued. It is evident that without this element, mitigation projects are not being completed as required by the permits, or are not being completed at all. Assuring compliance cannot be left to the applicant. There needs to be staff assigned solely to ensuring compliance with those permits.

Compliance staff must, at a minimum, do the following: 1) inspect the project site to ensure that wetland impacts have not exceeded that allowed by the permit; 2) inspect the mitigation area at completion to ensure the area meets the requirements of the permit (e.g., area, number of plants, plant spacing); 3) ensure that applicant or applicant's consultant provides the correct number of mitigation monitoring reports and submits them on a timely basis; and, 4) and field verify the information contained in the monitoring report as well as insuring continuing compliance with the permit requirements.

Most importantly, there must begin to be a recognition of wetland functions. These functions must be determined during the biological assessment by agency personnel and suggested as minimum goals for the proposed mitigation. The essence of wetland mitigation is the attempt to create a "natural" wetland area and, in order to effectively accomplish this, wetland functions must be considered. Wetland permitting has concentrated heavily on the area of impact versus the area of mitigation and, as a result, wetland functions were often implied. In other words, it was almost expected that if the mitigation met the permit requirements, then it functioned like the wetland it replaced. Unfortunately, natural systems do not follow this logic. Therefore, it is essential that an attempt be made to create wetlands which function like, or better than, those being destroyed. FDEP in West Palm Beach is beginning to take this step and it is hoped that the remaining FDEP districts, as well as all wetland permitting agencies, will do the same in order to assure the success of Florida's wetland permitting programs.

LITERATURE CITED

Burke, D. G., E. J. Meyers, R. W. Tiner, Jr., and H. Groman. 1989. Protecting nontidal wetlands. American Planning Association. Planning Advisory Service Report Number 412/413. Chicago, Illinois.

Dahl, T. E. 1990. Wetlands losses in the United States 1780's to 1980's. U.S. Department of the Interior, Fish and Wildlife Services, Washington, D.C.

Eliot, W. 1985. Implementing mitigation policies in San Francisco Bay: Acritique. Prepared for the California State Coastal Conservancy. Oakland, California.

Florida Department of Environmental Regulation (FDER). 1991. Report on the effectiveness of permitted mitigation. Florida Department of Environmental Regulation. Tallahassee, Florida.

Kruczynski, W. L. 1990. Options to be considered in preparation and evaluation of mitigation plans. pp. 555-570. In: J. A. Kusler and M. E. Kentula (Editors), Wetland Creation and Restoration: The Status of the Science, Part 2: Perspectives. Island Press, Washington, D.C.

SHOULD WETLANDS BE CONSTRUCTED IN STORMWATER TREATMENT PONDS?

Kym Rouse Demora
ST. JOHNS RIVER WATER MANAGEMENT DISTRICT
618 E. South Street, Orlando, FL 32801
and Department of Biology
UNIVERSITY OF CENTRAL FLORIDA
Orlando, FL 32816

ABSTRACT

Wetlands often are created in stormwater ponds in Central Florida; sometimes the littoral zones of the ponds are planted to create habitat for fish and wildlife to compensate for the loss of wetland habitat as a result of land development. Studies have shown that urban runoff, that enters stormwater ponds, contains a significant amount of heavy metals and the heavy metals concentrate in the sediments of stormwater ponds. Fish that inhabit stormwater ponds serve as a food source to wading birds and other wildlife. This research was designed to determine if the fish that live in stormwater treatment ponds, constructed in the Orlando area, bioaccumulate significant concentrations of heavy metals, and if there were differences in heavy metal concentrations between species with different foraging strategies. Redear sunfish (Lepomis microlophus), largemouth bass (Micropterus salmoides), and bluegill sunfish (Lepomis macrochirus) were collected from stormwater ponds and natural lakes and ponds in the Greater Orlando area and analyzed for cadmium, nickel, copper, lead, and zinc. Redear sunfish, a bottom feeder, collected from the stormwater ponds, contained significant ($p<0.005$) concentrations of cadmium, nickel, copper, lead, and zinc. Largemouth bass, a predator that occupies a high position on the food chain, collected from the stormwater ponds, contained significant ($p<0.005$) concentrations of copper. The results of this study suggest that the fish that live in stormwater ponds represent a potentially contaminated food source.

INTRODUCTION

Permits must be obtained from the St. Johns River Water Management District for land development, which include the required stormwater management systems that are designed to meet certain criteria. Stormwater ponds provide a benefit to water quality by removing heavy metals and nutrients from the water before it leaves the project area, and reduce the off-site discharge to relieve downstream flooding. The District requires that many wet-detention stormwater ponds be vegetated for the biological treatment of stormwater. Also, the District commonly approves plans that create habitat for fish and wildlife by planting desirable wetland and aquatic vegetation in the littoral zones of stormwater ponds to compensate for the loss of wetland habitat as a result of land development (a process known as mitigation).

Wilbur and Hunter (1979) and Owe, et. al. (1982) have shown that very large quantities of heavy metals are found in urban runoff. Heavy metal sources are largely associated with the operation of motor vehicles, atmospheric fallout, and road surface materials (Harper, 1985). Some sources of heavy metals are displayed in Table 1.

Table 1. Sources of heavy metals found in stormwater runoff.

SOURCE	Cd	Ni	Cu	Pb	Zn
Gasoline	XX		XX	XX	XX
Exhaust Emissions		XX		XX	
Motor Oil and Grease	XX	XX	XX	XX	XX
Antifreeze			XX		XX
Undercoating				XX	XX
Brake Linings		XX	XX	XX	XX
Rubber	XX		XX	XX	XX
Asphalt		XX	XX		XX
Concrete			XX	XX	XX
Diesel Oil	XX				
Engine Wear			XX		

Wigington, et al., 1983; Harper, 1985; Whalen and Cullum, 1988; and Harper, 1990. Cd=Cadmium; Ni=Nickel; Cu=Copper; Pb=Lead; Zn=Zinc.

Several investigations have been conducted to determine the fate of pollutants within stormwater treatment pond systems (Nightingale, 1975; Wigington, et al., 1983; Harper, 1985; Yousef, et al., 1985; Nightingale, 1987). The results of these studies suggest that heavy metals concentrate in the sediment of stormwater ponds. Sediments represent the most concentrated physical pool of metals in aquatic environments, and are ingested by many types of aquatic organisms (Luoma, 1983).

This research was designed to determine if the fish that live in stormwater treatment ponds, constructed in the Orlando area, bioaccumulate significant concentrations of cadmium, nickel, copper, lead, and zinc. Bioconcentration refers to that process whereby chemical substances enter aquatic organisms through the gills or epithelial tissue directly from the water; bioaccumulation is a broader term referring to a process, which includes bioconcentration, but also any uptake of chemical residues from dietary sources (Macek, et al., 1979). The various species of fish that inhabit stormwater ponds serve as a food source to wildlife, especially wading birds. In order to determine if there are differences in heavy metal concentrations in fish with different foraging strategies, three species of centrarchids, each with substantially different foraging strategies, were selected for this study: redear sunfish, Lepomis microlophus; largemouth bass, Micropterus salmoides; and bluegill sunfish, Lepomis macrochirus.

The redear sunfish depends largely on mollusks for food; it has highly developed grinding teeth located in its throat, which are capable of crushing snails (McClane, 1978). Redear sunfish in Florida eat midge larvae, snails, scuds, prawns, and mayfly and dragonfly naiads (Wilbur, 1969). Upon finding a snail on the bottom, the redear assumes a vertical position and literally dives into the sediment, head first (Wilbur, 1969).

The food of the young largemouth bass consists of tiny crustaceans. Larger bass eat insects, crayfish, frogs, and fishes (McClane, 1978). Bass in ponds without other fish rely on crayfish, frogs, large insects, and young bass (Carlander, 1969).

The food of the bluegill sunfish consists of insects and some vegetation (McClane, 1978). Young bluegill feed on smaller crustaceans and aquatic insects (Carlander, 1969).

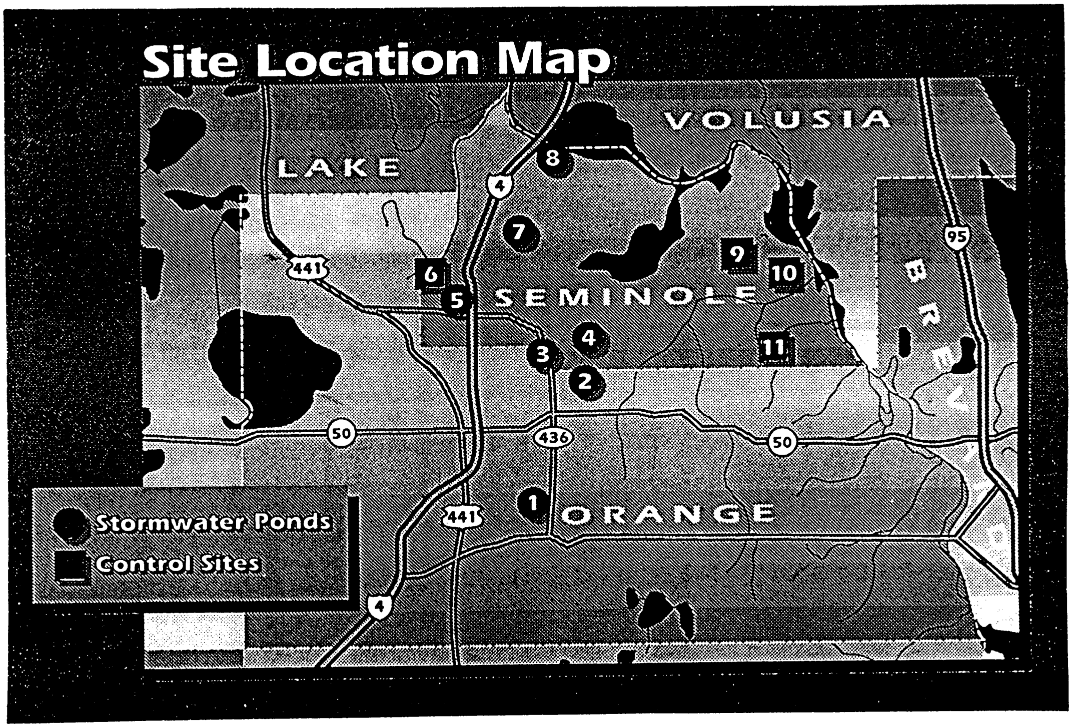


Figure 1. Location map of stormwater pond sites and control sites where fish were collected for heavy metal analysis, December, 1991-March, 1992, Orlando, Florida.

STUDY SITE

Fish were collected from seven stormwater ponds and four natural control sites, located in the Greater Orlando area (Figure 1). Three species of fish, with different foraging strategies, were collected from the stormwater ponds and control sites: largemouth bass (Micropterus salmoides), redear sunfish (Lepomis microlophus), and bluegill sunfish (Lepomis macrochirus). Of the seven stormwater ponds selected, three serve shopping centers, two serve apartment complexes, and two serve road projects.

Stormwater ponds selected for study met the following criteria: the stormwater pond is designed such that the pond always contains enough water to support a fish population; the project is a shopping center, an apartment complex, or a road; the project was built between 1986 and 1988; seining of the stormwater pond was possible; and, wading birds had been observed feeding in the pond. Control sites selected met the following criteria: the pond or lake did not receive any urban or road runoff; the pond or lake was accessible; and permission to access it had been obtained from the land owner.

MATERIALS AND METHODS

Fish were collected during the fall of 1991 and the winter of 1992. Fish were collected from the stormwater ponds using a 1.83 m (6-ft) deep by 16.75 m (55-ft) long seine with a .953-cm (3/8-in) mesh size and a 4 foot deep by 16.75 m (55-ft) long seine with a .635-cm (1/4-in) mesh size. A 1.83 m (6-ft) deep by 45.7 m (150-ft) long gill net with a 3.81-cm (1-1/2 in) mesh was used to collect some of the bass from the three stormwater ponds containing them. Five largemouth bass were collected from each of the three stormwater ponds containing them; five bluegill sunfish were collected from each of the three ponds containing them; and five redear sunfish were collected from each of the three ponds containing them. A total of 15 individuals of each of the three species was collected from the stormwater ponds; and, forty-five fish were collected from the seven stormwater ponds. Redear sunfish were collected from one of the control sites with the 1.83 m (6-ft) by 16.75m (55-ft) seine. Fish were collected from three of the control sites with an electrofishing boat. A total of 15 individuals of each of the three species (45 fish total) were collected from the control sites. As much as it was possible, the individuals of each species collected from both the stormwater ponds and the control sites were of similar size. Fish considered to be in the small-size range for each species were collected, small enough to be eaten by wading birds.

Upon collection, each fish was measured, tagged, and placed in a Nasco Whirl-Pack bag. The fish were placed on ice and later frozen until they were taken to the laboratory for analysis.

A composite sediment sample was collected from each of the seven stormwater ponds and four control sites on, or near, the date that the fish were collected. A sediment sample was collected from three different locations in each pond with an Ekman dredge. Each composite sediment sample was placed in a Nasco Whirl-Pak bag, and kept on ice until it was taken to the laboratory for analysis.

The fish (90 individuals) and the composite sediment samples (11 samples) were taken to Flowers Chemical Laboratories, Inc. (481 Newburyport, Altamonte Springs, Florida) for heavy metal analysis. At the laboratory, each fish was weighed to the nearest 0.1 of a gram on a triple-beam balance (Triple Beam Balance, 700 Series, OHAUS).

Each fish sample was pureed in a Waring blender. A subsample (0.2 to 0.5 grams) of each pureed fish sample was used for the microwave digestion procedure. Each sample was digested according to EPA Method 3051. The concentration of cadmium was determined for each sample using the Atomic Absorption Direct Aspiration Method, EPA Method 7130. The concentration of nickel was determined for each sample using the Atomic Absorption Direct Aspiration Method, EPA Method 7520. The concentration of copper was determined for each sample using the Atomic Absorption Direct Aspiration Method, EPA Method 7210. The concentration of lead was determined for each sample using the Atomic Absorption Furnace Technique, EPA Method 7421. The concentration of zinc was determined for each sample using the Atomic Absorption Direct Aspiration Method, EPA Method 7950. A representative 0.5 grams of each sediment sample was digested, as described above. Each sediment sample was analyzed for cadmium, nickel, copper, lead, and zinc, using the methods described above.

SAS (SAS Institute, 1985) was used on an IBM 4381 mainframe computer to analyze the data. The data was analyzed by using an analysis of variance (ANOVA). The Bonferroni Multiple Comparisons Procedure was used to compare the means. The level of significance was set at $p<0.005$.

RESULTS

Cadmium

Redear sunfish collected from the stormwater ponds contain a mean cadmium concentration that is significantly higher ($p<0.005$) than the redear collected from the control sites (Figure 2). Largemouth bass collected from the stormwater ponds contained a mean cadmium concentration that was significantly higher ($p<0.005$) than the bass collected from the control sites (Figure 2). The largemouth bass collected from the stormwater ponds contained the highest mean cadmium concentration (Figure 2). The mean cadmium concentration in the bluegill sunfish collected from the stormwater ponds was higher than the bluegill collected from the control sites; this was not demonstrated well in Figure 2, because the values are so small. The mean cadmium concentrations were significantly different ($p<0.005$) for the three species of fish. The mean cadmium concentration of the composite sediment samples collected from the stormwater ponds was higher than the composite sediment samples collected from the control sites (Figure 10).

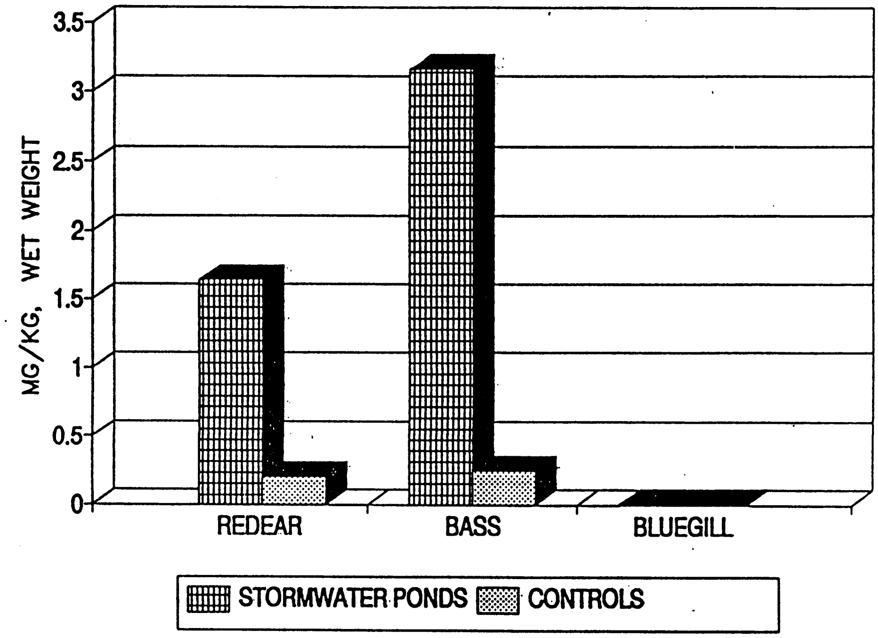


Figure 2. The mean cadmium concentrations of the redear sunfish, the largemouth bass, and the bluegill sunfish collected from the stormwater ponds and the control sites, December, 1991-March, 1992, Orlando, Florida.

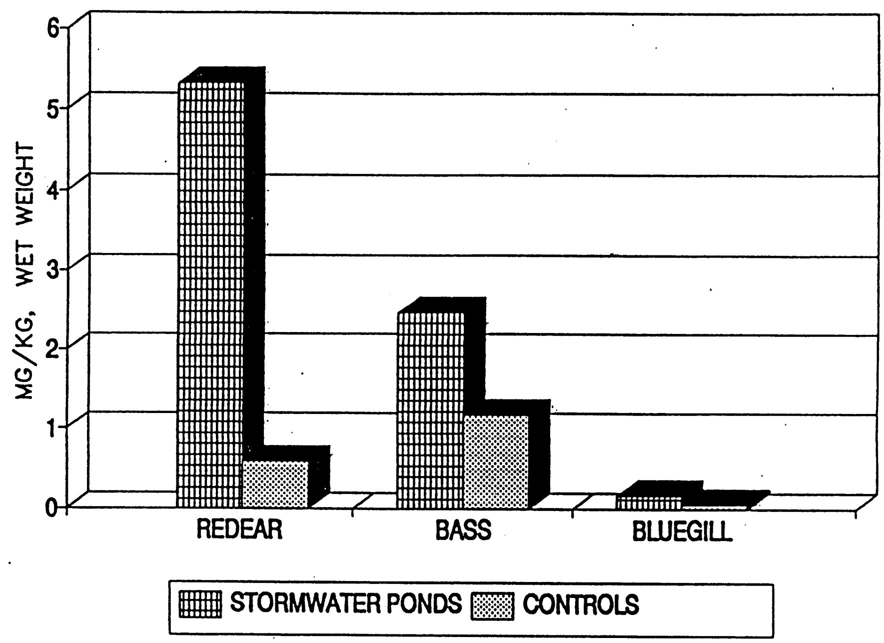


Figure 3. The mean nickel concentrations of the redear sunfish, the largemouth bass, and the bluegill sunfish collected from the stormwater ponds and the control sites, December 1991-March 1992, Orlando, Florida.

Nickel

The mean nickel concentration of the redear sunfish collected from the stormwater ponds is significantly higher ($p<0.005$) than the redear collected from the control sites (Figure 3). The redear sunfish collected from the stormwater ponds contain the highest mean nickel concentration (Figure 3). Largemouth bass and bluegill sunfish collected from the stormwater ponds contain a higher mean nickel concentration than the bass and bluegill collected from the control site (Figure 3). The composite sediment samples collected from the stormwater ponds contain a mean nickel concentration approximately 20 times higher than the composite sediment samples collected from the control sites (Figure 10).

Copper

Redear sunfish collected from the stormwater ponds contain a mean copper concentration that is significantly higher ($p<0.005$) than the redear collected from the control sites (Figure 4). The mean copper concentration of the largemouth bass collected from the control sites is slightly higher than the bass collected from the stormwater ponds (Figure 4). The mean copper concentration of the bluegill sunfish collected from the stormwater ponds is significantly higher ($p<0.005$) than the bluegill collected from the control sites (Figure 4). The redear sunfish collected from the stormwater ponds contain the highest mean copper concentration (Figure 4). The mean copper concentration of the composite sediment samples collected from the stormwater ponds is higher (approximately 4 times) than the composite sediment samples collected from the control sites (Figure 10).

Lead

The mean lead concentration of the redear sunfish collected from the stormwater ponds is significantly higher ($p<0.005$) than the redear collected from the control sites (Figure 5). Largemouth bass collected from the stormwater ponds contain a higher mean lead concentration than the bass collected from the control sites (Figure 5). The mean lead concentration is the bluegill sunfish collected from the stormwater ponds is higher than the bluegill collected from the control sites (Figure 5). The redear sunfish collected from the stormwater ponds contain the highest mean lead concentration (Figure 5). The mean lead concentration of the bluegill sunfish is significantly different ($p<0.005$) from both the redear and the bass. The mean lead concentration of the composite sediment samples collected from the control sites is approximately 2 times higher than the composite sediment samples collected from the stormwater ponds (Figure 10).

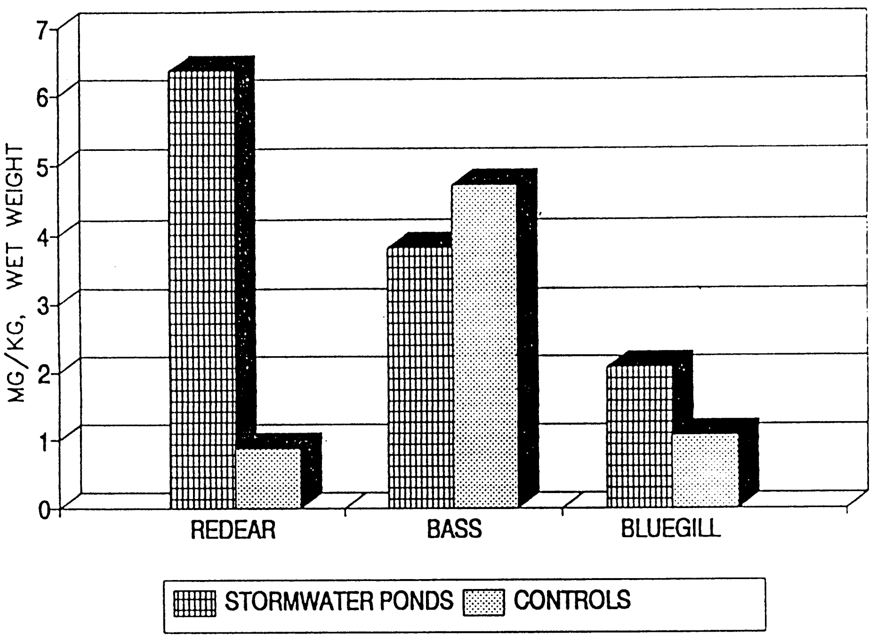


Figure 4. The mean copper concentrations of the redear sunfish, the largemouth bass, and the bluegill sunfish collected from the stormwater ponds and the control sites, December, 1991 - March, 1992, Orlando, Florida.

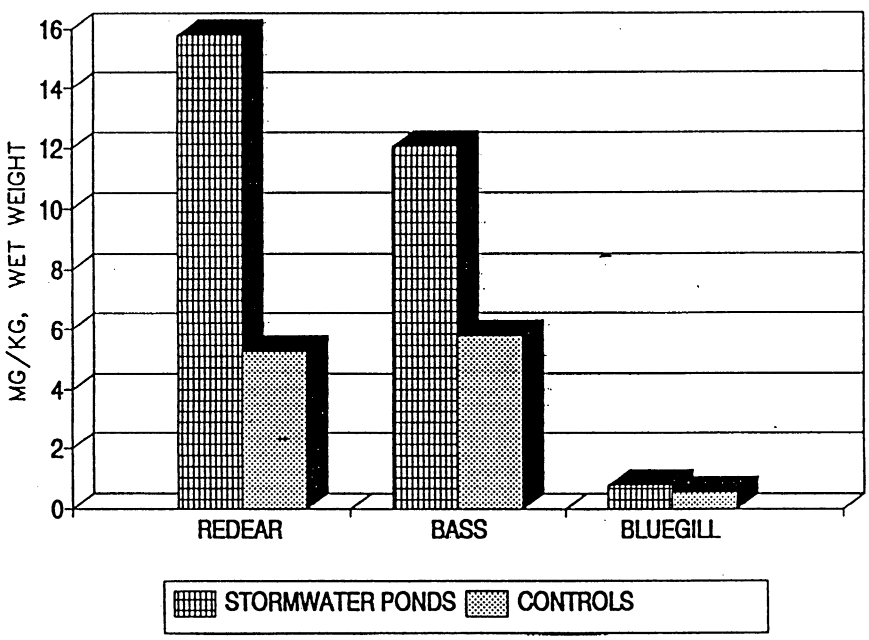


Figure 5. The mean lead concentrations of the redear sunfish, the largemouth bass, and the bluegill sunfish collected from the stormwater ponds and the control sites, December, 1991 - March, 1992, Orlando, Florida.

Zinc

Redear sunfish collected from the stormwater ponds contain a mean zinc concentration that is significantly higher ($p<0.005$) than the redear collected from the control sites (Figure 6). The redear sunfish collected from the stormwater ponds contain the highest mean zinc concentration. The mean zinc concentration of the largemouth bass collected from the stormwater ponds is significantly higher ($p<0.005$) than the bass collected from the control sites (Figure 6). Bluegill sunfish collected from the stormwater ponds contain a mean zinc concentration that is higher than the bluegill collected from the control sites (Figure 6). The mean zinc concentration of the composite sediment samples collected from the stormwater ponds is approximately three times higher than the composite sediment samples collected from the control sites (Figure 10).

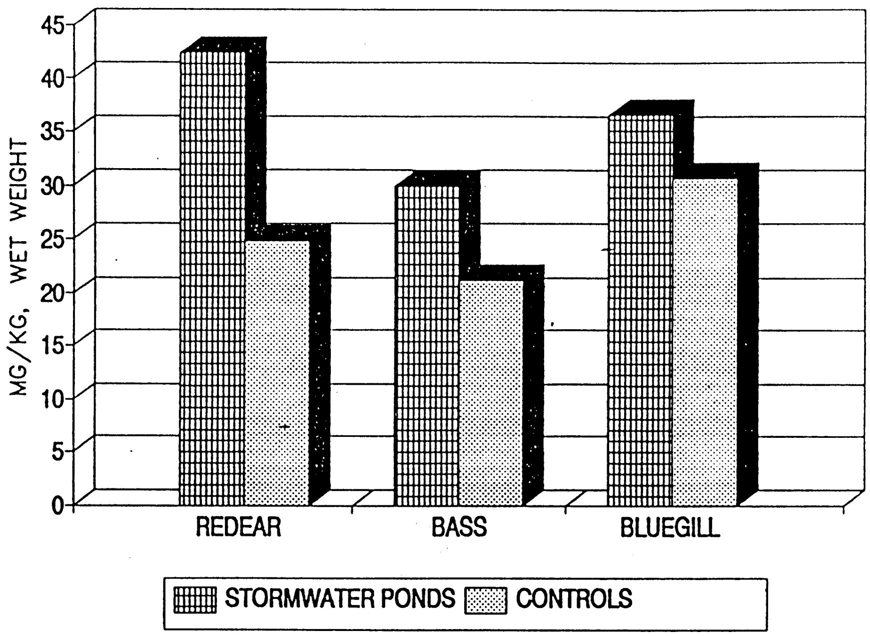


Figure 6. The mean zinc concentrations of the redear sunfish, the largemouth bass, and the bluegill sunfish collected from the stormwater ponds and the control sites, December, 1991 - March, 1992, Orlando, Florida.

The mean cadmium concentration is higher in the redear sunfish, the largemouth bass, the bluegill sunfish, and the composite sediment samples collected from the stormwater ponds (Figures 7, 8, 9, and 10). The redear sunfish, the largemouth bass, the bluegill sunfish, and the composite sediment samples collected from the stormwater ponds contain a higher mean nickel concentration than the redear, the bass, the bluegill, and the composite sediment samples collected from the control sites (Figures 7, 8, 9, and 10). The mean copper concentration is higher in the redear sunfish, the bluegill sunfish, and the composite sediment samples collected from the stormwater ponds than the redear, the bluegill, and the composite sediment samples collected from the control sites (Figures 7, 9, and 10). The largemouth bass collected from the control sites contain a slightly higher mean copper concentration than the bass collected from the stormwater ponds (Figure 8). The mean lead concentration is higher in the redear sunfish, the largemouth bass, and the bluegill sunfish collected from the stormwater ponds (Figures 7, 8, and 9). The composites sediment samples collected from the control sites contain a higher mean lead concentration than the composite sediment samples collected from the stormwater ponds (Figure 10). The redear sunfish, the largemouth bass, the bluegill sunfish, and the composite sediment samples collected from the stormwater ponds contain a higher mean zinc concentration than the redear, the bass, the bluegill, and the composite sediment samples collected from the control sites (Figures 7, 8, 9, and 10).

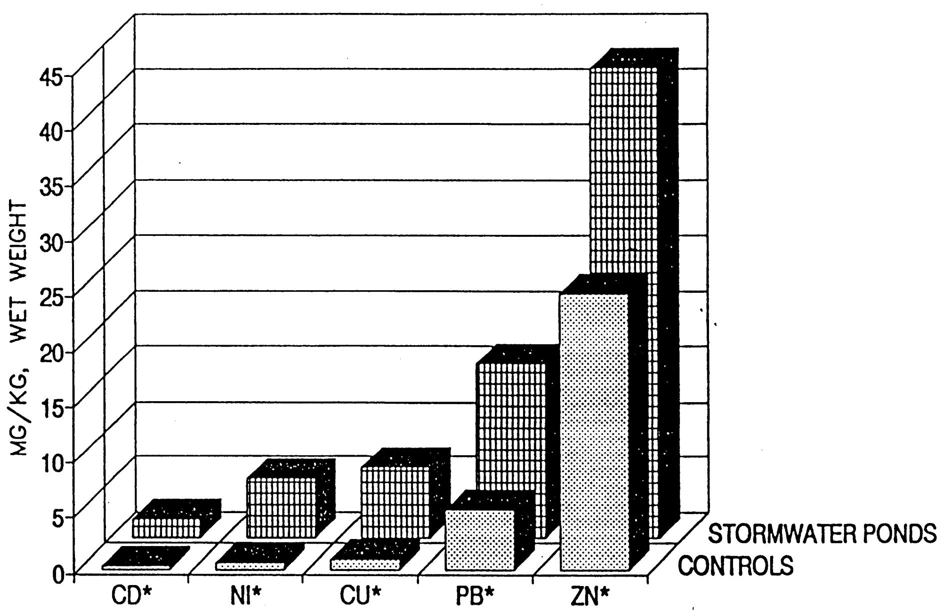


Figure 7. The mean cadmium, nickel, copper, lead, and zinc concentrations of the redear sunfish collected from the stormwater ponds and the control sites, December, 1991 - March, 1992, Orlando, Florida (asterisks indicate significantly higher concentrations from the stormwater ponds at $p<0.005$).

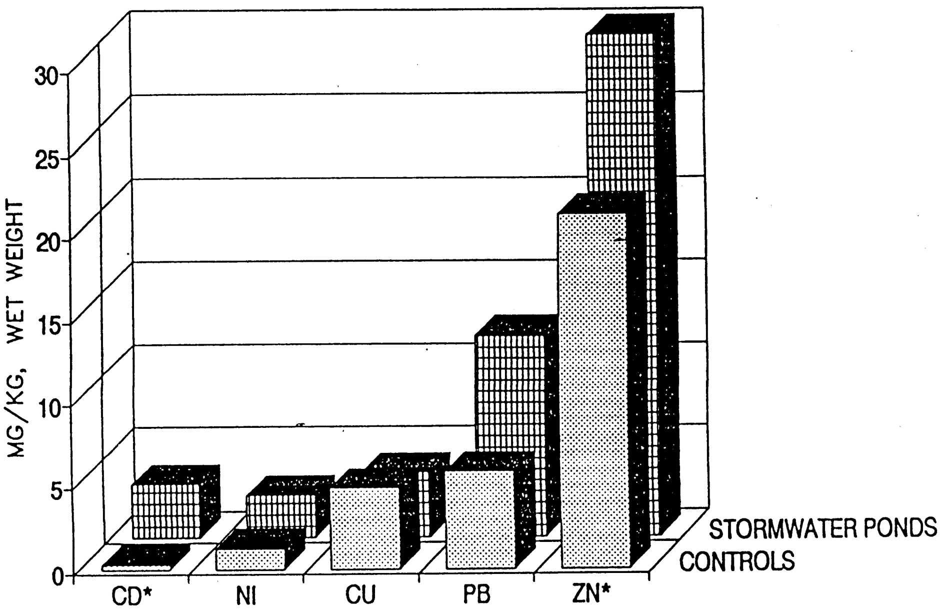


Figure 8. The mean cadmium, nickel, copper, lead, and zinc concentrations of the largemouth bass collected from the stormwater ponds and the control sites, December, 1991 - March, 1992, Orlando, Florida (asterisks indicate significantly higher concentrations from the stormwater ponds at a $p<0.005$)

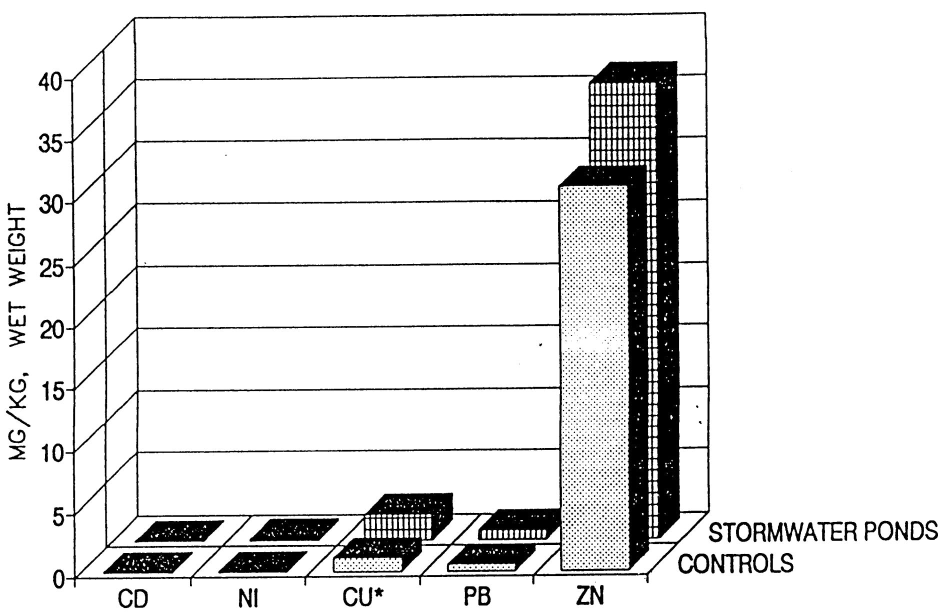


Figure 9. The mean cadmium, nickel, copper, lead and zinc concentrations of the bluegill sunfish collected from the stormwater ponds and the control sites, December, 1991 - March, 1992, Orlando, Florida (asterisk indicates significantly higher concentrations from the stormwater ponds at a $p>0.005$).

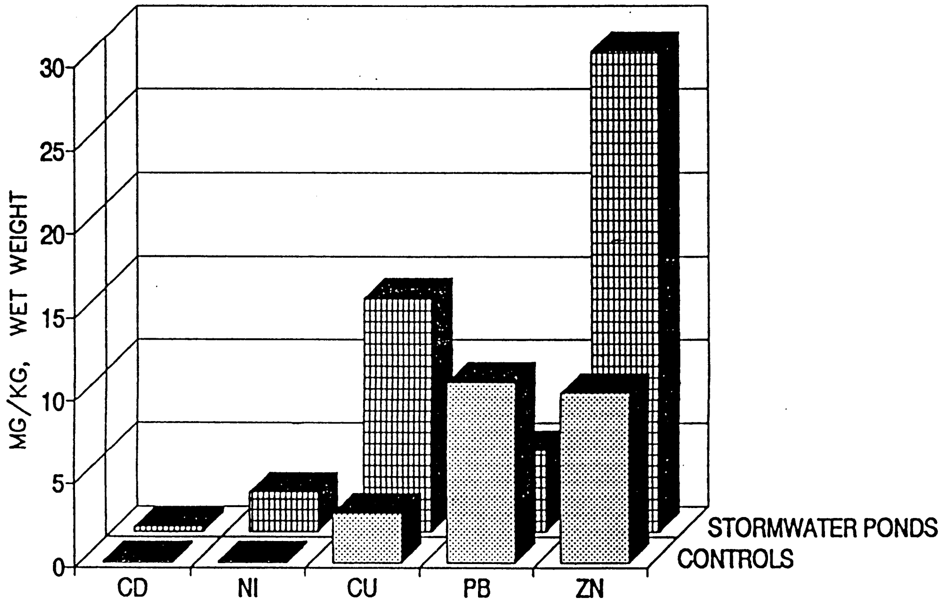


Figure 10. The mean cadmium, nickel, copper, lead, and zinc concentrations of the composite sediment samples collected from the stormwater ponds and the control sites, December, 1991 - March, 1992, Orlando, Florida.

The redear sunfish, collected from the stormwater ponds, bioaccumulated significant ($p<0.005$) concentrations of cadmium, nickel, copper, lead, and zinc. Largemouth bass, collected from the stormwater ponds, bioaccumulated significant ($p<0.005$) concentrations of cadmium and zinc. Bluegill sunfish, collected from the stormwater ponds, bioaccumulated significant ($p<0.005$) concentrations of copper.

DISCUSSION

Cadmium

The mean cadmium (a non-essential metal) concentration of the redear sunfish collected from the stormwater ponds is similar to values for redear collected from a contaminated system in a similar study (Murphy, et al., 1978). Redear collected from the control sites contain a higher mean cadmium concentration than values obtained for redear from an uncontaminated system (Murphy, et al., 1978). The uncontaminated system used in the other study appears to be more pristine or unpolluted than the controls used in this research. The mean cadmium concentration of the largemouth bass collected from the stormwater ponds (3.16 mg/kg, wet weight) is much higher than values obtained in similar studies (Murphy, et al., 1978; Sorensen, 1991). Largemouth bass collected from the control sites have a mean cadmium concentration similar to values obtained from a contaminated system in a similar study (Murphy, et al., 1978). The mean cadmium concentration of the bluegill collected from the stormwater ponds is lower than the range of values obtained for bluegill in similar studies (Atchison, et al., 1977; Giesy and Wiener, 1977; Murphy, et al., 1978). Bluegill collected from the control sites in this study contain a lower mean cadmium concentration than values obtained for bluegill from an uncontaminated system (Murphy, et al., 1978; Atchison, et al., 1977; Giesy and Wiener, 1977).

In similar studies, the degree of association of fish with sediments seems to play a critical role in cadmium accumulation. A study by Murphy, et al., (1978), determined that diet influenced cadmium accumulation in fish. Comparison of cadmium levels in redear sunfish (a snail-eater) with those in bluegill showed almost three-fold higher cadmium concentration in redear than in bluegill sunfish. Carnivorous largemouth bass contained among the lowest whole body trace metal concentrations, while the more omnivorous bluegill generally contained higher concentrations. In this research, redear contain much higher cadmium concentrations than bluegill sunfish, which is consistent with other studies. However, largemouth bass contain the highest mean cadmium concentration. Either the diet of the largemouth bass in this study is different from bass collected in similar studies, or other factors are involved. The bass in Central Florida may eat more insects and crustaceans than do bass from the midwest; this could account for the higher cadmium concentrations in the largemouth bass in this study. The diet of the bluegill in Central Florida may contain more insects than bluegill collected in other studies, which may account for the lower cadmium concentrations. Wiener and Giesy (1979) determined that trace metal concentrations in fish are apparently related to factors other than food habits and trophic status. This was shown by significant differences in trace metal concentration in pickerel and largemouth bass, which occupy the highest trophic level and have similar dietary preferences. They suggested that numerous factors play a role in determining the level of accumulated cadmium; including cadmium source, exposure level, distance from contamination source and the presence of other ions. Factors other than food habits may be why the largemouth bass in this study contain higher cadmium concentrations than values obtained for bass in similar studies, or why the mean cadmium concentration is higher than that of the redear sunfish, the bottom feeder. These factors also may be why the bluegill in this study contain fairly low cadmium concentrations.

The redear sunfish, the largemouth bass, and the bluegill sunfish collected from the stormwater ponds contain more cadmium (significantly more at a $p<0.005$ in the case of the redear and bass) than the fishes collected from the control sites. The results are similar to a study done by Murphy, et al., (1978) where concentrations of cadmium in muscle were significantly greater in samples of both largemouth bass and bluegill from the contaminated site than in corresponding samples from uncontaminated site in Palestine Lake, Indiana. Because the sediment samples obtained in this study are composite samples, they cannot be compared with values in the literature. However, the values are consistent with values obtained for the fish, in that, the composite sediment samples collected from the stormwater ponds contain a higher mean cadmium concentration than the sediment collected from the control sites.

The results suggest that cadmium is bioaccumulating in significant concentrations in the redear sunfish and the largemouth bass collected from the stormwater ponds. Cadmium also is bioaccumulating in the bluegill collected from the stormwater ponds. Bioaccumulation patterns are somewhat different from those observed in other studies. However, cadmium is a heavy metal that should be of concern regarding the fish living in stormwater ponds.

Nickel

Nickel is suspected of being essential for some plants and animals (Gough, et al., 1979). The mean nickel concentrations of the fish in this study are within the range of values reported in similar studies (Uthe and Bligh, 1971; Mathis and Cummings, 1973; Jenkins, 1980).

The bioaccumulation pattern of nickel in this study is consistent with similar studies on nickel and other heavy metals, in which those species most associated with the bottom sediments contain the highest heavy metal concentration. The redear sunfish, the bottom feeder, collected from the stormwater ponds contain the highest mean nickel concentration. The largemouth bass collected from the stormwater ponds have the second highest mean nickel concentration, followed by the bluegill. All three species collected from the stormwater ponds have a higher mean nickel concentration than the fish collected from the control sites (significantly higher for the redear). Composite sediment samples that were collected are consistent with the fish, in that the samples from the stormwater ponds have a higher mean nickel concentration.

The Panel on Nickel (1975), found that most foods including clams, scallops, shrimp, lobsters, crab, marine fishes and freshwater fishes contain nickel levels below 0.75 mg/kg. The mean nickel concentrations of the redear sunfish collected from the stormwater ponds, and the bass collected from the stormwater ponds and the control sites, are higher than 0.75 mg/kg.

The results suggest that nickel is bioaccumulating in all three species of fish collected from the stormwater ponds. Nickel is a heavy metal that should be a concern regarding the fish living in stormwater ponds.

Copper

Copper is a known essential trace element for plants and animals (Gough, et al., 1979). Several effects of exposure to sublethal concentrations of copper have been reported in fish. These include decreased survival, growth, and reproduction (Mount and Stephan, 1969; McKim and Benoit, 1971). In addition, fish exposed to sublethal copper concentrations accumulate copper in certain tissues, particularly gills and liver (Brungs, et al., 1973; Benoit, 1975).

The redear sunfish collected from the stormwater ponds have a mean copper concentration that is higher than values obtained in other studies (Lucas, et al., 1970; Giesy and Wiener, 1977). The mean copper concentrations of the redear sunfish collected from the control sites is within the range of values obtained in other studies. The largemouth bass collected from the both the stormwater ponds and the control sites have a mean copper concentration that is consistent with values obtained in similar studies (Lucas, et al., 1970; Giesy and Wiener, 1977). However, bass collected from the control sites have a higher mean copper concentration than the bass collected from the stormwater ponds. The mean copper concentration of the bluegill sunfish collected in this study is similar to values obtained in similar studies (Lucas, et al., 1970; Giesy and Wiener, 1977).

Elevated levels of copper in fish tissues are associated with development of enhanced tolerance to subsequent copper exposure (Dixon and Sprague, 1981). Fish collected from the control sites may have previously been exposed to copper (possibly from copper-based herbicides and pesticides) and have developed enhanced tolerance. Enhanced tolerance may account for why the largemouth bass collected from the control sites have a higher mean copper concentration than the bass collected from the stormwater ponds.

Mathis and Cummings (1973) reported the highest copper concentration in sediments and in animals living in, or on, the sediments. The tissue concentration of most metals, therefore, appears to be more greatly influenced by association with bottom sediments rather than position in the food chain in aquatic organisms (Wren, et al., 1983). The results of this study are consistent with the literature in that the redear sunfish, the species most associated with the bottom sediment, collected from the stormwater ponds, have the highest mean copper concentration (followed by the largemouth bass, then the bluegill). The composite sediment samples collected from the stormwater ponds have a higher mean copper concentration than the composite sediment samples from the control sites, which is consistent with the results obtained for the fish. Giesy and Wiener (1977) reported no apparent biomagnification of copper, which is consistent with the results obtained in this study.

The results suggest that copper is bioaccumulating towards significant ($p<0.005$) levels in the redear sunfish and the bluegill sunfish collected from the stormwater ponds. Because the largemouth bass collected from the control sites appear to have enhanced copper tolerance, the results do not show a significant accumulation in the bass living in stormwater ponds. Copper is a heavy metal that should be a concern regarding the fish living in stormwater treatment ponds.

Lead

Lead is a non-essential, toxic metal (Davies, et al., 1976). The maximum daily "safe" level of lead that has been determined by the World Health Organization is 0.45 mg/kg (Sorensen, 1991). The FDA standard for lead in seafood is 0.5 mg/kg (Suffern, et al., 1981). The mean lead concentrations of the redear sunfish, the largemouth bass, and the bluegill sunfish collected in this study exceed 0.45 mg/kg.

The mean lead concentration of the redear sunfish and the largemouth bass collected from the stormwater ponds and control sites is higher than values obtained in similar studies (Wiener and Giesy, 1979; Wiener, et al., 1982). This suggests that the "background" lead levels in the redear and the bass from the control sites are as high or higher than contaminated sites in other parts of the United States.

The redear sunfish and the largemouth bass collected from the stormwater ponds have a mean lead concentration that is much higher than values obtained in similar studies (McIntosh and Bishop, 1976; Wiener and Giesy, 1979; Wiener, et al., 1982). The bluegill sunfish collected from the stormwater ponds and the control sites have a mean lead concentration that is within the range of values obtained in similar studies.

Many studies have shown that whole fish levels of lead are always higher in fish exposed to contaminated sediments than in controls (Sorensen, 1991), which is consistent with the results of this study. The redear sunfish, the largemouth bass, and the bluegill sunfish collected from the stormwater ponds have higher mean lead concentration than the fish collected from the control sites. The mean lead concentration is statistically significantly higher in the redear from the stormwater ponds ($p<0.005$). The mean lead concentration is significantly higher ($p=0.0076$), from a biological point of view, in the bass from the stormwater ponds than the bass collected from the control sites. The bluegill collected from the stormwater ponds also contain a mean lead concentration that could be considered biologically significant, because it is higher than the bluegill collected from the control sites. McIntosh and Bishop (1976) found that bluegill had a mean lead concentration that was significantly higher than the metal concentrations in similar species from lakes and ponds with no known metal inputs, which is similar to the results obtained in this research.

The results are consistent with the literature in that no trophic level biomagnification of lead was observed among fishes of different trophic position (Wiener and Giesy, 1979; Wren, et al., 1983). The results are similar to those found with other metals in that the redear sunfish collected from the stormwater ponds, which is the species most associated with the bottom sediments, has the highest mean lead concentration.

Gasoline combustion accounts for more than 80 percent of lead in air (Lagerwerff, 1972). Anthropogenic burdens of lead are evident even in remote lakes in undeveloped watersheds, due to long-range atmospheric transport and deposition (Wiener, 1977). This may explain why the composite sediment samples collected from the control sites have a higher mean lead concentration than the composite sediment samples collected from the stormwater ponds. Because the lead in the sediment from the control sites is more of a historical nature, it may not be as available to the fish living in the control ponds and lakes. Atmospheric deposition could explain why the fish collected from the control sites have higher mean lead concentrations than reported in many other studies. However, because the fish that live in stormwater ponds receive continuous significant inputs of lead, the mean lead concentration of the fish collected from the stormwater ponds is higher than the fish from the control sites.

The results of this study suggest that lead is bioaccumulating in the fish that live in stormwater ponds. Lead is a heavy metal that should be of concern in the fish living in stormwater ponds. The results suggest also that fish collected from the control sites appear to have lead levels above approved "safe" or tolerable levels. A more detailed study should be done to determine if the lakes and ponds in Central Florida are contaminated by lead.

Zinc

Zinc is an essential element for human and animal growth; it is a constituent of all cells and an important cofactor for certain enzymes (Phillips and Russo, 1978). In excess, zinc is known to be toxic to fish, causing mortality, growth retardation, tissue alterations, respiratory and cardiac changes, inhibition of spawning, and a multitude of additional detrimental effects which threaten survival.

The mean zinc concentration of the redear collected from the stormwater ponds, although significantly higher ($p<0.005$) than in the redear collected from the control sites, is much lower than values obtained for redear sunfish in the contaminated West Basin, Palestine Lake, Indiana (Murphy, et al., 1978). The mean zinc concentration of the redear sunfish collected from the control sites is similar to values obtained in a similar study (Murphy, et al., 1978). The mean zinc concentration of the largemouth bass collected from the stormwater ponds is higher than values obtained in similar studies (Murphy, et al., 1978; Wiener and Giesy, 1979). Largemouth bass collected from the control sites have a mean zinc concentration that is similar to that of bass collected from an uncontaminated system in Indiana (Murphy, et al., 1978). The mean zinc concentration of the bluegill sunfish collected from the stormwater ponds and the control sites is within the range of values obtained in similar studies (McIntosh and Bishop, 1976; Wiener and Giesy, 1977; Murphy, et al., 1978).

Murphy, et al., (1978) found that largemouth bass and bluegill sunfish in an ecosystem heavily contaminated by trace metals accumulated significantly more zinc than bass and bluegill from an uncontaminated ecosystem. The results obtained by Murphy, et al., (1978) are consistent with the results obtained in this study. All three species collected from the stormwater ponds have higher mean zinc concentrations than the fish collected from the control sites (significantly higher at a $p<0.005$ for redear and bass). Other researchers obtained similar results for bluegill (McIntosh and Bishop, 1976). The composite sediment samples collected from the stormwater ponds also have a higher mean zinc concentration than the composite sediment samples collected from the control sites, which is consistent with the results obtained for the fish.

The results of this study indicate that no trophic level biomagnification is occurring for zinc, which is consistent with results obtained by Wiener and Giesy (1979). The redear sunfish collected from the stormwater ponds, the species most associated with the bottom sediments, contain the highest mean zinc concentration. This is consistent with the results obtained for most of the heavy metals in this study. One difference, as compared to the other heavy metals, is that the bluegill sunfish collected from the stormwater ponds have a mean zinc concentration that is higher than the bass collected from the stormwater ponds. This is consistent with a study done by Murphy, et al., (1978) in which bluegill contained significantly greater concentrations of zinc than did bass, probably because of differences in feeding behavior.

Bioaccumulation of zinc is occurring in the redear sunfish, the largemouth bass, and the bluegill sunfish that live in stormwater ponds. The bioaccumulation patterns, observed in this study, are consistent with those previously reported. Zinc is a metal that should be a concern regarding the fish living in stormwater treatment ponds.

SUMMARY

The results of this study indicate that cadmium, nickel, copper, lead, and zinc, present in stormwater runoff, are bioaccumulating in the fish living in stormwater ponds. Redear sunfish, the species most associated with the bottom sediments, collected from stormwater pond, are bioaccumulating significant concentrations ($p<0.005$) of cadmium, nickel, copper, lead, and zinc. Largemouth bass, the predator, collected from stormwater ponds are bioaccumulating cadmium and zinc to statistically significant ($p<0.005$) concentrations. Biologically significant concentrations of nickel, copper, and lead are bioaccumulating in the bass living in stormwater ponds. These concentrations are biologically significant because they are higher than the concentrations found in bass collected from uncontaminated environments. The omnivorous bluegill sunfish that lives in stormwater ponds is bioaccumulating significant ($p<0.005$) concentrations of copper. The cadmium, nickel, lead, and zinc concentrations in the bluegill sunfish collected from the stormwater treatment ponds represent biologically significant concentrations because they are higher than the concentrations found in the bluegill collected from uncontaminated environments. It is interesting to note that even though the stormwater ponds were only 3-5 years old when the fish were collected from them, significant levels of the five heavy metals are found in the fish. It is not known what the effect is on the wading birds and other wildlife that are feeding on the fish living in stormwater ponds; it is beyond the scope of this study. Research is needed to determine not only how often wading birds and wildlife forage in stormwater ponds, but what effect eating fish containing significant heavy metal concentrations has. Studies also should be done on the heavy metal concentrations present in the other faunal components that live in stormwater ponds (insects, crustaceans, frogs, turtles, etc.). Because the results of this study show that the fish that live in stormwater ponds, especially the species most associated with the bottom sediments, are bioaccumulating significant concentrations of heavy metals, attracting wildlife to these ponds should be discouraged. Stormwater ponds help keep the lakes and rivers of Florida from becoming polluted from urban runoff; however, the fish that live in them represent a potentially contaminated food source.

ACKNOWLEDGEMENTS

I would like to thank the St. Johns River Water Management District for funding this project and all the people who work for the District who helped me, the staff of the Florida Game and Freshwater Fish Commission for lending me their seines and taking me out on their electrofishing boats, and my Graduate Committee for all their help with this project: Lorrie Hoffman, Buck Snelson, Jack Stout, and David Vickers.

LITERATURE CITED

Atchison, G. J., B. R.Murphy, W. E. Bishop, A. W. McIntosh, and R. A. Mayes. 1977. Trace metal contamination of bluegill (Lepomis macrochirus) from two Indiana lakes. Trans. Am. Fish. Soc. 106(6):637-640.

Benoit, D. A. 1975. Chronic effects of copper on survival, growth, and reproduction of the bluegill (Lepomis macrochirus). Trans. Am. Fish. Soc. 104:353-358.

Brungs, W. A., E. N. Leonard,, and J. M. McKim. 1973. Acute and long-term accumulation of copper by the brown bullhead, Ictalurus nebulosus. J. Fish. Res. Bd. Can. 30:583-586.

Carlander, K. D. 1969. Handbook of freshwater fishery biology, Volume 2. The Iowa State University Press, Ames, Iowa. 431 p.

Davies, P. H., J. P. Goettl,, Jr., J. R. Sinley, and N. F. Smith. 1976. Acute and chronic toxicity of lead to rainbow trout, Salmo gairdneri, in hard and soft water. Water Res. 10:199-206.

Dixon, D. G. and J. B. Sprague. 1981. Copper bioaccumulation and hepatoprotein synthesis during acclimation to copper by juvenile rainbow trout. Aq. Toxicol. 1:69.

Giesy, J. P., Jr. and J. G. Wiener. 1977. Frequency distributions of trace metal concentrations in five freshwater fishes. Trans. Am. Soc. 106(4):393-403.

Gough, L. P., H. T. Shacklette, and A. A. Case. 1979. Element concentrations toxic to plants, animals, and man. Geological Survey Bulletin 1466, U. S. Government Printing Office, Washington, D.C. 80 p.

Harper, H. H., 1985. Fate of heavy metals from runoff in stormwater management systems. Ph.D. Dissertation, University of Central Florida, Orlando, Florida. 389 p.

Harper, H. H. 1990. Stormwater pollutants and their removal. Stormwater Short Course, Florida Engineering Society and Florida Department of Environmental Regulation. October 11-12, 1990. pp. 59-91.

Jenkins, D. W. 1980. Nickel bioaccumulation in aquatic biota. In: Nickel in the environment, J. O. Nriagu (Editor). Wiley Interscience, New York.

Lagerwerff, J. V. 1972. Lead, mercury, and cadmium as environmental contaminants. In: Micronutrients in agriculture. Soil Sci. Soc. of America, Inc. pp. 593-636.

Lucas, H. F., Jr., D. N. Edgington, and P. J. Colby. 1970. Concentrations of trace elements in Great Lakes Fishes. J. Fish. Res. Bd. Can. 27:677-684.

Luoma, S. N. 1983. Bioavailability of trace metals to aquatic organisms-a review. Sci. Total Environ. 28:1-22.

McClane, A. J. 1978. McClane's field guide to freshwater fishes of North America. Henry Holt and Company, New York. 212 p.

McKim, J. M. and D. A. Benoit. 1971. Effects of long-term exposures to copper on survival, growth, and reproduction of brook trout (Salvelinus fontinalis). J. Fish. Res. Bd. Can. 28:655-662.

Macek, K. J., S. R. Petrocelli, and B. H. Sleight III. 1979. Considerations in assessing the potential for, and significance of, biomagnification of chemical residues in aquatic food chains. Aquatic Toxicology. ASTM STP 667. L. L. Marking and R. A. Kimerle, (Editors). American Society for Testing and Materials. pp. 251-268.

Mathis, B. J. and T. F. Cummings. 1973. Selected metals in sediments, water, and biota in the Illinois river. Journal WPCF 45(7):1573-1583.

Mount, D. I. and C. E. Stephan. 1969. Chronic toxicity of copper to the fathead minnow (Pimephales promelas) in soft water. J. Res. Bd. Can. 26:2449-2457.

Murphy, B. R., G. J. Atchison, and A. W. McIntosh. 1978. Cadmium and zinc in the muscle of bluegill (Lepomis macrochirus) and largemouth bass (Micropterus salmoides) from an industrially contaminated lake. Environ. Pollut. 17:253-257.

Murphy, B. R., G. J. Atchison, A. W. McIntosh, and D. J. Kolar. 1978. Cadmium and zinc content of fish from an industrially contaminated lake. J. Fish. Biol. 13:327-335.

Nightingale, H. I. 1975. Lead, zinc, and copper in soils of urban storm-runoff retention basins. American Water Works Ass. J. 67:443-446.

Nightingale, H. I. 1987. Accumulation of As, Ni, Cu, and Pb in retention and recharge basins soil from urban runoff. Amer. Wat. Res. Ass., Water Resources Bulletin. 23(4):663-672.

Owe, M., P. J.Craul,, and H. G. Halverson. 1982. Contaminant levels in precipitation and urban surface runoff. Am. Wat. Res. Ass., Water Resources Bulletin. 18(5):863-868.

Panel on Nickel. 1975. Nickel. Committee on Medical and Biological Effects of Environmental Pollutants, National Research Council, National Academy of Sciences, Washington, D.C. 277 p.

Phillips, G. R. and R. C. Russo. 1978. Metal bioaccumulation in fishes and aquatic invertebrates. EPA-600/3-78-103. 121 p.

SAS Institute Inc. 1985. SAS user's guide: statistics, version 5 edition. Cary, NC: SAS Institute Inc. 956 p.

Sorensen, E. M. B. 1991. Metal poisoning in fish. CRC Press, Boca Raton, FL. 374 p.

Suffern, J. S., C. M. Fitzgerald, and A. T. Szluha. 1981. Trace metal concentration in oxidation ponds. Journal WPCF. 53(11):1599-1608

Uthe, J. F. and E. G. Bligh, 1971. Preliminary survey of heavy metal contamination of Canadian freshwater fish. J. Fish. Res. Bd. Can. 28(5):786-788.

Whalen, P. J., and M. G. Cullum. 1988. An assessment of urban land use/ stormwater runoff quality relationships and treatment efficiencies of selected stormwater management systems. SFWMD Tech. Pub. 88-9 55 p.

Whipple, Jr. W. and J. V. Hunter. 1977. Nonpoint sources and planning for water pollution control. Journal WPCF. 49:15-23.

Wiener, J. G. and J. P. Giesy, Jr. 1979. Concentrations of Cd, Cu, Mn, Pb, and Zn in fishes in a highly organic softwater pond. J. Fish. Res. Bd. Can. 36:270-279.

Wiener, J. G., G. A. Jackson, T. W. May, and B. P. Cole. 1982. Longitudinal distribution of trace elements (As, Cd, Cr, Hg, Pb, and Se) in fishes and sediments in the upper Mississippi River. Proceedings: 15th Annual Meeting of the Miss. Riv. Res. pp. 139-170.

Wigington, Jr. P. J., C. W. Randall, and T. J. Grizzard. 1983. Accumulation of selected trace metals in soils of urban runoff detention basins. Am. Wat. Res. Ass., Water Resources Bulletin. 19(5):709-718.

Wilbur, R. L. 1969. The redear sunfish in Florida. FGFWFC. Fishery Bulletin No. 5. Dingell-Johnson F-22. 64 p.

Wibur, W. G. and J. V. Hunter. 1979. Distributions of metals in street sweepings, stormwater solids, and urban aquatic sediments. Journal WPCF 51:2810-2822.

Williford, W. A., M. J. Mac, and R. J. Hesselberg. 1987. Assessing the bio-accumulation of contaminants from sediments by fish and other aquatic organisms. Hydrobiologia 149:107-111.

Wren, C. D., H. R. MacCrimmon, and B. R. Loescher. 1983. Examination of bioaccumulation and biomagnification of metals in a precambrian shield lake. Water, Air, and Soil Pollution 19:277-291.

Yousef, Y. A., M. P. Wanielista, and H. H. Harper. 1985. Fate of pollutants in retention/detention ponds. Proceedings: Stormwater management "an update." M. P. Wanielista and Y. A. Yousef, (Editors). University of Central Florida, Environ. Systems Engr. Institute, Pub. No. 85-1, Orlando, Florida.

RESTORING SEAGRASS BEDS: SOME NEW APPROACHES WITH RUPPIA MARITIMA L. (WIDGEON-GRASS)

Michael J. Durako, Judith J. Shup, Carmen J. Andress,
FLORIDA MARINE RESEARCH INSTITUTE
100 Eighth Avenue SE
St. Petersburg, FL 33701

David A. Tomasko
SARASOTA BAY NATIONAL ESTUARY PROGRAM
1550 Ken Thompson Parkway
Sarasota, FL 34236

ABSTRACT

Experimental seagrass plantings were conducted in three artificially-created ponds in Central Sarasota Bay. Planting units were produced from locally collected Ruppia maritima (Widgeon-grass) that had been clonally micropropagated in vitro. Explants used to produce the planting units were rhizome segments containing five short-shoot nodes. Explants were propagated axenically in sucrose-enriched media supplemented with the cytokinin 2-isopentyl adenine (2iP). Culture conditions resulted in high rates of node production and branching; nodes and branches exhibited doubling times of less than two weeks. We used a new planting technique in which the micropropagated plants were placed in biodegradable cotton-mesh bags that contained river-rock ballast in three configurations. This technique protected and anchored the fragile planting units prior to rooting. Leaves and roots rapidly grew through openings in the cotton-mesh bags, and plants became rooted before the bags decomposed (about two months). Survival of the plants in Pond No. 1 ranged from 60% to 97% after three to five months and was greatest in planting units with rock ballast secured to the bag corners. Survival ranged from 15 to 63% in Pond No. 2, which had less tidal flushing, rockier sediments, and a persistent accumulation of macroalgae. No planting units survived in Pond No. 3, seemingly because of unsuitable sediments and disturbance by humans. Plants expanded rapidly, and areal coverage doubled approximately every month. These preliminary results indicate that Widgeon-grass, produced through micropropagation, may be suitable for planting in situ. In addition, the use of the cotton-bag technique enhances establishment and survival of this structurally fragile species.

INTRODUCTION

Seagrass communities provide critical habitat and trophic structure for ecologically and economically important fishery species, and they are a major force in stabilizing coastal ecosystems (Thayer, et al., 1984; Zieman and Zieman, 1989). Unfortunately, seagrasses have significantly declined in areal coverage and quality in many coastal areas. Losses of 80 to 90% of historical seagrass coverages have occurred in areas such as Tampa and Chesapeake Bays (Lewis, et al., 1985;

Dennison, et al., 1993). Much of this loss has been attributed to excessive nutrient loading and the resulting poor water quality. Reductions in nutrient loading in many coastal systems over the past ten years have contributed to improved water quality, and seagrasses have recolonized some shallow areas that had been barren for several decades (Johansson and Lewis, 1992). However, lack of recruitment stock may limit the extent to which natural recolonization can occur in suitable sites.

Because resource managers are recognizing the importance of seagrasses in the production and maintenance of living marine resources, much emphasis has been placed on developing techniques for restoring seagrass beds. Most previous seagrass-restoration projects have depended on the destructive and labor-intensive harvest of planting units from natural donor beds. The only non-destructive approaches have been based on the collection of Thalassia testudinum Banks ex Konig seedlings and sprigs from beach wrack (Thorhaug, 1974; Durako and Moffler, 1981, 1984; Derrenbecker and Lewis, 1983; Phillips and Lewis, 1983). In vitro micropropagation offers a potentially low-cost, highly-efficient, and non-destructive technique for propagating seagrasses at rates that are much higher than those obtained with other methods of propagation. This biotechnological approach can also provide genetically uniform material for bioassaying various types of environmental stresses and allows for the identification, selection, and clonal propagation of stress-tolerant genotypes. In addition, the production of vigorous planting stock can be adjusted to meet the sporadic demands of planting projects (Ailstock, 1986).

Ruppia maritima L. (Widgeon-grass) is a cosmopolitan seagrass that occurs throughout the world in a variety of ecological niches. In addition, Ruppia is characteristically an early and frequent recolonizer of sites previously occupied by climax seagrass species. Although Ruppia has not been widely used in previous seagrass planting studies, this species may be better suited for initial testing of restoration-site suitability than other species are, because it has the broadest physiological tolerances of any seagrass. In this report, we describe the results of several trial plantings. We used a new planting technique in which micropropagated Ruppia was placed in ballasted, biodegradable cotton-mesh bags.

MATERIALS AND METHODS

Micropropagation

Ruppia maritima was micropropagated using the protocol described by Koch and Durako (1991), with some modifications. We modified the sterilization procedure to decrease contamination rates by including a vacuum treatment that infiltrates the lacunar spaces of the explants (i.e., field-collected rhizome sections) with sterilizing and antimicrobial agents.

Plant material, consisting of rhizome sections with attached short-shoots, was collected from Sarasota Bay in February, 1992. The sections were cut into single-branch segments having at least five shoots, and were soaked in a fungicide solution (Captan, 1.5 g/L of 20 ppt Sterile Artificial Seawater [SASW]) overnight on an orbital shaker. Segments were then trimmed to three to five-node explants using aseptic techniques in a laminar-flow hood. Batches of 10-20 explants were surface sterilized by vacuum infiltration (40-cm Hg vacuum for 10 minutes) of a Clorox/SASW solution (1:5 vol/vol) with 0.1% Tween as a surface wetting agent. Following infiltration, explants were rinsed three times in sterile seawater.

Single, surface-sterilized explants were aseptically transferred to 24-well plates containing 2 ml of a sterile-filtered antibiotic solution (0.20 g/L each of rifampin and erythromycin in 20 ppt SASW). Explants were infiltrated (40-cm HG vacuum for 30 minutes) with antibiotics on the orbital shaker and then were equilibrated to ambient pressure for 24 hours. Individual explants were then directly transferred to 24-well plates containing 2 ml of culture medium. The medium consisted of Koch and Durako basal salts (Sigma No. K-1254), 1% (wt:vol) sucrose, and 10 mg/L 2iP (N^6-[2-Isopentenyl] adenine) in 20 ppt SASW at pH 5.6. Cultures were illuminated (12 h light:12 h dark) by cool-white fluorescent lamps with a PAR (photosynthetically active radiation photon-flux density) of approximately 100 $\mu E\ m^{-2}\ s^{-1}$ and were maintained at 26°C in a temperature-controlled culture room. After 10 days incubation, uncontaminated plants were aseptically transferred to sterile 25 x 150-mm culture tubes containing 40 ml of fresh media. The rapidly growing plants were micropropagated by subdividing into 5 to 10-node segments at monthly intervals.

One month prior to being planted into the field, plants were transferred to culture tubes containing media lacking sucrose to "harden" the plants and promote root production. Cultures were also placed in growth chambers with high-intensity lighting (500-700 $\mu E\ m^{-2}\ s^{-1}$) to acclimatize.

Planting Site

Clonally micropropagated Ruppia maritima was planted in three ponds (designated Pond No. 1, No. 2, and No. 3) at the Sarasota BayWalk site on City Island in Central Sarasota Bay, Florida. Sarasota BayWalk is a 4.5-acre habitat-restoration and enhancement site created as an Early Action Demonstration Project through the Sarasota Bay National Estuary Program (SBNEP). The objective of the BayWalk project was to create highly-productive, diverse, and integrated habitats by removing non-native plant species, lowering land elevations, and planting appropriate native coastal and wetland plants (Roat, et al., 1992). Pond No. 1 is adjacent to the SBNEP office and is the only natural pond on the site. This pond is bordered on the south side by a natural mangrove forest and on the north side by a planted Spartina alterniflora Loisel (fringe marsh). Pond No. 1 has a direct connection to Sarasota Bay via a small tidal creek; shallow channels connect Pond No. 1 to Pond No. 2, and Pond No. 2 to Pond No. 3. Ponds No. 2 and No. 3 were created to provide suitable habitat for fish. These ponds have rockier sediments, experience less tidal flushing, and have more variable salinities than does Pond No. 1. The shoreline of Pond No. 2 consists of planted Spartina alterniflora. Pond No. 3 is the largest, deepest, and least flushed of the three ponds. The sediments in this pond consist of a thin veneer of sand over a compacted, rocky base. The shoreline of this pond has sparse plantings of Rhizophora mangle L. and some Spartina. Ponds No. 2 and No. 3 are not within sight of the SBNEP office and may have been subject to disturbance by humans.

Planting Units

Previous test plantings of Ruppia (Durako, unpublished data) showed that the staple technique (Fonseca, et al., 1982), currently the most widely used method for planting seagrass, was inappropriate for Widgeon-grass because of its fragile rhizome. Thus, a new planting technique was developed in which biodegradable cotton-mesh (cheesecloth) bags that contained river-rock ballast were used. This technique anchored and protected the fragile planting units (PUs). Several bag configurations were tested:

Treatment I (T-I): bags contained loose stones and were tied with a knot or jute twine,

Treatment II (T-II): bags were constructed as in T-I and were tied with jute twine to the inside corners of 25 cm x 25 cm PVC quadrats (4 bags quad^{-1}) to protect the PUs from bioturbation, and

Treatment III (T-III): bags contained stones that were sewn into the corners. All bags were approximately 8 cm x 10 cm.

In the laboratory, micropropagated plants were transferred from culture tubes and inserted into the bags, the open end of each bag was folded over and stapled, and the bags were placed in a cooler chest in layers separated by paper towels moistened with media. The cooler chest containing PUs was then transported to the field state. Planting units were installed in the ponds within three hours of initial preparation.

Field Planting

Test plantings were conducted on three dates. All PUs were installed on approximately 25-cm centers. Within a pond, PUs were installed in an area that was apparently uniform. On October 13, 1992 we installed a total of 177 PUs (A symbols in Figure 1). In Pond No. 1, three rows of ten t-I bags were alternated with two rows of five T-II quadrats each (70 PUs). In Pond No. 2, a row of ten T-I bags was placed on either side of one row of five T-II quadrats (40 PUs). Pond No. 3 was planted in the same configuration as Pond No. 1, except a single T-I bag replaced a T-II quadrat in one of the two T-II quadrat lines (67 PUs).

A second group of 50 PUs, consisting entirely of T-III bags, was installed on November 25, 1992 (B symbols in Figure 1). A row of ten PUs was placed parallel to one of the outside T-I rows in Pond No. 1 (10 PUs). Two rows of ten PUs each were placed outside and parallel to the two outside T-I rows in both Pond No. 2 (20 PUs) and Pond No. 3 (20 PUs).

The final group of 138 PUs, again consisting entirely of T-III bags, was installed on December 16, 1992 (C symbols in Figure 1). In this planting, a plot with three rows of eight PUs and two rows of seven PUs each was planted in Pond No. 1 (38 PUs), a 5 x 6-unit plot was planted in Pond No. 2 (30 PUs), and a 7 x 10-unit plot was planted in Pond No. 3 (70 PUs).

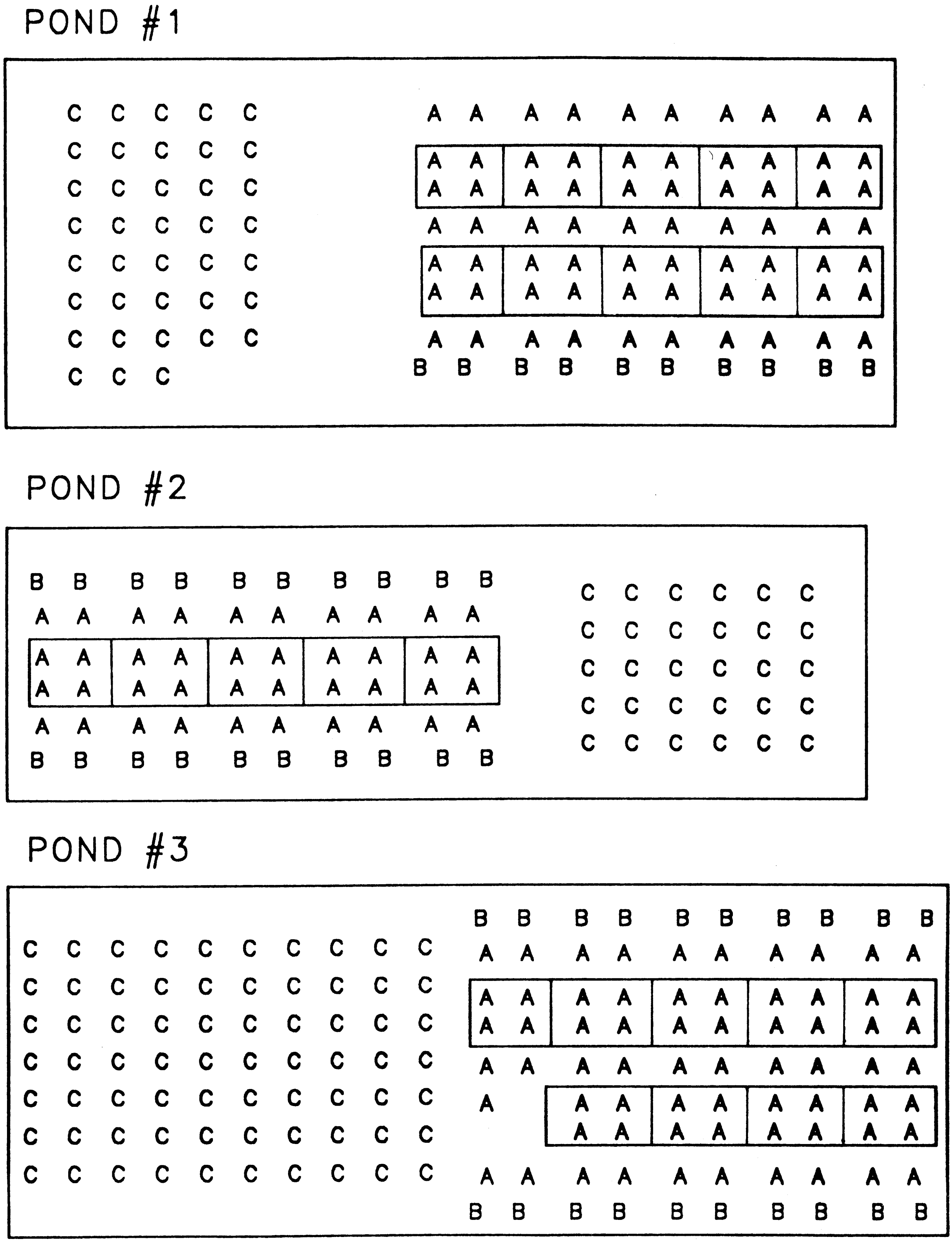


Figure 1. Diagram of the relative placement of micropropagated Ruppia maritima plants within planting areas in three ponds at Sarasota BayWalk. Letters refer to planting date: A=10/13/92, B=11/25/92, C=12/16/92. The A's signify T-1 planting units (PUs), the A's in boxes signify T-II quadrant PUs, B's and C's signify T-III PUs (see Materials and Methods).

Survival of the PUs was monitored regularly until March 28, 1993. Areal coverage rates of T-I and T-III bags in Pond No. 1 were estimated by averaging the width (d, in cm) of the PUs on two perpendicular axes and by computing areal spread as:

$$[\pi(d/2)^2]/\Delta days = cm^2/day$$

Because of the close proximity and rapid coalescence of the PUs within the quadrats, areal coverage rate could not be measured for Treatment II. Doubling time (DT) in days was calculated as

$$DT = .301/[(\log x_1 - \log x_0)/\Delta t]$$

where x_1 and x_0 are the number of nodes and branches for cultured plants or the area of a PU at t_1 and t_0 days; $\Delta t = t_1 - t_0$.

RESULTS AND DISCUSSION

More than 350 robust plants of Ruppia maritima were produced from fewer than 20 five-node rhizome segments using the in vitro micropropagation techniques described by Koch and Durako (1991). The axenic Ruppia cultures had high rates of growth and branching in vitro. Figure 2 illustrates characteristic rates of node production and branching during a month-long propagation cycle that was initiated with single-branch, five-node explants. The growth rates shown in this figure yielded doubling times of 13.5 days for nodes and 11.0 days for branches. These results are comparable to the two-week doubling times reported for axenic sago pondweed, Potamogeton pectinatus L., cultured in vitro in nutrient-enriched media (Ailstock, et al., 1991).

Leaves rapidly grew through the ballasted cheesecloth bags used for planting the Ruppia, and the plants became rooted and started to spread before the bags deteriorated. Doubling times for the areal coverage of T-I and T-III plants in Pond No. 1 averaged from 25 to 37 days, and areal coverage rates averaged from 5 to 19 cm^2 day^{-1} (Figure 3). The areal coverage rates for Ruppia fell within the range of values reported by Fonseca, et al., (1987) for transplants of field-harvested PUs of shoal grass, Halodule wrightii Aschers. (11.4 cm^2 day^{-1} for the northern Gulf of Mexico to 74 cm^2 day^{-1} for the Florida Keys), and manatee grass, Syringodium filiforme Kutzing (4.7 cm^2 day^{-1} for the northern Gulf of Mexico to 40 cm^2 day^{-1} for the Florida Keys). Because these ranges are similar and our Ruppia data represent only fall and winter growth, the micropropagated material seems to be robust and has a high growth potential. The relatively long DT and low coverage rate of the December 16th plantings probably resulted from the low water temperatures (water temperatures declined from 27.2° C on 12/16/92 to about 14.5°C on 2/21/93) and short daylengths for this planting group. Even so, the 38-unit plot installed on December 16th completely coalesced within three months.

Pond No. 1 seemed to provide the best conditions for survival and growth of the Ruppia (compare Figures 4 and 5), perhaps because it is a natural, relatively well-flushed pond. The first group of bags (T-I) had an initial 37% loss of units that

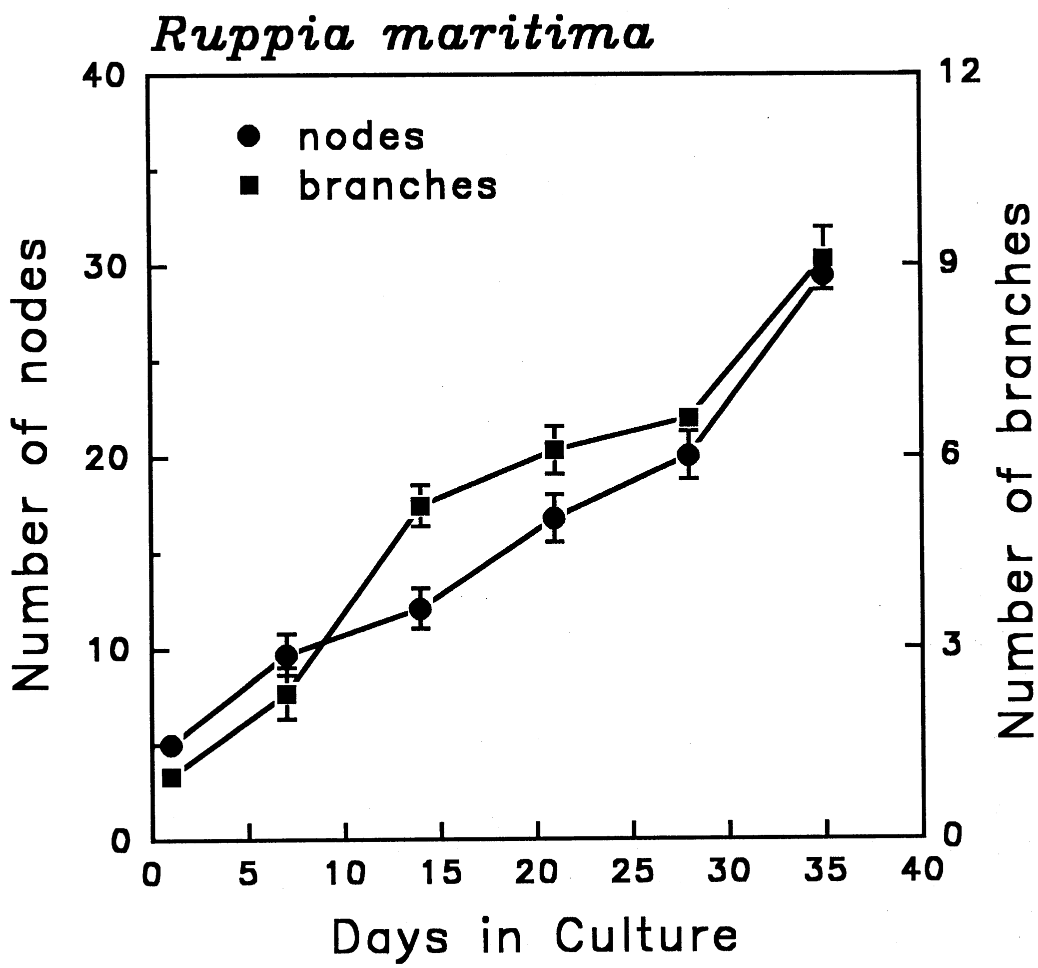


Figure 2. Production (mean ± s.e.; n=10) of does and branches for axenic micropropated Ruppia maritima during a one-month propagation cycle.

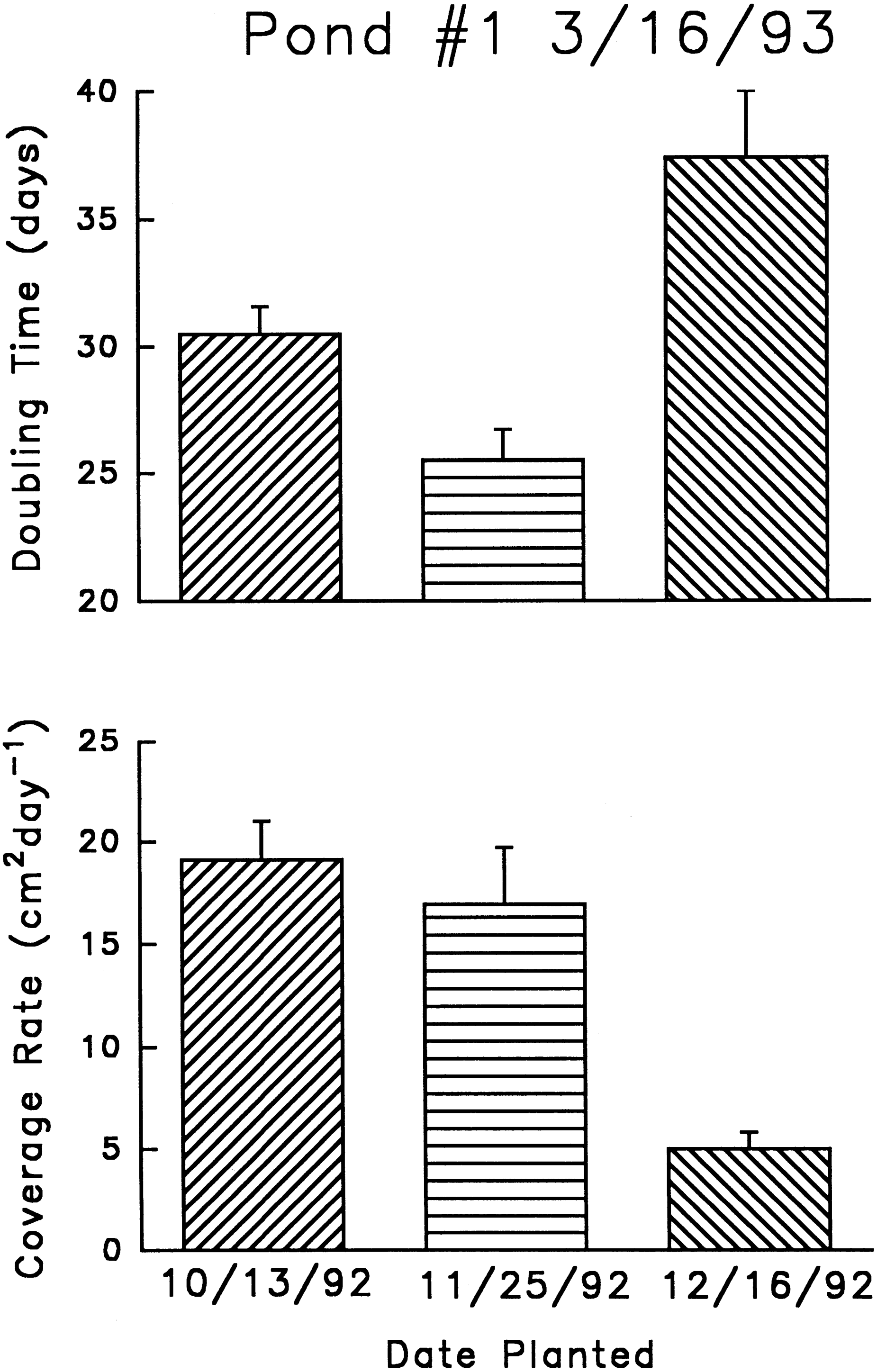


Figure 3. Areal-coverage doubling time and rate of spread for T-I (10/13/92) and T-III (11/25/92) and 12/16/92) plants in Pond No. 1 as of 3/16/93.

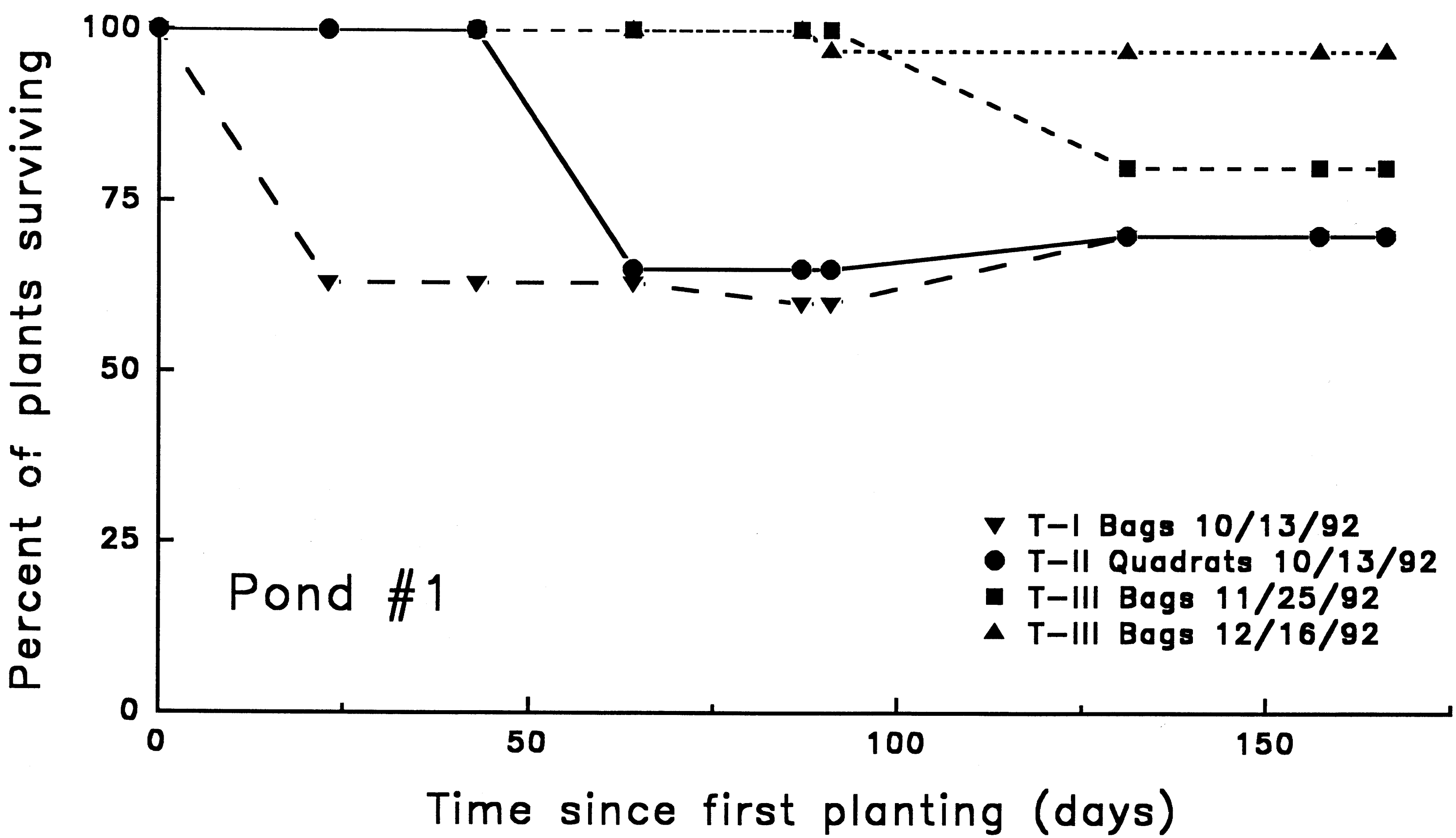


Figure 4. Survival of micropropagated _Ruppia maritima_ planted on three dates in Pond No. 1.

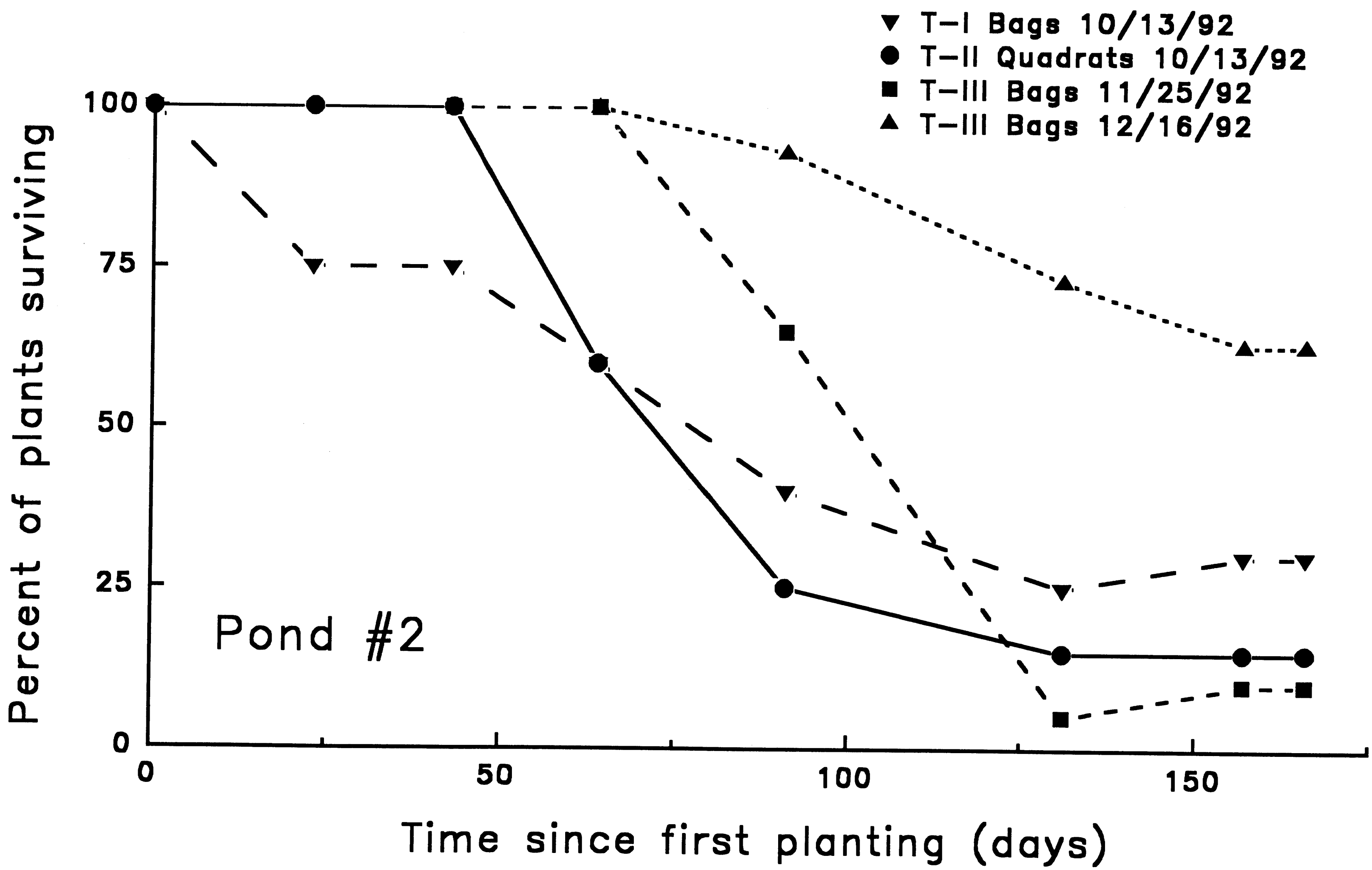


Figure 5. Survival of micropropagated Ruppia maritima planted on three dates in No. 2

seemed to be due to a combination of factors related to the T-I bag design. The knots and jute twine used as bag closures attracted the attention of curious herons and egrets who picked up the bags by the knots and flung them out of the planting grid (personal observation). We later observed some of these (displaced) units growing outside the grid, so the true survival rate is higher than shown in Figure 4. Because the knots and twine tended to float up in the water column, they became fouled with algae, and this may have shaded out the underlying Ruppia. This material was also relatively slow to break down; the twine persisted long after the cheesecloth had disappeared (after about two months).

Because biogenic effects have been reported to be one of the most frequent problems associated with the failure of seagrass restoration projects (Fonseca, 1990), PVC quadrats (the T-II treatment) were used to protect PUs from decapod excavations and from sediment blow-outs caused by rays. No loss of T-II PUs occurred in Pond No. 1 (Figure 4) or Pond No. 2 (Figure 5) for the first one and one-half months (perhaps the size of the quads protected the bags from "acute heron/egret displacement"). However, the twine that secured the PUs to the PVC quadrats became increasingly fouled, and we observed 35% to 40% mortality during the second month.

We found no evidence of bioturbation by benthic organisms or by rays, but we observed problems with the T-I and T-II PUs associated with the jute twine and knotted bag closures. Therefore, we redesignated the cheesecloth bags so they would lie flat on the bottom. This was accomplished by securing the rock ballast into the corners of the bags and stapling the bag closed. Figure 4 illustrates the increased survival rate of plants in bags of the T-III design over those in bags of the T-I and T-II designs for two Pond No. 1 plantings on November 25th (80% survival after 123 days) and December 16th (97% survival after 102 days). Because they lay flat on the bottom, the T-III bags had increased surface contact with the sediment, and the top side had better exposure to light. The T-III bags also deteriorated in a more uniform manner.

Compared to plant survival in Pond No. 1, survival in Pond No.2 (Figure 5) and Pond No. 3 (100% mortality or loss) was much lower for all PU treatments and for all three planting dates (Figure 4). The last group of PUs in Pond No. 2, which were installed in an area with the least rocky sediments of the pond, had the best survival (63% after 102 days). Pond No. 2 experienced a sustained, pond-wide bloom of macroalgae that prevented us from measuring areal spread and eventually covered most of the PUs. No PUs survived more than about one month in Pond No. 3; at least some of the loss was associated with disturbance by humans. We found four of our PVC quadrats neatly stacked on the shoreline during on of our first monitoring visits, and we also observed evidence of foot traffic in the planting grids. The lower survival rates in Ponds No. 2 and No. 3 may also have been partially a result of sediment-related factors. Both ponds were artificially created and had coarser and rockier sediments than did Pond No. 1. In Pond No. 3, the underlying sediments had been mechanically compacted, which may have prevented the PUs from rooting.

SUMMARY

Restoring and creating seagrass beds is of paramount importance in compensating for previous and current seagrass losses. Much information is still needed before we can effectively manage and restore seagrass systems (Fonseca, 1990). However, the newly developed micropropagation procedures and planting techniques for species such as Ruppia maritima (Koch and Durako, 1991) provide both ecologically and economically sound alternatives to the current seagrass restoration practices, which can be labor-intensive and destructive. Tissue culture and micropropagation have become well-established approaches for producing and rapidly multiplying many agriculturally and horticulturally important crops, but these approaches have only recently been applied to submersed aquatic angiosperms (Ailstock, 1986; Durako, 1988; Ailstock, et al., 1991; Koch and Durako, 1991; Bird, et al., 1993). One of the greatest benefits of these biotechnological approaches is that they may allow us to select and produce high numbers of vigorous planting units without damaging natural donor beds. The results of this study indicate that plant material produced by in vitro micropropagation is capable of vigorous growth in situ. The ballasted cheesecloth-bag technique introduced here provides a biodegradable alternative to the use of persistent metal staples for anchoring planting units. The bag technique may also reduce installation time because divers will not need to "plant" the units, and the bags may enhance survival of fragile transplants by reducing plant breakage.

ACKNOWLEDGEMENTS

This work was supported by funds provided through the National Oceanic and Atmospheric Administration, Coastal Ocean Program, Grant No. EHP-23. Additional Logistic support was provided by the Sarasota Bay National Estuary Program.

LITERATURE CITED

Ailstock, M. S. 1986. Clonal propagation of Potamogeton pectinatus in axenic culture. Pp. 11-17 In F. J. Webb (Editor), Proc. 13th Annual Conf. Wetlands Restoration Creation. Hillsborough Community College, Tampa, FL.

Ailstock, M. S., W. J. Fleming, and T. J. Cooke. 1991. The characterization of axenic culture systems suitable for plant propagation and experimental studies of the submersed aquatic angiosperm (Potamogeton pectinatus (sago pondweed). Estuaries 14:57-64.

Bird, K. T., B. R. Cody, J. Jewitt-Smith, and M. E. Kane. 1993. Salinity effects on Ruppia maritima L. cultured in vitro. Bot. Mar. 36:23-28.

Dennison, W. C. et al. 1993. Assessing water quality with submersed aquatic vegetation. BioScience 43:86-94.

Derrenbecker, J. A. and R. R. Lewis. 1983. Seagrass habitat restoration Lake Surprise, Florida Keys. Pp. 132-154 In: R. H. Stovall (editor), Proc. 9th Annual Conf. Wetlands Restoration and Creation. Hillsborough Community College, Tampa, FL.

Durako, M. J. 1988. Turtle grass (Thalassia testudinum Banks ex Konig) - a seagrass. Pp. 504-520 In Y. P. S. Bajaj (Editor), Biotechnology in Agriculture and Forestry, Vol. 6, Crops II. Springer-Verlag, Heidelberg.

Durako, M. J. and M. D. Moffler. 1981. Variation in Thalassia testudinum seedling growth related to geographic origin. Pp. 99-117 In: R. H. Stovall (Editor), Proc. 8th Annual Conf. Wetlands Restoration and Creation. Hillsborough Community College, Tampa, FL.

Durako, M. J. and M. D. Moffler. 1984. A qualitative assessment of five artificial growth media on the growth and survival of Thalassia testudinum (Hydrocharitaceae) seedlings. Pp. 73-92 In: F. J. Webb (Editor), Proc. 11th Annual Conf. Wetlands Restoration and Creation. Hillsborough Community College, Tampa, FL.

Fonseca, M. S. 1990. Regional analysis of the creation and restoration of seagrass systems. Pp. 171-193 In: J. A. Kusler and M. E. Kentula (Editors), Wetland Creation and Restoration - The Status of the Science. Island Press, Washington, DC.

Fonseca, M. S., W. J. Kenworthy, and G. W. Thayer. 1982. A low cost transplanting technique for eelgrass (Zostera marina L.). Coastal Engineering Technical Aid No. 82-6. U.S. Army Engineer Coastal Engineering Research Center, Ft. Belvoir, VA.

Fonseca, M. S., G. T. Thayer, and W. J. Kenworthy. 1987. The use of ecological data in the implementation and management of seagrass restorations. Pp. 175-187 In: M. J. Durako, R. C. Phillips and R. R. Lewis (Editors), Proc. Symp. Subtropical-Tropical Seagrasses Southeastern United States. Fla. Mar. Res. Publ. No. 42.

Johansson, J. O. R. and R. R. Lewis. 1992. Recent improvements of water quality and biological indicators in Hillsborough Bay, a highly impacted subdivision of Tampa Bay. Science of the Total Environment, Elsevier, Amsterdam. pp. 1199-1215.

Koch, E. M. and M. J. Durako. 1991. In vitro studies of the submerged angiosperm Ruppia maritima: auxin and cytokinin effects on plant growth and development. Marine Biology 110:1-6.

Lewis, R. R., M. J. Durako, M. D. Moffler, and R. C. Philips. 1985. Seagrass meadows of Tampa Bay: A review. Pp 210-246 In: S. F. Treat, J. L. Simon, R. R. Lewis, and R. L. Whitman (Editors), Proc. Tampa Bay Area Sci. Inf. Symp. (May, 1982). Fla. Sea Grant Publ. 65. Burgess Publishing Co., Minneapolis, MN. 663 pp.

Phillips, R. C. and R. R. Lewis. 1983. Influence of environmental gradients on variations in leaf widths and transplant success in North American seagrasses. J. Mar. Tech. Soc. 17:59-68.

Roat, P., C. Ciccolella, H. Smith, and D. Tomasko (Editors). 1992. Sarasota Bay: Framework for Action. Sarasota Bay National Estuary Program, Sarasota, FL.

Thayer, G. W., W. J. Kenworthy, and M. S. Fonseca. 1984. The ecology of eelgrass meadows of the Atlantic coast: A community profile. U.S. Fish Wildlife Serv. FWS/OBS-84/02. 147 pp.

Thorhaug, A. 1974. Transplantation of the seagrass Thalassia testudinum Konig. Aquaculture 4:177-183.

Zieman, J. C. and R. T. Zieman. 1989. The ecology of the seagrass meadows of the west coast of Florida: A community profile. U.S. Fish Wildlife Serv. FWS/OBS-89. 155 pp.

COST EFFECTIVE MEASURES APPLIED TO ACHIEVE PERMIT COMPLIANCE AND HABITAT DIVERSITY AT A COUNTY OWNED MITIGATION SITE

Eric P. Fehrmann
Program Manager
PINELLAS COUNTY DEPARTMENT OF ENVIRONMENTAL MANAGEMENT
440 Court St.
Clearwater, FL 34616

ABSTRACT

Due to contract disputes, a Pinellas County mitigation site received virtually no maintenance for approximately two years. During that time the site had become infested with invasive species and the desirable vegetation was not thriving. The Southwest Florida Water Management District (SWFWMD) issued a letter of non-compliance for this site and the Pinellas County Department of Environmental Management was contacted by the Public Works/Engineering Design Section to assist in site remediation.

The site was assessed for hydroperiod, desirable and invasive species, herbaceous cover, and tree survival. Several options were discussed and the County decided to design an in-house multi-directional, non-destructive approach. It was able to gain the support of SWFWMD to try this most cost-effective option first. The County modified the hydroperiod and recontoured areas at the site. Post-construction monitoring revealed that the trees experienced a tremendous growth spurt as a result of the more favorable hydroperiod, and many species of wildlife previously not observed began to utilize the area. Invasive species, particularly cattail, declined markedly with no herbicide use.

Coordination with the Pinellas County Jail enabled inmate volunteers to plant over 500 trees as well as Pickerelweed, Spatterdock and 6900 Maidencane plants with no cost for labor. The site is functioning well at this time, and the Department will continue monitoring the site to ensure success.

INTRODUCTION

Studies conducted by contractors on behalf of Pinellas County, resulted in the design of a flood control strategy for the Spanish Oaks and Spanish Pines subdivisions located in northern Pinellas. In late 1985, construction was started on the Curlew Pond No. 3 flood storage facility to help alleviate flooding in these areas. Storm events, such as hurricane Elena, may have helped speed the project along.

Curlew Pond #3 receives water from Curlew Pond No. 1, Curlew Pond No. 2 and from the Spanish Pines subdivision. Water eventually outfalls into Curlew Creek. The drainage basin covers approximately 223 hectares. Curlew Pond No. 3

required extensive excavation to match grades with the two existing ponds. Nearly 400,000 cubic meters of material were excavated to construct the site. Much of the site was situated over green marine clays, which contributed to some of the unique hydrological characteristics of the site.

In 1987, after the flood control project was completed, a 1.52-hectare forested mitigation site was required for 0.6 hectares of maple hardwood impact associated with construction of the Jerry Lake Weir. Mitigation was also required for minor impacts associated with a sidewalk for C.R. 1. The permitting agencies agreed that Pond No. 3 was a suitable site for the mitigation. A total of 3.0 hectares were planted - somewhat more than what was required.

The overall pond site encompasses 5.4 hectares of land. The mitigation site is in the northern 3.0 hectares of the pond, the flood attenuation and storage, and the underdrain system, is located in the southern 2.4 hectares.

The original flood design was performed by a contractor, but due to irreconcilable differences, the contract was terminated. The pond was eventually designed by the Engineering Department's Planning & Programming Division.

The pond is located in an area that had an average elevation of 12 meters NGVD. The area was vegetated with a sand hill community that included sand pine, sand live oak and slash pine and was fairly open under the canopy. The Parks Department spaded about 400 sand live oaks which were transported to Howard Park in Tarpon Springs, and about 400 sand pines, which were relocated to Sand Key Park in Clearwater.

The pond was excavated about 6 meters into the soil to match drainage pipe inverts from Pond No. 2. In many areas, the bottom 4 meters was composed of green gumbo marine clay. An underdrain system was then installed in the southern portion of the site.

The pond was designed with a horseshoe shape to save a stand of mature live oaks with 0.5 m + DBH's. The oak stand is used by a variety of wildlife, including hawks, owls and small mammals.

The original pond design did not specify an elevated berm separating the mitigation area from the underdrain, but there was a concern by the permitting agencies that the underdrain system would draw down the water table in the mitigation area. The berm was constructed at the same elevation as the outfall structure, 4.14 meters NGVD. The Pinellas County Construction Division was worried that water would stage up behind the berm and cause overhydration in the mitigation area, and installed an equalizer pipe in the center of the berm. The permitting agencies restated their concerns about water level drawn down, and ordered the pipe removed.

During planting of the mitigation area, the planting consultant warned that the water appeared too deep and that the plants and trees would not likely thrive at the site, but no action was taken at that time, due to the previously stated agency concerns about low water levels.

Anywhere from 10 cm to 20 cm of muck was placed in the mitigation area as a growing substrate. Approximately 5,000 trees were planted at the site on 3.1-meter centers and consisted of a mixture of cypress, maple, tupelo, laurel oak and sweet bay. Trees were planted in groupings of like species. No herbaceous mitigation was required for the permit, but pickerelweed, arrowhead, bulrush, juncus and blue flag iris were planted, although unfortunately, also on 3.1-meter centers.

Due to the large amount of unvegetated area and the hydrologic conditions, invasive species began to proliferate. The general contractor had hired an environmental contractor to maintain the pond and provide monitoring reports. When the funds allocated by the general contractor were spent on maintenance (after only 6 months) the subcontractor stopped performing maintenance, or doing mitigation reports because they were not getting paid anymore.

No maintenance was performed for approximately two years, as the County was trying to find a way to hold the general contractor responsible for maintenance. Unfortunately, there was no legal recourse to make either company responsible to perform the work. The County writes contracts differentially now.

The pond's conditions were observed by SWFWMD in late summer of 1991 and the Engineering Department received a letter of non-compliance shortly thereafter. The Engineering Design Section contacted the Environmental Support Section and requested assistance in assessing the pond and determining the best and most cost-effective option.

STUDY SITE

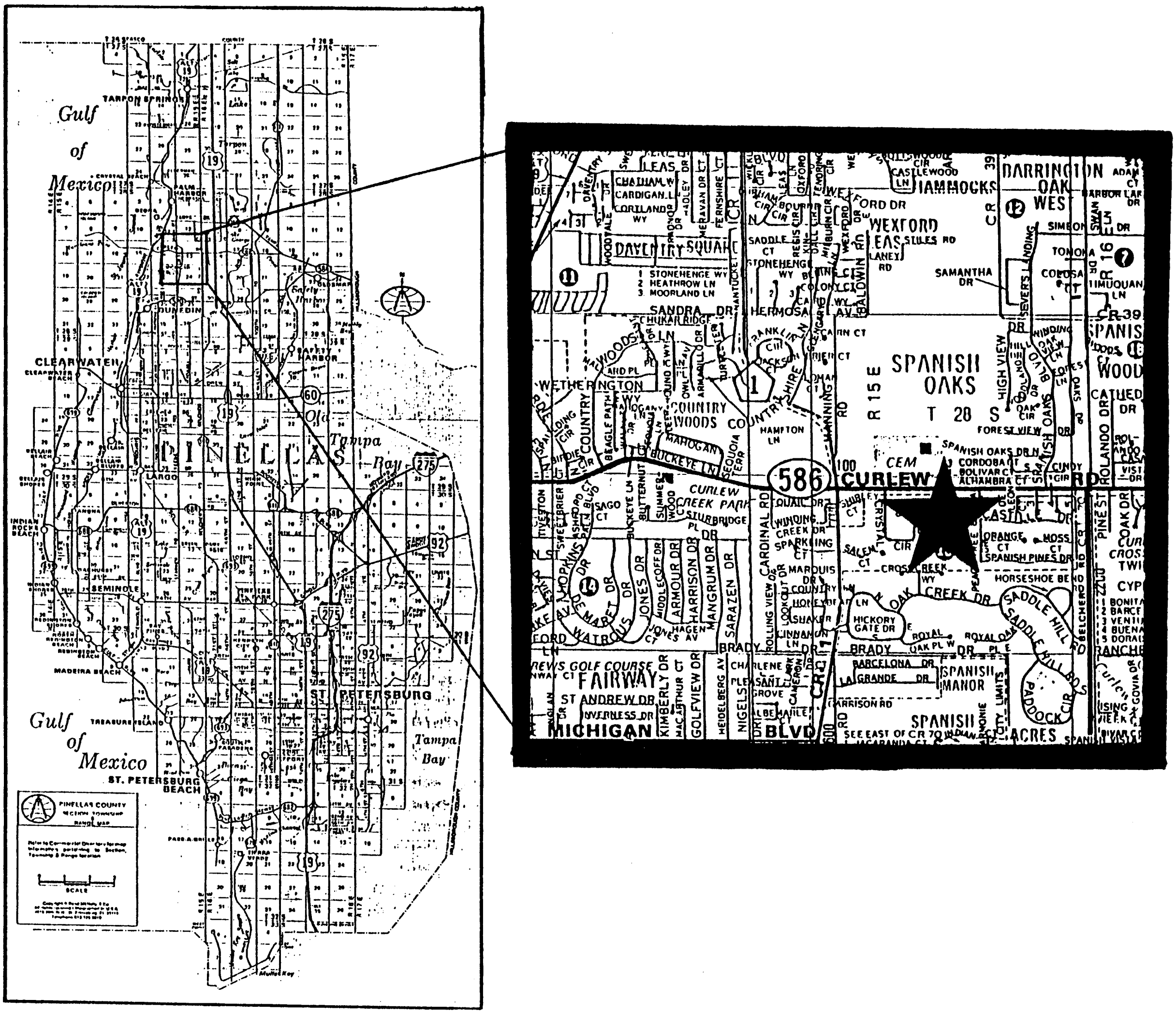


Figure 1. The study site is located in northern Pinellas County. The area is adjacent to sand ridge, aligned north to south, and underlain by green marine clays.

MATERIALS AND METHODS

The first site visit was performed in September of 1991, at the request of the Engineering Design Section. The assessment team went out to gain an overall perspective of the site and try to make a general assessment of the conditions.

The mitigation area was staged up to the berm with water and the team could only look around the edge of the pond. The trees looked very stressed and had few leaves. Invasive species seemed to be everywhere and coverage approached

90%. Water levels seemed to be slightly lower in the south side of the pond, but it was also prime cattail habitat and, in fact, supported a near monoculture of this invasive species. Only three small clumps of willow were present.

The Environmental Support Services Section coordinated with Engineering Design and the Construction Division to devise a better way to assess the pond. It was decided that the best option would be to drain as much water as possible out of the mitigation area by tapping into the main underdrain pipe. This would afford us the opportunity to better assess the vegetation within the pond and examine the muck depth for uniformity. At this time, the Environmental Support Section authored a 26 point memo outlining Pinellas County's options on pond planting and hydrology modifications.

SWFWMD met with county representatives at the site and agreed to let the county drain the pond as part of the plan for bringing the pond into compliance. In November, the underdrain main pipe was accessed and rim ditch construction started along the western side of the pond to help drain off the standing water.

It took about a month and a half for the pond to drain and dry sufficiently to where no one could walk around with rubber boots without going waist deep into the muck holes. In early January of 1992, members of the Environmental Support Services Section, Engineering Design, Construction and Planning and Programming returned to the site to perform another assessment.

Once among the vegetation, it was noted that many of the trees had died, probably from excessive inundation. A surprising amount of trees had survived, though, and they seemed to be putting out new growth. Almost all the trees were stunted and had grown adventitious roots. Some of the trees were easily uprooted because the muck had not solidified and the trees had not anchored themselves, but most of the cypress seemed to have fared better than the other species.

Most of the cypress looked like baseball bats; they were short, but very thick at the base. The trees were probably trying to either stabilize or maximize gas exchange by maximizing basal growth and lenticil surface area. It was at this time that the site appeared to have much more potential for long-term success than previously thought.

Several options were discussed at this time to develop a plan to get the mitigation site back into compliance:

1. Herbicide the whole pond and start over. This would be extremely expensive and decent existing habitat would be destroyed. There is no guarantee that the cattails will not reestablish themselves in the area, either by seeding or vegetative reproduction.

2. Herbicide the south cattail pond, bury the cattails in a hole on site, bring in new muck, and move the viable trees and herbaceous vegetation from the north pond to the south pond and plant heavily with new herbaceous species to suppress invasives; then replant the north side and nurture for banking. Again, this would be very expensive and time consuming.

3. Use a combination of hand-labor and machinery to clear out the invasives from between the trees in the north mitigation site. Selectively herbicide remaining invasives. Recontour the site for better invasive suppression and better habitat value, lower the weir elevation to keep water levels lower, and implement a stringent maintenance program. Construction stated that past hand labor activities proved fruitless on other projects and heavy machinery could not operate on this type of environment. There were also doubts about the ensuring a stringent maintenance program if it was to be contracted out.

None of these options appeared satisfactory and discussion continued about the basic facts of the Site.

1. It appeared that no matter how far the outflow weir elevation was reduced, the pond would not totally dry out, because of the natural clay lining and the constant groundwater seep from the side slopes into the pond.

2. The permit had no requirement for success for herbaceous species, so as long as the trees could outcompete the invasives, they would eventually shade them and kill them off.

3. Almost twice the required amount of mitigation area was planted than was required for the Jerry Lake Weir Project.

4. There was a sizable wildlife population utilizing the pond.

5. Stormwater treatment was still being accomplished.

SWFWMD was invited back to the site to view the pond in it's drawn-down condition. SWFWMD met with the same evaluation team at the site in the beginning of February, 1992.

By this time the trees had started to flush out and the cattails seemed to be somewhat suppressed by the drier conditions. The team walked the site and pointed out how well most of the surviving trees were doing. It was also interesting to note the doughnut-shaped growth rings formed from the widely spaced herbaceous plantings. Because the plants could not form contiguous stands, they grew in rings outward from single plants, with dead plants in the center and new growth on the outside of the ring.

The current habitat value of the site was also pointed out, and it was agreed that it would not be environmentally sound to lose the habitat and trees if we had to remove all vegetation from the site.

Further discussion with SWFWMD resulted in approval to allow the site to grow out over the summer, to ascertain if the trees would actually gain height and canopy cover or remain stunted. This was done in order to help determine whether overinundation or the clay was causing the stress response. Although the growth spurt looked promising, SWFWMD wanted to make sure that the trees were not permanently stunted. SWFWMD requested a long-term plan based on tree growth success.

Based on the discussion, if the trees started to increase in height, SWFWMD would consider allowing the county a greater than 10% invasive species coverage on a temporary basis. The county would also have to replant trees that had died. To document tree growth, Environmental Support Services tagged and measured selected trees for monitoring of height and DBH.

The underdrain access would have to be sealed again to ensure long-term tree growth and survival and it was known that the weir height needed to be reduced; but, it was not known by how much. The assessment team returned in April, 1992 to do some survey work to determine if there were large elevation changes in the pond, or if the surface was relatively flat.

The elevation of the weir was also measured to see if it was installed correctly and the berm was surveyed to determine if it was, in fact, hindering water flow. The height of adventitious roots on the trees in both the mitigation area and the water treatment area were surveyed to determine standing-water depth in each of the areas.

Surprisingly, the pond bottom was fairly level. The elevation of the adventitious roots on the trees were very consistent and measurements were within several centimeters of each other. Even more surprising, the treatment pond's trees had adventitious roots at the same elevation as the mitigation site, indicating that the underdrain system was not drawing down the system as was thought when the pond was first built.

The survey indicated that the weir was constructed at the plan elevation and the berm had a similar elevation as the weir.

It was decided that the weir elevation should be reduced by about 15 centimeters to maintain a summer stage-up of water, with a bleed-down oriface 40-46 centimeters below that to maintain a winter drawdown to stimulate growth and suppress invasives. A debris skimmer would be added to the outfall structure.

It was also agreed that a deep winding channel should be cut through the middle of the mitigation area to get more open water habitat for waterfowl and wading birds. In addition, sediment sumps were designed into the project. The spoil would be used to construct slightly elevated areas to plant the replacement maples. A temporary outfall was designed by the Engineering Design Division.

Construction on the rim ditch and meandering channel began in June of 1992, and took approximately eight weeks to complete. The topsoil (muck) was scraped off the construction area and placed in the maple planting area, and smoothed into a shallow-dish shape to help hold water and give the replants a healthy start. The clay underlayer was hauled off site. Once the site stabilized, a plug was installed in the underdrain pipe and a temporary riser was constructed to stage the water up to an elevation approximately 15 centimeters lower than the original weir elevation. A permanent gate valve was installed on the underdrain to facilitate easier draining of the site for future maintenance. A catwalk and staff gauge was also installed. All excavation work was performed by county personnel from the Highway Department.

RESULTS AND DISCUSSION

The water reached the design elevation within a few days and the channel filled to the top of bank. Vegetation started recruiting immediately along the shore banks, but no cattails appeared to be invading the area. Within a short period of time, a varied fish population developed, including some larger species. Many reptiles thrived in the new habitat, and a river otter had taken up residence at the site and was observed in the channel every time the site was visited. Many species of small mammals took advantage of the superior site conditions and extended their range throughout the site. Wading birds and waterfowl were also observed frequenting the newly-created habitat.

The Environmental Support Services Section performed a baseline assessment of the trees on the site, and allowed the site to remain in the staged-up conditions through the fall of 1992. Several monitorings for tree growth were performed in the succeeding months and tree growth was phenomenal. Trees were exhibiting 20% to 30% growth in height and DBH after only 6 months of monitoring (Figure 2). Relative biomass of the trees appeared to have more than doubled. The trees were much healthier and had branched significantly with a tremendous rate of leaf growth. In an interesting side note, the trees within the cattails were as tall, or taller, than those in the more open areas. More research need to be performed to determine if invasive species competition is as much of a problem for tree growth as previously assumed.

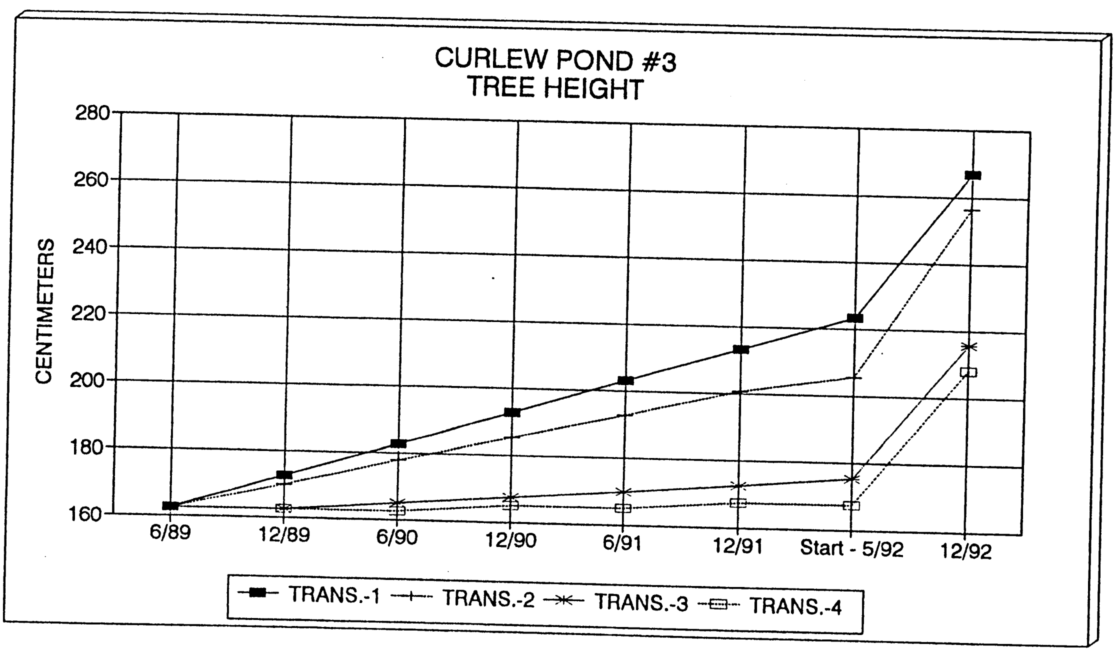


Figure 2.

A planting plan was designed that would replace the trees that had died, as well as fill in some of the areas that had not been planted previously. In addition, some herbaceous plants were specified to vegetate the banks and middle of the channel. Over 500 trees were specified with species centered on red maple (Acer rubrum), but included cypress (Taxodium distichum), tupelo (Nyssa aquatica), pop ash (Fraxinus caroliniana), laurel oak (Quercus laurifolia) and sweet bay (Magnolia virginiana). Herbaceous plants centered on pickerelweed (Pontedaria cordata), which seemed to be doing well in other areas of the site, and spatterdock (Nuphar luteum).

The assessment group decided to perform replanting of trees in November of 1992. To attempt to save replanting labor costs, the Environmental Support Section contacted the Pinellas County Jail to determine if jail trustees could perform the planting. The jail agreed to supply trustees, a corrections officer, and a landscape instructor to oversee the work crew. Surprisingly, when the planned date arrived there were not enough trustees to form a work crew.

The replanting project was delayed until January of 1993. At this time the jail was able to supply a work crew and the plant material was ready for delivery. As a bonus, 8,000 maidencane (Panicum hemitomon) plants were available that the County had purchased, but could not use on another mitigation site (a Bald Eagle was nesting in the work area and site preparation was delayed).

The Highway Division excavated holes for tree planting with a mini backhoe to save manual labor effort for actual planting. The weekend prior to the planting date, a heavy thunderstorm inundated the work area and planting was delayed another week.

Finally, all parties were able to coordinate and perform the planting. The site conditions were still poor from the rain, but progress was swift. The trustee work crew was split in half, with one crew planting maidencane and the other carrying trees into the planting site and installing them. Under the direction of the Environmental Support Section, the trees were planted according to their hydrological requirements, as well as in an aesthetically pleasing random manner. Maidencane was planted in a "floodplain" type area near the outfall structure, as well as in the voids between the trees. The work was completed within one week.

Follow-up observations in March of 1993 revealed that all the newly-planted trees had sprouted new growth in spite of a severe unnamed storm that occurred only a couple of weeks earlier. The water level returned to the design elevation and although tree species appeared fairly compatible with hydrological levels and characteristics, it was decided to reduce the design water elevation 7.5 centimeters. A quarterly monitoring report was also performed in March. Tree height and DBH was measured, but there was minimal growth. A sizable growth increase is expected to be recorded during the next quarterly report, based on tree bud size and quantity.

CONCLUSION

Success of a mitigation project cannot be simply measured and judged using numerical indices or limited variables. A site can be in compliance with permit conditions but not a success. Conversely, a site can be out of compliance, but be an environmental success. Success of the mitigation site lies with the establishment of a functioning wetland community, including a wide variety of floral and faunal species. Designers must learn to look beyond permit conditions when creating wetlands, and create systems made up of a variety of habitats that complement each other.

ACKNOWLEDGEMENTS

Appreciation is given to Jim Meyer and Steve Robinson of the Pinellas County Environmental Support Services Section; to Kevin Murren, Rudy Garcia, and James Richter of the Pinellas County Public Works Department; and Karen Gruenhagen of SWFWMD for their hard work and dedication to make this project a success.

WETLAND CREATION IN AN URBAN BUSINESS PARK

Albert S. Garlo
NORMANDEAU ASSOCIATES, INC.
25 Nashua Road
Bedford, New Hampshire 03110

ABSTRACT

During the spring of 1990, two wetland basins (ponds) with emergent/shrub borders of approximately four acres were created as compensation for wetland filling in an adjacent business park. This mitigative measure was a condition of the U.S. Army Corps of Engineers permit. The wetland creation occurred on poorly-drained soil which had been in agricultural use for many years. The hydrology is primarily groundwater fed with a minor input from adjacent surface runoff. The water levels of the basins fluctuate 2-3 feet seasonally. Many species of aquatics, emergents and shrubs were planted. As there were no existing surface water or emergent wetlands nearby, the present wetland vegetation either resulted from the plantings or invasion of native species from some distance. After four growing seasons, it appears that although many of the plantings have survived, the emergent edge, unless further managed, will become dominated by species that naturally pioneer wetlands through seeding and grow aggressively in this region (i.e., cattail, purple loosestrife, spikerush, water-plantain, willow and cottonwood). Because purple loosestrife is such an aggressive plant in disturbed wetlands in this region, its spread into this wetland might have been slowed if cattail had been initially planted to establish a dense cover.

INTRODUCTION

During the spring of 1990, two created wetland basins of approximately four acres were incorporated into the development of a business park in Colchester, Vermont pursuant to permit conditions of the U.S. Army Corps of Engineers (COE). The emergent zone was extensively planted and the wet meadow seeded with several herbaceous wetland species. This report describes the plant community after four growing seasons and assesses the results of the herbaceous marsh plantings.

SITE DESCRIPTION

Prior to excavation of the wetland basins, this site was primarily level agricultural land. In depressions the deep sandy, poorly-drained Scarboro and Au Gres soil series (USDA 1974) were determined by the COE to be wetland. These were typical "dryer end" wetlands characterized by hydric soils and facultative plant communities. Young stands of willow (Salix sp.), quaking aspen (Populus tremuloides), white ash (Fraxinus americana) and grey birch (Betula populifolia) had colonized after disturbances caused both by agriculture and the utilization of this site as a staging area for materials during the construction of an interstate highway. As the wetland basins were excavated partially within these COE jurisdictional wetlands, technically a portion of the mitigation was considered wetland enhancement and the area excavated from upland considered wetland creation. In any case, there was not a nearby source for most herbaceous wetland plants, except cattail (Typha latifolia) and purple loosestrife (Lythrum salicaria) in roadside ditches.

Monitoring well data for the 12 months prior to construction indicated a groundwater table generally 2-4 feet below surface, with a seasonal fluctuation of approximately 2 feet. Two wetland basins were constructed (north and south ponds), utilizing groundwater as the primary hydrology augmented with surface drainage from the site development. During flooded conditions, surface water from the south pond can overflow through a wetland swale to the natural surface drainage of the surrounding area. The wetland basins, in the deepest areas, are approximately 5 feet deep and grade gradually (1:10 slope) to wet meadow and upland meadow. A sinuous shoreline and an island to enhance wildlife value were incorporated in the design. The north and south wetland basins, as they appeared in September, 1993, are shown in photos 1 and 2, respectively.

During the excavation of the basins the topsoil was stockpiled and finish-graded over the wetlands. This material was a sandy loam. The plantings were done during June, 1990 in this sandy loam substrate with no fertilizer amendment. The herbaceous species planted relative to hydric zone are given in Table 1.

A total of 1,450 shrubs of several native species were planted above the high water elevation. These were container-grown nursery stock and survival was excellent, except for small areas of mortality due to late spring flooding. All exposed area was hydroseeded with a grass/wildflower mix consisting of the following species: Canada anemone (Anemone canadensis), columbine (Aquilegia canadensis), swamp milkweed (Asclepias incarnata), New England aster (Aster novae-angliae), Canada tick trefoil (Desmodium canadensis), joe-pye weed (Eupatorium maculatum), great St. Johnswort (Hypericum pymridatum), cardinal flower (Lobelia cardinalis), smooth penstemon (Penstemon digitallis), big bluestem (Andropogon gerardi), and switchgrass (Panicum virgatum).

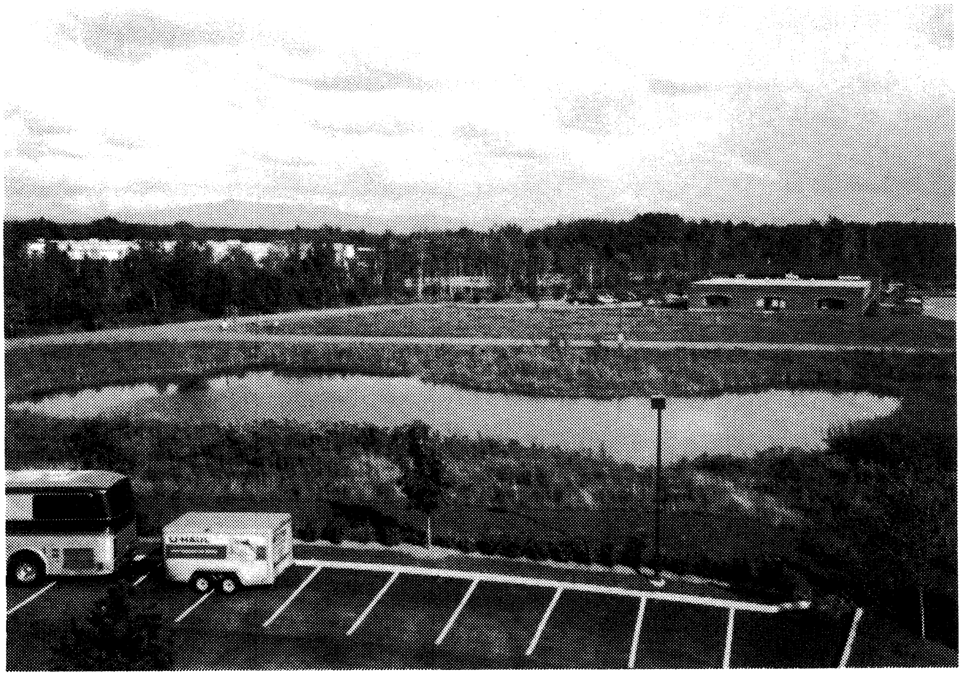


Photo 1. North wetland basin, September, 1993.

Photo 2. South wetland basin, September, 1993.

Table 1. Wetland Plantings by Water Depth, Species and Quantities.

Water Depth (ft)	Species	Quantities
>3	Sago pondweed (Potamogeton pectinatus)	3,500 tubes
2-3	Waterlily (Nymphaea odorata)	900 tubes
1-2	Arrowhead (Sagittaria latifolia)	1,800 tubes
	Giant burreed (Sparganium eurycarpum)	1,100 plants
	Pickerel weed (Pontederia cordata)	700 plants
0-1	Lotus (Nelumbo lutea)	500 seed
	Blue flag (Iris versicolor)	500 plants
	Sweet flag (Acorus calamus)	1,000 plants
	Bulrush (Scirpus acutus)	1,000 roots
	River bulrush (Scirpus fluviatilis)	1,000 roots
	Arrow arum (Peltandra virginica)	1,000 plants

Photo 3 shows the placement of the 1.0 m^2 quadrat in Zone B in vegetation typical of the wetland.

METHODS

Vegetation sampling occurred September 7, 1993 at the end of the fourth growing season. The water level was still quite low, about 18 inches below average spring water levels. Fifteen sets of three 1.0 M^2 plots, were randomly located around the south wetland basin. Each set of three plots sampled three hydric zones. Zone A, placed at the water's edge, sampled vegetation which is exposed only during late summer. Zone B, place 10 feet up from the water's edge, is exposed during most of the summer and fall, but is flooded during winter and spring. Zone C, placed 20 feet from the water's edge, is flooded only during very high water levels, but is saturated during much of the growing season. Randomization was accomplished by laying out a tape measure around the pond and consulting a random number table to choose the location for each plot set. The average distance between each plot set was approximately 50 feet. Photo 3 shows the placement of the 1.0 M^2 quadrat in Zone B in vegetation typical of the wetland. Vegetation sampling included species identification and a visual estimate of percent cover for each species, and the plot as a whole.

RESULTS AND DISCUSSION

The plant species observed and the number of plots in which they occurred for each hydric zone (out of a total of 15 plots per hydric zone) are given in Table 2. The vegetation sampling confirmed the visual observation of the vegetation development over four growing seasons. Of the herbaceous marsh plantings, most of which are still surviving on the site, only pickerel weed has established itself with a degree of success. Although the deep water zone (>3 feet) was not included in the sample plots, it was obvious that waterlily and sago pond weed, which were also planted, are well established. The abundance of New England aster, swamp milkweed, cardinal flower, blackeyed susan, and joe-pye weed indicates these species were probably established from the wildflower seed mix. Big bluestem (Andropogon gerandi) is uncommon and was most likely established from the seed mix.

The fourth growing season was the first year that there was not substantial bare ground around this wetland. Only one plot, which was in Zone A, had less than 100 percent total cover. The visual estimates of percent cover of species within the plots was generally correlated well with species frequency. Most plots had between 9 and 12 species. The lowest number of species in a plot was four, which was a plot dominated by low-growing spikerush. Even in the dense cattail/purple loosestrife plots these species combined accounted for less than 50% of total plot cover, except for one plot which was 60%. The wetter zones (A and B) are largely colonize by the annuals (beggars tick, smartweed, umbrella sedge and barnyard grass) in relatively stunted or low-growing forms. These zones are still open for the spread of cattail and purple loosestrife, which are the most aggressive colonizers at this wetland. Animal activity, especially muskrat, ducks, and wading birds, is very evident in this wetland, which will affect the future vegetation development.

Within the wetland creation literature there are surprisingly few papers that focus on the long-term ability of herbaceous planting to influence the eventual

Table 2. Number of 1.0 M^2 plots in which each species occurred out of a total of 15 plots for each hydric zone: Zone A exposed during late summer and draughty condition otherwise flooded; Zone B exposed during summer flood during winter and spring; and Zone C saturated soil, seldom flooded.

Plant Species	Frequency Zone A	Zone B	Zone C
Cattail (*Typha latifolia*)	4	10	4
Purple loosestrife (*Lythrum salicaria*)	5	14	6
Soft rush (*Juncus effusus*)	-*	6	6
Rush (*Juncus articulatus*)	2	2	-
Rush (*Juncus dudleyi*)	-	1	7
Water plantain (*Alisma subcordatum*)	6	1	1
Spike rush (*Eleocharis smallii*)	9	9	-
Spike rush (*Eleocharis acicularis*)	6	2	-
Umbella sedge (*Cyperus strigosus*)	7	7	5
Umbella sedge (*Cyperus rivularis*)	3	3	-
Smartweed (*Polygonum lapathifolium*)	10	-	-
Smartweed (*Polygonum saggitatum*)	2	1	2
Beggar-ticks (*Bidens frondosa*)	12	10	-
Water-purslane (*Ludwigia palustris*)	9	-	-
Pickerel weed (*Pontederia cordata*)	5	-	-
Hard stem bulrush (*Scirpus acutus*)	3	2	-
Rice cut-grass (*Leersia oryzoides*)	3	1	1
Barnyard grass (*Echinochloa crusgalli*)	7	6	-
Panic grass (*Panicum dichotomiflorum*)	3	1	-
Panic grass (*Panicum depauperatum*)	2	3	-
Swamp milkweed (*Asclepias incarnata*)	1	1	-
Common horsetail (*Equisetum arvense*)	1	5	1
Giant burreed (*Sparganium eurycarpum*)	-	1	-
Willow (*Salix sp.*)	-	5	1
Waterhorehound (*Lycopus americanus*)	-	3	2
Queen Anne's lace (*Daucus carota*)	-	1	6
Common plantain (*Plantago major*)	-	4	-
Red clover (*Trifolium pratensis*)	-	1	3
Bird's foot trefoil (*Lotus corniculatus*)	-	2	5
Common ragweed (*Ambrosia artemesiifolia*)	-	5	2
Field-sow thistle (*Sonchus arvensis*)	-	1	1
Cardinal flower (*Lobelia cardinalis*)	-	-	1
Blue vervain (*Verbena hastata*)	-	-	2
New England aster (*Aster novae-angliae*)	-	1	10
Small white aster (*Aster vimineus*)	-	-	3
Tall goldenrod (*Solidago altissima*)	-	-	5
Sweet goldenrod (*Solidago odora*)	-	-	1
Lance-leafed golden rod (*Solidago graminifolia*)	-	3	9
Black-eyed Susan (*Rudbeckia hirta*)	-	-	3

Table 2 (Continued)

Plant Species	Zone A	Frequency Zone B	Zone C
Joe pye weed (*Eupatorium maculatum*)	-	2	2
Bonset (*Eupatorium perfoliatum*)	-	3	3
Bluejoint grass (*Calamagrostis canadensis*)	-	2	4
Fowl meadow grass (*Poa palustris*)	-	-	5
Red fescue (*Festuca rubra*)	-	-	2
Grass (*Muhlenbergia mexicana*)	-	-	1
Nodding ladies' tressses (*Spiranthes cernua*)	-	-	2
Sedge (*Carex vulpinoidea*)	-	-	1
Cow vech (*Vicia cracca*)	-	-	3
Aliski clover (*Trifolium hybridum*)	-	1	-
Slender gerardia (*Gerardia tenuifolia*)	-	2	-
Big bluestem (*Andropogon gerardi*)	-	-	2
Sensitive fern (*Onoclea sensibilis*)	-	-	1
Rough bedstraw (*Galium asprellum*)	-	-	1
Wool grass (*Scirpus cyperinus*)	-	1	3
Bentgrass (*Agrostis prenans*)	-	1	-
Marsh fern (*Thelypteris palustris*)	-	-	1
Speckled alder (*Alnus rugosa*)	-	-	1
Meadowsweet (*Spirea latifolia*)	-	1	-
Dwarf cinquefoil (*Potentilla canadensis*)	-	-	1
Reed canary grass (*Phalaris arundinada*)	-	-	1
Corn grass (*Panicum clandestinum*)	-	-	1
Cottonwood (*Populus deltoides*)	-	3	1
Grey birch (*Betula populifolia*)	-	-	1
Red maple (*Acer rubrum*)	-	1	-

* Dash indicates the species was not found in this plot.

dominance of vegetation in created wetlands. The Wetland Demonstration Project (Southern Tier Consulting, 1987) documented "mixed results" from several marsh plantings for two growing seasons and generally recommended planting. Crabtree, et al., (1992) evaluated wetlands created by the Department of Transportation in 14 states. This evaluation did not focus on herbaceous plantings. However, in most instances either plantings were not done, or they had no influence on the dominant vegetation at the time of evaluation. Most of the information on planting success or failure probably exists as COE mitigation monitoring reports waiting to be researched, evaluated and summarized. Because of the presence of purple loosestrife in this region, its spread into this wetland might have been reduced if cattail had been initially planted to establish a dense cover. Cattail is not considered to be a problem-invasive plant in Vermont, while it is of high wildlife habitat value for many species.

This wetland is now at its most aesthetically-appealing stage for an urban setting. The wildflower meadow and ponds make wading birds and other wildlife easy to observe. During data collection, a great blue heron would move from one pond to the other to keep a few hundred yards distance. Healthy wetlands are dynamic, changing environments (Mitsch and Gosselink, 1986). Without management, this wetland will become less attractive, both from a wetland science and aesthetic point of view. Without management, in a few years, purple loosestrife is likely to be the dominant vegetation from the open water up to the point where it is shaded out by trees (cottonwood, willows, and grey birch). To avoid this, it is intended to hand-weed the purple loosestrife until cattail and the other emergent vegetation are sufficiently dense to retard its spread. Mowing once every few years and removal of tree invasion within the shrub plantings can keep an open and aesthetically-appealing setting.

LITERATURE CITED

Crabtree, A., E. Day, A. Garlo and G. Stevens. 1992. Evaluation of Wetland Mitigation Measures, Vol. I. FHWA-RD-083. Federal Highway Administration. McLean, Virginia.

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

Southern Tier Consulting. 1987. Wetland Demonstration Project. New York Department of Transportation and Federal Highway Administration.

United States Department of Agriculture. 1974. Soil Survey of Chittenden County. Soil Conservation Service. 149 pp + maps.

AMENDMENTS TO MICHIGAN'S WETLAND PROTECTION ACT, THE ONLY STATE-ASSUMED SECTION 404 WETLANDS PROGRAM

Eugene Jaworski, PhD
Department of Geography-Geology
EASTERN MICHIGAN UNIVERSITY
Ypsilanti, Michigan 48197

Gerald A. Fisher, Esq.
KOHL, SECREST, WARDLE, LYNCH, CLARK and HAMPTON
30903 Northwestern Highway
Farmington Hills, Michigan 48334-2516

ABSTRACT

In December, 1992, the Governor of the State of Michigan signed into law two amendments which have modified P.A. 203 of 1979 (Wetlands Protection Act). The two amendments contained several important provisions, though basically causing local wetland ordinances to regulate wetlands consistent with the State statute. By means of P.A. 203, the State of Michigan, via the Department of Natural Resources, has been administering the Section 404 wetlands program since August, 1984. However, during the past few years, anti-wetland regulation forces consolidated sufficient strength to force the amendment of the Wetland Protection Act. Comprised of the Michigan Home Builders Association, developers, real estate firms and others, this group secured introduction and considerable backing for a Senate Bill which called for the elimination of local wetland ordinances, wetland review boards, and wetland buffers. In the committee meetings, which were moderated by a state senator, compromise negotiations were undertaken which focused on establishing a standard wetland definition and use of a single wetland permit application, among other things. The compromise was eventually reached because of a strong response from local governments and environmentalists seeking to preserve this significant regulatory authority. Nevertheless, the wetland use permit system, on the state level, is hampered by staffing and fiscal constraints. Meanwhile, on the municipal and township level, the more pro-active communities are attempting to implement measures to respond to the act amendments to avoid substantive regulatory impediments. To these local communities, wetland preservation is deemed important for environmental reasons, and local residents often regard wetland preservation as a quality of life issue. In contrast, impacted landowners and developers often view wetland protection as a taking and a tool to effect no growth.

INTRODUCTION

Support for wetland preservation in the State of Michigan is inconsistent among the various public and private interest groups. Private developers, realtors, as well as some engineering firms, in particular, have expressed complaints about the lack of accurate wetland maps and the difficulty of obtaining necessary wetland permits (Tilton, 1988). In 1992, this wetland opposition was translated into a number of legislative initiatives which were focused, in large part, on the elimination of local wetland ordinances. Senate Bill 522, which was backed by the Michigan Homebuilders Association, called for the preemption of wetland regulation at the state level and would have severely curtailed local wetland ordinances.

In December, 1992, the Governor of the State of Michigan signed two amendments which modified Michigan's Wetland Protection Act. These two 1992 amendments contained several important changes, though basically forcing local wetland ordinances to regulate wetlands consistent with the State Statute. Public Act 203 of 1979 (Goemaere-Anderson Wetland Protection Act), is the primary legislation which protects wetlands in Michigan. P.A. 203 defines wetlands identical to the federal wetlands manual, and requires a permit when regulated wetlands are to be developed. Wetlands which are contiguous to lakes, streams, drains, and ponds are regulated, as well as those greater than 5 acres in size (Wyckoff, 1988). There are additional jurisdictional criteria, and in counties of less than 100,000 people, non-contiguous wetlands are not regulated until a wetlands inventory is completed.

ASSUMPTION OF THE 404 PROGRAM

The U.S. Environmental Protection Agency (U.S. EPA) is responsible for the administration of the Clean Water Act, including Section 404 of the Clean Water Act of 1977 (33 U.S.C. 1344). Section 404 provides the basis for the primary federal program governing activities in wetlands. In August, 1984, the U.S. EPA transferred the administrative authority of the Section 404 Wetlands Program to the State of Michigan. The Michigan Department of Natural Resources (Michigan DNR) was able, in part, to assume authority of the Section 404 wetlands program because of the similarity between Section 404 regulations and Michigan's P.A. 203. In addition, the State of Michigan had other related legislation in place, such as P.A. 346 (Inland Lakes & Streams Act), and demonstrated a capability to effectively administer the wetlands program (Cwikiel, 1992).

In practice, the Michigan DNR and the U.S. Army Corps of Engineers jointly administer the federal wetlands regulatory program. The regulatory authority of the Corps of Engineers stems from Section 10 of the Rivers & Harbors Act of 1899 and Section 404 of the Clean Water Act of 1977. The Corps of Engineers has retained jurisdiction over wetland activities occurring in Great Lakes coastal waters and connecting channels, as well as in major tributaries to the upstream limit of navigability (Ibid.). In places where the Corps of Engineers has jurisdiction, wetland permits are required from both the Corps of Engineers and Michigan DNR. Michigan is currently the only state which has assumed authority over the Section 404 Wetlands Program. Nevertheless, the U.S. EPA maintains oversight authority over major wetland projects, including those involving more than 10,000 cubic yards of wetlands fill.

OVERVIEW OF MICHIGAN'S WETLANDS ACT

Wetland Jurisdiction Criteria

In accordance with P.A. 203 of 1979, as amended, including the Administrative Rules which were promulgated in June, 1988, the following jurisdictional criteria apply to wetlands in counties over 100,000 persons (Dean & Jaworski, 1991):

- All wetlands contiguous to lakes, streams, drains, and ponds, including the Great Lakes.

- All non-contiguous wetlands greater than 5 acres in size.

- Non-contiguous wetlands smaller than 5 acres, provided one of the conditions listed below are met:

  ~ Wetland is situated within 500 feet of lake, stream, drain or pond, unless it can be proven that no surface or groundwater connection exists.

  ~ Wetland contains one acre or more of permanent open water.

  ~ The wetland in question is essential to the preservation of the natural resources of the State, and the Michigan DNR has so notified the landowner.

Wetland Permit Process.

Beginning in 1983, the Michigan DNR started to vigorously enforce the permit requirements of P.A. 203, as well as to cite wetland violations. During these early years, the permit process did not proceed very well, partly because reliable maps showing wetland locations did not exist and a trained cadre of private wetland consultants was not yet available. Although the Michigan DNR had collaborated with U.S. Fish & Wildlife Service in producing the National Wetland Inventory (NWI) Maps, these maps were generated at a scale of 1:24,000 and only approximately 75% of the State was inventoried. After it assumed authority over the Section 404 Wetlands Program in 1984, the Michigan DNR initiated an in-house wetlands training program. By June, 1988, when the Administrative Rules of P.A. 203 were issued, the Michigan DNR's wetlands permit program had matured.

At present, a wetland project typically begins with a wetland delineation by a private wetland consultant or by a public wetland specialist from a community which has a local wetland ordinance. The wetland mapping is performed on either a large-scale topographic map or on an aerial photograph of suitable scale. Surveying in of the staked or flagged wetland boundaries is usually necessary to assure accuracy of the wetland limits. Next, the site plan is prepared by a landscape architect or civil engineer who attempts to avoid encroaching on regulated wetlands. With, or without, the aid of a wetland consultant, a wetland permit form of the Michigan DNR is filled out and a wetlands permit application, complete with necessary drawings, is forwarded to the Michigan DNR for approval.

According to P.A. 203, the Michigan DNR has 90 days to approve or deny the submitted wetland permit application. This 90-day review period commences no the day the permit application is determined to be complete, or on the date of a public hearing, provided such a hearing has been requested. During the first 60 days of the review period, the local community may provide review comments to the Michigan DNR. Unless health, safety, or regional issues are involved, the Michigan DNR generally does not approve a wetlands permit for which the local community has recommended denial. Some developers, and other permit applicants, will attempt to schedule a meeting with the Michigan DNR representative in charge of the permit file just prior to the end of the mandatory review period.

Local Wetlands Ordinances

The original Statute, as well as the amendments to P.A. 203 in 1992, both provide for the enactment of local wetland ordinances by counties, municipalities, and townships. As indicated in Appendix A, 38 local governments in Michigan have established wetland ordinances. Of these, 46% are located in communities in Oakland County. Situated in populous southeast Michigan, Oakland County is the wealthiest county in the state. A previous paper by Jaworski & Manhart (1988), revealed that communities with wetland ordinances tend to be wealthy and currently undergoing considerable residential or commercial growth.

Although some of the local wetland ordinances are special, most are a part of zoning ordinances. Special ordinances are also referred to as regulatory ordinances. Typically, the wetland ordinances contain sections on definitions, activities allowed without a permit, activities requiring a permit, permit requirements, and permit review procedures. As part of the ordinance, many of these communities also maintain a semi-official wetlands map which shows the general location of wetlands within the jurisdiction. However, these local wetland ordinances vary from community to community, particularly in regards to wetland definitions, minimum size of regulated wetlands, width of wetland buffer, and permit review guidelines. Approximately half of these ordinances regulate wetlands as small as two acres and employ a fixed 25-foot wide wetland buffer.

DISSATISFACTION WITH WETLAND PERMIT PROCESS

Even though the Michigan DNR maintains that 90% of all wetland permits applications eventually obtain approval, over the past few years considerable anti-wetland permit sediment has accumulated. Led by the Michigan Homebuilders Association, and inclusive of private developers, realtors, as well as selected attorneys and engineering firms, this group of dissatisfied individuals began to promote the amendment of P.A. 203 of 1979. While admitting that P.A. 203 could benefit from amendment, members of the Michigan DNR and environmental groups were reluctant to recommend legislative redress lest the wetlands act be weakened. However, in 1992, the opposition group began to focus on eliminating and/or restricting local wetland ordinances.

Developers' Dirty Dozen

In 1991 and 1992, a number of bills were introduced into the Senate and House of Representatives of the State of Michigan. Most notable was Senate Bill 522 which was drafted in September, 1991. Senate Bill 522 had the effect of galvanizing the environmental community, and much misinformation began to appear in the media. However, there were 12 points that the Michigan Homebuilders Association was determined to insert in Senate Bill 522. These 12 points of discussion are outlined in Appendix B. The environmental community quickly began to refer to these 12 points as the "dirty dozen".

Although some of the 12 points of discussion contained in Senate Bill 522 were less realistic than others, the Michigan Homebuilders Association was attempting to make a statement about the wetland permit process in Michigan. Of particular concern to this group, was that wetland definitions varied among local wetland ordinances, and that wetland consultants often disagreed when performing wetland delineations. Moreover, there was a general feeling among many developers that there was no standardization in regard to wetland permit applications, and that permit review needed streamlining. The wetlands ordinance of West Bloomfield Township, Oakland County, was singled out as being the most anti-development because of its regulation of wetlands as small as 0.25 acres and its employment of a separate and autocratic wetlands review board. Also, the incorporation of a wetland buffer in the local ordinances was often regarded as an additional taking and another means of reducing the density of development.

Senator Honigman's Committee

Beginning in the Spring of 1992, Senator Honigman of Oakland County chaired a bipartisan committee of informed individuals who were seriously interested in amending P.A. 203. Both the pro-development and pro-environment groups were well represented at the monthly meetings, which usually included approximately 25 individuals. Even though Senate Bill 522 initially served as a basis of discussion, the Legislative Services Council began to draft two senate bills as these meetings progressed. Senator Honigman worked diligently to avoid open confrontation and sought consensus on the various points of debate. What emerged in October, 1992 were two amendments to P.A. 203, which were eventually signed by Governor Engler in December, 1992.

1992 AMENDMENTS TO P.A. 203

The two amendments to P.A. 203 of 1979, are P.A. 295 and P.A. 296 of 1992. These two amendments comprise the first change in Michigan's Wetland Protection Act since it took effect in October, 1980. Although the amendments were effective immediately, local communities have until June, 1994 to be in compliance with the amended wetlands act.

Principal Changes

The overall intent of the amendments is to allow for local regulation of wetlands, but only as provided in the amended state wetlands act. It is apparent that the Michigan Homebuilders Association was willing to accept the development constraints imposed by P.A. 203 with its minimum wetland regulatory size of 5 acres and no wetland buffer, but not amenable to local ordinances which could regulated smaller wetlands and include wetland setbacks.

Outlined below are the principal changes which are contained in the amendments to P.A. 203. Greater detail concerning the amendments is contained in Jaworski (1993) and Fisher (1993).

~ Local wetland ordinances shall not provide a different definition of wetlands than is provided in the State Statute, which is identical to the federal definition of wetlands, except that a wetland ordinance may regulate wetlands less than 5 acres in size.

~ Upon receiving an application for a wetland use permit involving a wetland that is less than 2 acres in size, the local government shall approve the permit unless it determines that the wetland is essential to the preservation of the natural resources of the community and provides these findings to the applicant.

~ Prior to enactment of a wetlands ordinance, the local community must complete and make available to the public, at a reasonable cost, an inventory of all the wetlands within the community.

~ Any county, municipality, or township that adopts an ordinance regulating wetlands shall use the same permit form currently being utilized by the Michigan DNR, and each person applying for a wetlands permit shall make application directly to the local government. The local government shall forward a copy of each permit application to the Michigan DNR.

~ Though the amendments were silent on the buffer issue, some pro-development attorneys stated that the intent of the amendments was to eliminate wetland buffers on the local level. Case precedent may be necessary to clarify this issue, however.

~ With regard to the permit review period, the local community shall review an application in accordance to its wetlands ordinance and shall approve or deny the application within 90 days after receipt. The denial of a permit shall be accompanied by a reason in writing for denial.

~ If a wetlands permit is denied at the local level for a proposed wetlands use, the landowner may request a reevaluation of the affected property for property tax assessment purposes.

~ Local ordinances shall not require a wetlands use permit for uses and activities that are authorized without a permit under the State Statute.

~ The wetland permit applicant shall not be required to submit to more than one public hearing at the local level.

~ The local decision-making body which approves site plans, plats, and other related matters, shall be the same entity which will process wetland use applications. (This stipulation is intended to eliminate the need for wetland review boards).

Impact on Local Wetland Ordinances

As a result of these two amendments to P.A. 203, all 38 of the wetland ordinances must be revised prior to June, 1994, or risk being out of compliance. Those communities which have previously adopted wetland ordinances, but did not prepare wetland inventories, must do so and disseminate the findings to the public prior to June, 1994. Because the amendments did not specify a scale or level of detail for these wetland inventory maps, communities could employ National Wetland Inventory Maps and other available data sources including soil survey maps.

To assist local communities with wetland ordinances in addressing the amendments, a voluntary advisory group entitled the "Southeast Michigan Wetland Coalition" has been established. This paper, as well as the works by Jaworski (1993) and Fisher (1993), are part of that community assistance program. Other informal efforts are underway, including the providing of consulting by private environmental firms. The Michigan DNR is also responding by increasing its educational function to include a wetlands identification workshop to be held in June, 1993.

There are three issues, however, which will be somewhat difficult for local governments with wetland ordinances to resolve. These issues include: 1.) regulation of wetlands smaller than 5 acres, particularly those under 2 acres in size; 2.) review of wetland permit applications within 90 days, especially when site plan and other approvals require much more time; and, 3.) the establishment of wetland buffers of setbacks. In order to regulate wetlands smaller than 2 acres, specific field evaluation procedures must be established and the landowners notified of the results of such investigations. The 90-day review period can be addressed by providing an option for applicants to waive the 90-day limit until associated approvals have been obtained. With regard to buffers, some effort is being expended on the notion of establishing setbacks from natural features, such as wetlands, and including such provisions in appropriate zoning ordinances.

PERSPECTIVES ON AMENDMENTS TO P.A. 203

Although it may be too soon to be conclusive, it appears that wetland ordinances on the local level in Michigan will not be abandoned. The Michigan DNR has announced open support for local jurisdictions for several reasons, including the value of community's "eyes and ears" regarding unpermitted wetland activities. Most of the 38 communities with ordinances recognize that the State of Michigan has very limited staff and financial resources. Moreover, if local communities wish to address related matters of stormwater management, flood control, and erosion control, it is necessary to provide for wetland protection through local ordinances.

It is also important to realize that the amendments were, in large part, a result of serious criticisms levelled at wetland protection in Michigan by pro-development forces. These changes in Michigan's Wetland Protection Act may eventually be regarded as healthy, mid-course adjustments to environmental preservation that is less well accepted than floodplain protection. As the wetland debate continues at the federal level (Alper, 1992), the Michigan Homebuilders Association, and other groups in Michigan will continue addressing the taking issue, insisting on tax relief for unbuildable areas, and taking public entities to court who capriciously deny wetland use permits. Thus, if these amendments help provide for a more standard definition of wetlands, streamline the permit application process, and encourage local communities to be more flexible in their review of environmental permits, then perhaps the anti-wetland sentiment in Michigan has been reduced somewhat.

ACKNOWLEDGEMENTS

The authors appreciate the support of the Southeast Michigan Wetlands Coalition in preparing this paper. Manuscript review was graciously provided by Steve Sadewasser, Michigan DNR; Steve Marshall, City of Southfield; and Mark Wyckoff, Planning & Zoning Center. The word processing and duplication skills of Lisa Merrell, J & L Consulting Services, was also of importance.

LITERATURE CITED

Alper, Joseph. 1992. "War over the Wetlands: Ecologists vs. the White House". Science Vol. 257 (August 21, 1992): pp. 1043-1044.

Cwikiel, Wilfred. 1992. "Michigan's Wetlands, Yours to Protect, A Citizen's Guide to Local Involvement in Wetlands Protection". 2nd Ed. Tip of the Mitt Watershed Council, Conway, MI. 84 pp.

Dean, Lillian F. and Eugene Jaworski. June, 1991. "Protecting Wetlands at the Local Level, Options for Southeast Michigan Communities". Rouge River Watershed Council, Detroit, MI. 32 pp.

Fisher, Gerald A. April 20, 1993. "Legal Issues and Ordinance Revisions". Unpublished Paper, Presented at Wetlands Workshop for Southeast Michigan Wetland Coalition, to Respond to Amendments to State Wetlands Act. Held in Novi, Michigan on April 20, 1993. Kohl, Secrest, Wardle, Lynch, Clark, and Hampton, Farmington, MI. 10 pp.

Jaworski, Eugene. February 18, 1993. "Recently Enacted Amendments to Michigan's Wetland Protection Act (P.A. 203 of 1979). Unpublished Seminar Report, J & L Consulting Services and Southeast Michigan Wetlands Coalition, Ypsilanti, MI. 5 pp.

Jaworski, Eugene and Nanette Manhart. 1988. "Role of Local Governments regarding Wetland Protection in Michigan". In: Proceedings of 15th Annual Conference on Wetlands Restoration and Creation. Held May 19-20, 1988, in Tampa, Florida. Hillsborough Community College, Plant City, FL. pp. 136-142.

Tilton, Donald L. Winter, 1988. "Living with Wetlands". Michigan Planner Vol. 8(1):3-5, 15.

Wyckoff, Mark A. November, 1988. "Wetlands Regulation, Part 1, The State and Federal Role". Wetland Regulation Series, Planning & Zoning News Vol. 7(1):8-15.

APPENDIX A

COMMUNITIES WITH WETLANDS ORDINANCES
(Per Michigan DNR, May 10, 1993)

Community	County
Addison Township	Oakland
Ann Arbor, City of	Washtenaw
Argentine Township	Genesee
Auburn Hills, City of	Oakland
Augusta Township	Washtenaw
Bloomfield Township	Oakland
Brandon Township	Oakland
Brighton Township	Livingston
Brownstown Township	Wayne
Burt Township	Cheboygan
Charleston Township	Kalamazoo
Clarkston, Village of	Oakland
Clyde Township	St. Clair
Fenton, City of	Genesee
Green Oak Township	Livingston
Grosse Ile Township	Wayne
Hamburg Township	Livingston
Hayes Township	Charleviox
Independence Township	Oakland
Lake Angelus, City of	Oakland
LaSalle Township	Monroe
Meridian Township	Ingham
Milford Township	Oakland
Mundy Township	Genesee
Novi, City of	Oakland
Oakland Township	Oakland
Orchard Lake, Village of	Oakland
Orion Township	Oakland
Oxford Township	Oakland
Pinckney, Village of	Livingston
Rochester Hills, City of	Oakland
Saugatuck Township	Allegan
Southfield, City of	Oakland
Waterford Township	Oakland
West Bloomfield Township	Oakland
White Lake Township	Oakland
Whitewater Township	Grand Traverse
Wixom, City of	Oakland

Not Yet Adopted

Brighton, City of	Livingston
Scio Township	Washtenaw

Some ordinances in this list require a DNR permit prior to the issuance of local development. Other ordinances require a separate local wetlands review and permit.

APPENDIX B

POINTS OF DISCUSSION ON SENATE BILL 522

1. All local ordinances would uniformly use the state definition of wetlands found in the Goemaere-Anderson Wetlands Preservation Act.

2. All local units must complete and make available to the general public, without cost, a wetlands inventory map prior to enacting a local ordinance. The local ordinance must allow for public hearings on the map and the map must be certified by the DNR. Those local units with wetlands ordinances already in place, would have one year to complete such a map or would lose their ability to enforce the ordinance. The state would have the option of either adopting the local wetlands map as part of its wetlands inventory, or doing its own mapping of the local unit. If the state did its own mapping of the local unit, that map would control.

3. Builders/developers/farmers would be given 100% density credit. If a 20 acre parcel contained 5 acres of wetlands and the density requirements was 1 dwelling per acre, the builder/developer would be allowed to build 20 dwellings on the 15 non-wetland acres.

4. Local units of government and individuals enforcing a local ordinance would be required to be certified as competent by the DNR and would have to use the DNR manual.

5. There would be one central board making zoning decisions.

6. Local wetlands permits would have to be approved or denied in 60 calendar days, and a written reason to give for any denial. Approval of the permit would be automatic on day 61.

7. Decisions on wetlands permitting, including approvals, denials and conditions, must be based solely on the standards of ordinance. No decision or conditions not related to the protection of the wetland, will be permitted. Any unit doing so, would automatically lose their authority to enforce their local ordinance.

8. All local ordinances would allow for mitigation within the county.

9. All local ordinances will contain language stating that "If a permit is denied for a proposed wetlands activity, the landowner may request a reevaluation of the affected property for assessment purposes, to determine its fair market value under the use restriction (Sec. 16 Goemaere-Anderson Wetland Protection Act).

10. All local ordinances shall contain language substantially similar to Sec. 17 (judicial review to protect wetlands owners rights) of the Goemaere-Anderson Wetland Protection Act.

11. All municipalities with a local ordinance would be required to formally notify owners of record of the possible change in the status of their property (Sec. 20 Goemaere-Anderson).

12. All municipalities with a local ordinance would be subject to the "takings" language of Sec. 21 of the Goemaere-Anderson Act.

A WATER-QUALITY SURVEY OF TWENTY-FOUR STORMWATER WET-DETENTION PONDS

Mark Kehoe, M.F.R.C.
Environmental Scientist
SOUTHWEST FLORIDA WATER MANAGEMENT DISTRICT
2379 Broad Street
Brooksville, Florida 34609

ABSTRACT

During the latter months of 1988 and continuing through December, 1989, the Southwest Florida Water Management District conducted a water-quality survey of twenty-four stormwater wet-detention ponds in the Tampa Bay region. The objectives of the survey were: 1.) to provide regional, base-line water-quality data in urban stormwater wet-detention ponds; 2.) to document whether the water quality met state water-quality standards, and 3.) to explore relationships among survey variables. The percentage of samples exceeding water-quality standards (total percent exceedence, %E) for samples taken when ponds were discharging was 10 percent or more for four variables (dissolved oxygen, total zinc, total copper and total cadmium), with the total lead standard exceeded in about 9 percent of samples. The results were similar if all samples were considered, but %E nearly doubled for cadmium (18%) and TSS (10%). Although inputs and outputs were not measured, the data (especially TSS and turbidity) indicated that wet-detention ponds were probably effective as sedimentation basins.

Sources of seasonal variability in stormwater constituents probably included antecedent conditions and first-flush effects, rainfall and water level changes, variable watershed and atmospheric sources, and in-pond processes. Multivariate analyses provided additional evidence that hydrologic conditions, pond dimensions and primary production (including aquatic macrophytes) had impacts on the water quality of the wet-detention ponds in this study. Land-use evaluation and cluster analysis suggested that multi-family residential ponds were among the ponds with the poorest water quality. Higher population density and more impervious surfaces are probably involved.

More than 80 percent of the data on four heavy metals (cadmium, lead, chromium, and nickel) had values that were below laboratory detection limits. Therefore, the analyses that were performed using one-half the detection limits for these data (i.e., censored data) should be viewed with caution.

INTRODUCTION

Over the last twenty-five years, a significant reduction in "point-source" discharges to the nation's surface waters has occurred. As a result, stormwater runoff is now recognized as the major "non-point source" of pollution to surface water bodies (USEPA, 1983; Livingston, 1986). Stormwater runoff is blamed in part for many negative ecological changes. For example, Lake Erie fisheries appeared to improve with the reduction of point source and non-point source pollution (Bastian, 1986; Marsalek, 1986). Stormwater may vary in water quality depending on such factors as rainfall amounts, land uses, or the specific treatment; i.e., "best management practice" (BMP), being employed (Wanielista, 1978; Terstriep, et al., 1986; and Whalen and Cullum, 1988). Heavy metals are one area of concern regarding the pollutants found in stormwater runoff. Source of heavy metals in stormwater runoff include petroleum products, antifreeze undercoating and galvanizing, brake linings, engine wear, rubber, asphalt, and concrete (Whalen and Cullum, 1988) and many others (e.g., algicides and herbicides with copper). Wetfall and dryfall atmospheric sources are suggested for most heavy metals and trace elements (Arimoto and Duce, 1987). Recent data indicate that rainfall in the Tampa Bay region is a significant source of some metals (Rushton and Dye, 1993).

Findings of the Florida Department of Environmental Regulation (FDER) determined that stormwater was responsible for most of the pollution loads entering Florida waters. It is estimated that stormwater pollution is responsible for: 1.) 80 to 95 percent of heavy metals loading into state waters; 2.) virtually all of the sediments deposited into state waters; 3.) 450 times the suspended solids and 9 times the BOD_5 substances of secondarily-treated sewage entering state waters; and, 4.) nutrient loads comparable to secondarily-treated sewage effluent entering state waters. Regulations now suggest an 80 percent reduction in annual pollutant discharge loads from new systems, and a 95 percent reduction if the discharge enters an Outstanding Florida Water (OFW). In Florida studies for a variety of land uses, the first half-inch of runoff (projected to annual loads) contains 80 to 95 percent of most pollutants. At the time of this survey, various BMP's or combinations of BMP's could meet the load reduction standards by treating the first one-half inch of runoff (Livingston, 1986).

Generally, stormwater treatment in Florida is accomplished through "retention" or "detention" (Livingston, 1986). By definition retention is diversion of a prescribed amount of stormwater runoff to a treatment area with no subsequent discharge to waters of the state. Thus, retention results in near total treatment of the diverted water. Detention is diversion of a prescribed amount of stormwater runoff to a treatment area for prescribed residence times. Detention includes controlled discharge of the treated volume of stormwater to receiving waters (Livingston, 1986). By backing up and holding a specific volume of runoff, wet-detention basins can provide good to excellent removal efficiencies for suspended solids, metals and nutrients, because the water column of wet-detention basins removes pollutants through sedimentation, degradation, vegetative uptake, and other physical and biological processes (Whalen and Cullum, 1988; Stahre and Urbonas, 1990).

During the latter months of 1988 and continuing through December, 1989, the Southwest Florida Water Management District (a/k/a "the District") conducted a water-quality survey of twenty-four stormwater wet-detention ponds in the Tampa Bay region. Three basic objectives for the survey were: 1.) to provide regional, base-line water-quality data; 2.) to document whether the water quality met state standards (1988 to 1990); and, 3.) to explore the relationships among variables.

STUDY SITES

The search for study sites resulted in selection of twenty-four wet-detention ponds with Chapter 40D-4 or 40D-40 permits (F.A.C., 1990) and located in the Tampa Bay region (Figure 1 and Table 1). Some ponds discharged into storm drains; therefore, not all receiving waters could be sampled. Data for six ponds (25%) were dropped from analyses (except for percent exceedence, %E), because they did not function correctly as wet-detention systems throughout the entire study period (Kehoe, 1992). Also, one of the ponds (Pond S) was the subject of an in-depth study of stormwater treatment efficiency (Rushton & Dye, 1993).

MATERIALS & METHODS

Characteristics of each pond and contributing basin were obtained from information in permit files of the District (Table 1). Three characteristics (pond area, maximum depth, and treatment volume) were determined to have relationships with certain water-quality variables. Surface-water grab samples and field measurements were taken during the two or three-day period immediately following storm events; however, not all storms during this study period were sampled. The result of this survey design was approximate monthly samples, during both wet and dry periods, allowing seasonal analysis of the data. All samples were collected between 8:00 a.m. and 5:30 p.m., and only the last date of the 2-3 day sample runs was reported. Samples were collected: 1.) within the pond, directly in front of the control structure (i.e., the outfall station); 2.) within the pond, located away from the point of discharge (i.e., the pond station); and, 3.) within the receiving water (if available), upstream of the discharge point (i.e., the receiving-water station). Most of the data analysis includes only outfall data, because only outfall samples had metal analyses performed. The data from receiving water stations were available for only a limited number of ponds (fifteen of twenty-four). Pond station and receiving water station data are summarized by Kehoe (1992).

Field measurements using a Hydrolab™ multiple-sensor water-quality instrument included measurements of temperature (°C),

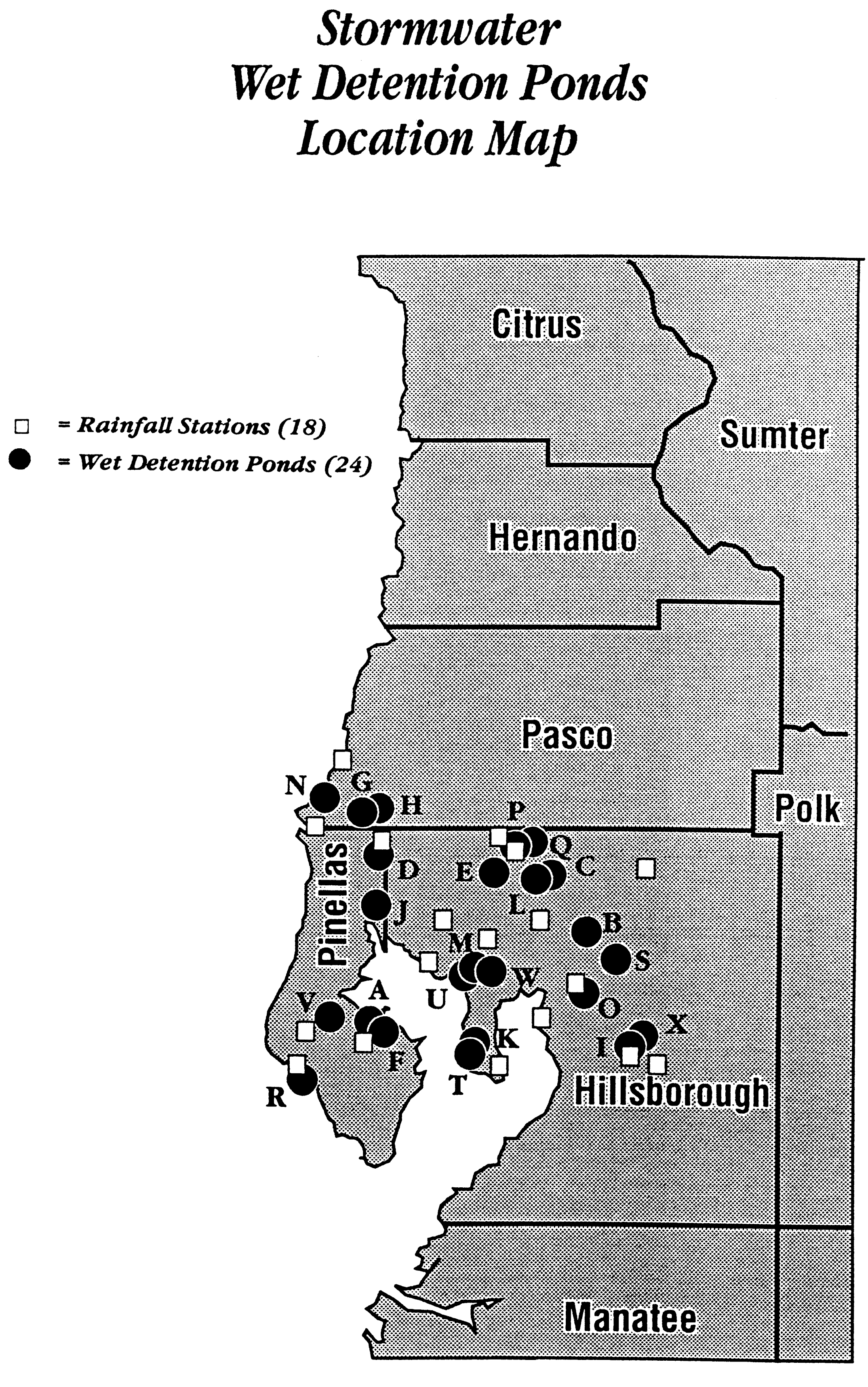


Figure 1. Locations of wet-detention ponds and rainfall stations.

Table 1. Permit Information for the twenty-four stormwater, wet-detention ponds. (Modified from Rushton, et al., 1989).

POND PSEUDONYM /LETTER		TYPE LC,HC SR,MR	PROJECT AREA (ACRES)	AREA DRAINED (ACRES)	IMPER VIOUS (%)	POND AREA (ACRES)	BASIN AREA (ACRES)	INLETS/ OUTLETS (NO.)	MAX. DEPTH (FT)	WATER TABLE (FT)	VOLUME (ACRE-FEET)	LITTORAL ZONE			
												SLOPE	%COVER EST	%COVER OBS	DEPTH (FT)
CARLL	A	LC	180.0	180.0	75	21.10	154.7	2/1	8.5	-2	15.45	8:1	35	8	.
GTEDS	B	LC	49.9	54.0	42	3.69	45.9	3/1	10	-6.8	5.03	10:1	35	45	3.0
COOPP	C	SR	19.5	39.0	40	3.50	39.0	2/1	14	-2	0.82	20:1	31	75	2.5
RIDGE	D	SR	78.3	78.3	55	2.95	34.3	4/1	4.5	-2.3	1.42	10:1	35	20	3.0
COPW5	E	SR	37.1	37.1	40	2.73	28.0	2/1	14	-2.5	1.36	4:1	36	45	3.0
CARLS	F	LC	180.0	180.0	75	2.30	25.3	1/1	8.5	-2	1.08	.	80	20	3.0
WPAIL	G	HC	63.6	64.2	66	2.20	23.8	1/1	7	-2	1.10	6:1	46	3	2.0
WPAIS	H	HC	63.6	64.2	66	1.84	7.4	1/1	7	-2	0.31	6:1	63	8	3.0
BLDAL	I	HC	19.0	19.0	44	1.64	17.6	4/1	12.0	-2.4	.	6:1	50	23	.
KENMA	J	HC	7.6	7.6	36	1.60	7.6	2/1	>3	-9.5	0.32	15:1	38	55	2.0
LIGHB	K	MR	11.8	11.8	57	1.44	11.9	2/1	9	-6	0.56	8:1	35	25	.
LASHV	L	MR	13.0	11.7	25	1.20	11.7	1/1	5.9	-10	0.55	.	100	95	1.5
MEBRV	M	MR	50.0	45.4	.	1.14	10.8	3/1	9.9	.	0.45	4:1	42	15	3.0
MARWA	N	MR	13.7	15.0	60	0.96	15.1	2/1	4	-2.5	0.63	.	40	15	3.0
EASTS	O	LC	7.5	12.0	7	0.80	12.0	2/1	2.5	.	0.55	4:1	35	100	1.0
LAKFN	P	SR	13.0	13.0	15	0.78	7.0	1/1	7.5	-4	0.30	4:1	0	28	.
LAKFS	Q	SR	13.0	13.0	15	0.76	6.0	1/1	7.5	-4	0.30	4:1	0	20	.
THORN	R	MR	3.6	3.6	41	0.60	3.6	2/1	3	-3.5	0.16	4:1	45	3	3.0
TAOFF	S	LC	9.5	6.3	30	0.35	6.3	1/1	3	-0.5	0.32	8:1	100	83	3.0
LIGHF	T	MR	5.2	4.8	76	0.32	4.9	2/1	2	-3.3	0.20	8:1	33	40	2.0
GEORG	U	LC	11.8	11.8	57	0.31	3.1	2/1	9	-6	0.19	8:1	35	0	.
TRICO	V	HC	1.8	1.8	71	0.15	1.8	3/1	2.23	-1.2	0.08	.	100	95	0.4
LIWAF	W	HC	4.8	3.4	47	0.12	1.9	1/1	2.3	-1.2	0.08	6:1	35	20	2.3
BLDAS	X	HC	19.0	19.0	44	0.04	1.4	2/1	5.2	-2.4	.	5:1	50	85	3.0

LEGEND: "LC" = LIGHT COMMERCIAL; "HC" = HEAVY COMMERCIAL; "." = MISSING DATA
"SR" = SINGLE FAMILY RESIDENTIAL; "MR" = MULTIFAMILY RESIDENTIAL
"EST" = ESTIMATE FROM PERMIT; "OBS" = OBSERVED FROM PHOTOS

dissolved oxygen (mg/L), pH (standard units), and specific conductance (umhos/cm or mmhos/cm), all taken on the bottom at the time that surface grab samples were collected. The time of day, discharge status of the control structure, and the depth to the bottom at the sample location were also recorded. The depth measurement did not coincide with the depth of the grab sample unless the water was very shallow (less than about 0.5m). Surface water grab samples were collected in 250 milliliter (mL) and 1.0 liter (L) opague polypropylene bottles. Depths of grab samples ranged from as little as 0.10 meter (4 to 5 inches to a maximum of about 0.50 meter (1.5 feet), when possible. Depths and sample locations were not consistent because of the fluctuating water levels and differing side slopes in ponds. Laboratory analyses were performed according to either Standard Methods (A.P.H.A., 1989) or EPA 600/4-79-020, Methods for Chemical Analysis of Water and Wastes (USEPA, 1979).

SAS™ Release 6.03 (1988) procedures and Lotus 1-2-3™ Release 2.01 (1986) were used to conduct statistical and graphical analyses of data. <u>Data transformations</u> (e.g., log 10) were used to improve normality for statistical analysis. Initial calculations and analyses (e.g., percent exceedence, medians, and means) were performed on the data from all 24 wet-detention ponds. Six ponds were subsequently deleted from all but the initial statistical analyses, due to improper operations or structural problems (Pond G, Pond I, Pond L, Pond N, Pond O, and Pond W) (Kehoe, 1992). Pond deletion resulted in the comparison of eighteen rather than twenty-four annual means for the variables, and had little effect on the graphical analysis of seasonal effects on date means.

Analytical and interpretive difficulties known as "left-censoring" occur when data values fall below a laboratory-specific detection limit for a variable. Means and other statistics were calculated for all eight metals by substitution of one-half the detection limit (DL) for left-censored values as suggested by Gilliom and Helsel (1986). This substitution method performs very well as a "first-look" at the estimation of means and standard deviations of censored data. Due to the uncertainty about censored data, only four metals were included in certain multivariate analyses. Their data were less than 50 percent left-censored (except Cu, 58 percent left-censored). The metals and their percent left-censoring are Zn (22%), Fe (<1%), Mn (50%), and Cu (58%). The metals deleted from multivariate analyses and the percent of left-censoring were Cd (82%), Cr (82%), Pb (83%), and Ni (88%).

RESULTS AND DISCUSSION

Discharge Frequency

Discharge frequency was calculated in two ways: 1.) annually (for ponds), equal to the number of sample dates a pond was observed discharging during the study period (Table 3); and, 2.) seasonally (for dates), equal to the number of ponds observed discharging on a sample date (not shown in Table 3) (Kehoe, 1992). In either case, discharge frequency is based on the percent (or number) of times that water was observed discharging through a control structure. While not all storm events could be sampled, the total discharge frequency (59% for all ponds on all dates) may be a good estimate for wet-detention ponds in the region because both wet and dry periods, a tropical storm, and more than 100 days of drought occurred during the survey. Note that seventeen outfalls were discharging on at least 50 percent of sampling dates. Annual, pond discharge frequencies ranged from zero (Pond R) to 100 percent (Pond U - with a known intermittent bleeddown due to a level-activated pump). Date discharge frequency demonstrates a distinct seasonal pattern than can be graphically related to rainfall, and that is directly correlated ($r=0.73$, $p=0.01$) to mean bottom depth data for the 11 sample dates, as well as the data for several other variables (Kehoe, 1992).

Exceedence of Standards (E%)

Evaluation of exceedence of state water-quality standards was based on whether samples collected "within" ponds exceeded specific standards; however, specific state water-quality standards do not apply directly to stormwater ponds, even though the standards apply to discharge from stormwater ponds. The interpretation presented here should not be used to conclude that exceedence of standards within the pond is equivalent to "discharging" water that exceeds standards, even though a relationship may exist. Also, the process of water flowing through a control structure could affect some parameters that may be influenced by water movement (e.g., dissolved oxygen and pH), that may be physically removed by skimmers (e.g., oils, greases, and trash), that may settle (e.g., suspended solids and turbidity), and that are absorbed or bonded to particles that settle such as nutrients, heavy metals, and pesticides (Stahre, and Urbonas, 1990). A distinction was made between samples collected when ponds were not discharging (about 41% of samples) (Figure 2).

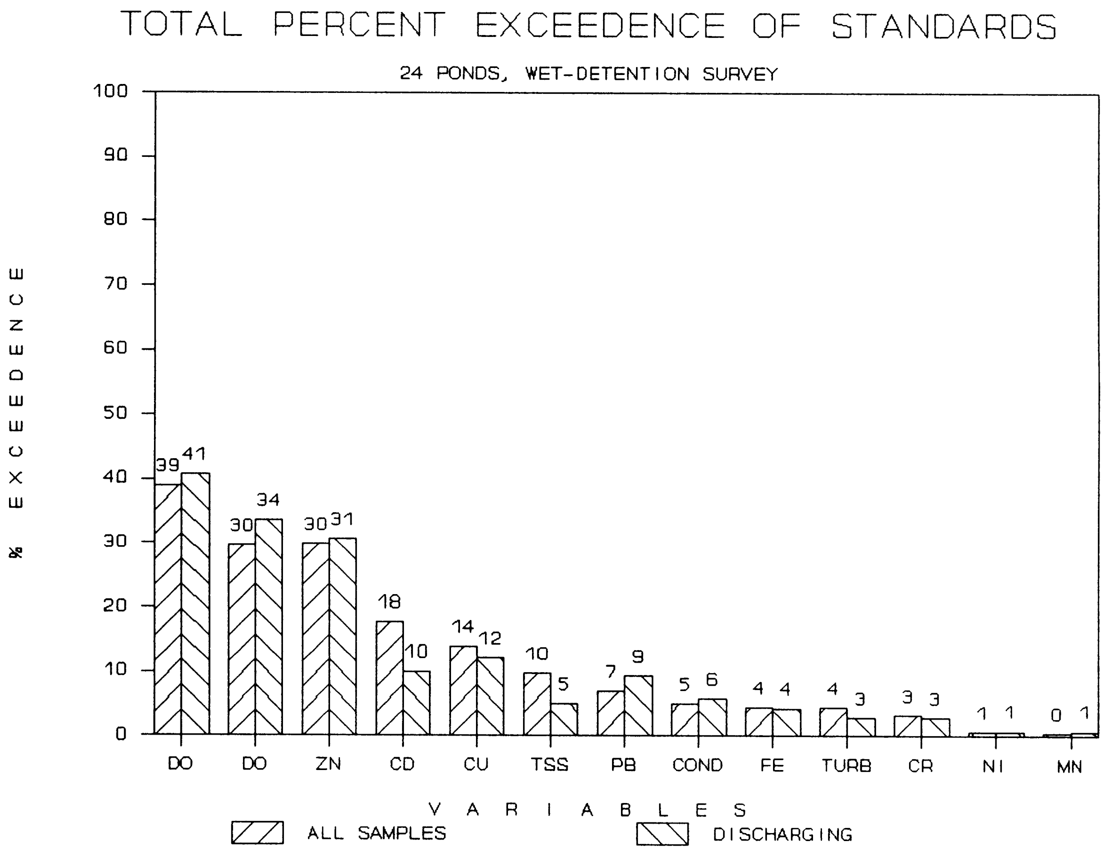


Figure 2. Total percent exceedence of water-quality standards, for all samples, and for samples only when outfalls were observed discharging.

Table 2 summarizes 1989-1990 water-quality standards for criteria that applied to state surface waters, to drinking water supplies, and to sewage treatment plant (STP) discharge. Of particular interest are the Chapter 17-302, Class III standards because the stormwater management systems that were studied discharged into Class III waters. Class III waters are designated to provide for "propagation and maintenance of healthy, well-balanced populations of fish and wildlife" (Chapter 17-302, F.A.C., 1990). In the absence of a Class III standard for manganese (Mn) and for total suspended solids (TSS), the Class II (shellfish) standard (for Mn) or an efficient sewage treatment plant's (STP) discharge concentration (for TSS) are compared. Also, Chapter 17-550 primary and secondary drinking water standards are included in Table 2 as a reference for some parameters.

Table 3 provides a summary of the median annual outfall station concentrations and the percent exceedence (%E) of the applicable water quality standards for each parameter at each outfall station. Figure 2 shows Total %E by parameter for all samples and also for samples only when ponds were discharging. Total %E represents the percentage of times, for all sampling dates and for all twenty-four ponds, that a standard was exceeded for a specific variable. The median value was initially utilized as the better measure of central tendency for censored and possibly non-normally distributed data. When compared with the mean, median of variables gives an indication of whether the distribution of data is normal or skewed (McClave and Dietriech, 1982). In Table 3, outfall stations are arranged by decreasing pond size (ponds are labeled alphabetically from largest to smallest, letter A through X), while columns are arbitrarily arranged by decreasing total percent exceedence (Total %E). The number of outfall stations that exceeded a standard one or more times (Number. of Ponds) is also listed in Table 3. Generally, the higher the Total %E (i.e., those standards more frequently exceeded) the greater the number of outfall stations (Number of Ponds) that exceeded standards for a variable (Table 3). The relationship is strongest for metals, and when total %E is greater than or equal to 10 percent. Finally, the number or percent exceedence of standards appeared to fluctuate with the seasonal (sample date) and annual (pond) means for variables, although an analysis was not carried out (Kehoe, 1992).

Dissolved Oxygen (DO)

While the 24-hour DO standards require diurnal (24-hour) sampling, valuable information was obtained by comparing single field measurements to the 24-hour mean (5.0 mg/L) and the 24-hour-lowest allowable (4.0 mg/L) DO standards. Thirty-nine percent (88 or 226 samples) of all outfall station DO measurements from 18 of 24 outfalls were below 5.0 mg/L (first column Figure 2), and 30 percent (67 of 226 samples) were below 4.0 mg/L (second column in Figure 2). The percent exceedence increases slightly if only samples when outfalls were discharging are considered. Most exceedences of the 24-hour DO standards occurred in nine ponds (%E = 60% or more, below 5.0 mg/L) with heavily vegetated outfalls that probably reduced circulation near the outfall and enhanced build-up of organic material. Also, DO measurements were generally recorded near the bottom, often beneath thick vegetative cover, and probably near or below light extinction depth in some ponds. The annual median DO concentrations were below 5.0 mg/L for all nine ponds. Log mean annual DO for outfalls was inversely correlated ($r=-0.66$, $p=0.003$) with estimated percent of pond area covered with aquatic macrophytes. The seasonal pattern of mean DO (i.e., date means) was probably inversely related to seasonal temperature changes (r=-0.53, p=0.09). There also appeared to be more exceedences of DO standards during the summer season and during wetter periods (Kehoe, 1992).

	Criteria						
	Chapter 17-302 Class			Chapter 17-550		STP	
Parameter	I	II	III	1'	2'	2'	Lab DL
Zinc (Zn)	0.03	1.0	0.03 "F" 0.1 "M"	----	5.0	----	0.005
Cadmium (Cd)	0.0008 "FS" 0.0012 "FH"	0.005	0.0008 "FS" 0.0012 "FH" 0.005 "M"	0.01	----	----	0.002
Copper (Cu)	0.03	0.015	0.03 "F" 0.015 "M"	----	1.0	----	0.01
Lead (Pb)	0.03	0.05	0.03 "F" 0.05 "M"	0.05	----	----	0.01
Iron (Fe)	0.3	0.3	1.0 "F" 0.3 "M"	----	0.3	----	0.02
Nickel (Ni)	0.1	0.1	0.1	----	----	----	0.01
Manganese (Mn)	----	0.1	----	----	0.05	----	0.01
Chromium (Cr)	0.05	0.05	0.05	0.05	----	----	0.01
Dissolved Oxygen (DO)	5.0	5.0/24 HR Mean 4.0/24 HR Lowest	5.0 "F" 5.0/24 HR Mean "M" 4.0/24 HR Lowest "M"	----	----	----	----
pH (Standard Units)	6.0 to 8.5 NB +/- 1.0	6.5 to 8.5 NB +/- 1.0	6.0 "F", 6.5 "M" to 8.5 NB +/- 1.0	----	[6.5 to 8.5]	----	----
Total Suspended Solids (TSS)	----	----	----	----	----	20.0	0.05
Turbidity (NTU)	29.0	29.0	29.0	1.0	----	----	----
Conductivity (umhos/cm)	1275	1275	1275 "F"	----	----	----	----

Legend:
DL = Detection Limit
"F" = Freshwater
"FS" = Freshwater (Soft, under 150 mg/L CaCO3)
"FH" = Freshwater (Hard, above 150 mg/L CaCO3)
"M" = Marine
NB = Natural Background
STP = Sewage Treatment Plant
---- = Not Established or Not Given
1' = Primary
2' = Secondary

Table 2. Water-quality standards for several criteria applicable to state waters (17-302), drinking water (17-550) or sewage treatment (STP). SWFWMD laboratory detection limits (DL) for metals also given. All values are mg/L unless indicated. (F.A.C. Chapters 17-302 and 17-550).

Table 3. Median annual outfall concentrations and percent exceedence (%E) of water-quality standards (by pond) for 24 wet-detention ponds, 1988-1989. Lab detection limits also given. All values are mg/L unless indicated. (Source: Kehoe, 1992).

Pond Pseudonym\	Landuse\	LTR	DO	%E	Zn	%E	Cd	%E	Cu	%E	TSS	%E	Pb	%E	CONDUCTIVITY mmhos/ cm (a)	%E	Fe	%E	TURBIDITY NTU (b)	%E	pH SU (c)	%E	Cr	%E	Ni	%E	Mn	%E	DF (%) (d)
CARLL	LC	A	7.30	18	0.021	27	<	30	<	9	1.78	0	<	9	0.700	0	0.076	0	1.8	0	7.50	0	<	9	<	0	<	0	91
GTEDS	LC	B	5.72	18	0.027	36	<	0	0.010	18	6.45	9	<	0	0.323	0	0.230	9	3.3	0	7.30	9	<	0	<	0	<	0	90
COOPP	SR	C	6.60	40	0.018	27	<	10	<	0	3.25	0	<	0	0.200	0	0.170	0	2.2	0	6.60	0	<	0	<	0	0.010	10	70
RIDGE	SR	D	2.70	89	0.015	11	<	13	0.019	22	4.08	0	<	11	0.246	0	0.260	0	6.9	0	6.70	11	<	0	<	0	<	0	73
COPW5	SR	E	2.40	100	0.003	45	<	30	<	9	5.10	11	<	18	0.253	0	0.140	0	2.0	0	6.55	0	<	0	<	0	0.010	0	36
CARLS	LC	F	8.36	0	0.011	0	<	20	0.019	45	2.50	9	<	9	0.383	0	0.093	0	2.0	0	7.90	0	<	0	<	0	0.010	0	73
WPAIL	HC	G	7.00	0	0.010	11	<	0	<	0	7.01	25	<	0	0.086	0	0.940	44	57.5	100	7.24	25	0.01	0	<	0	<	0	18
WPAIS	HC	H	6.50	18	0.014	18	<	30	<	18	8.54	36	<	9	0.231	0	0.229	0	11.3	9	7.60	9	<	9	<	0	<	0	36
BLDAL	HC	I	6.53	18	0.026	45	0.003	56	0.010	20	3.62	0	<	0	0.159	0	0.195	0	7.3	0	6.66	0	<	0	<	0	<	0	.
KENMA	HC	J	2.90	64	0.012	0	<	13	<	0	3.66	0	<	11	0.178	0	0.116	0	1.7	0	6.52	0	<	0	<	0	<	0	30
LIGHB	MR	K	6.51	27	0.020	18	<	10	0.011	10	7.80	0	<	0	0.319	0	0.330	9	5.1	0	7.47	0	<	0	<	0	0.027	0	50
LASHV	MR	L	4.80	60	0.021	27	<	30	<	9	8.41	0	<	9	0.117	0	0.171	0	4.4	0	6.99	0	<	0	<	0	<	0	55
MEBRV	MR	M	6.70	0	0.021	40	<	22	<	10	3.20	0	<	0	0.126	0	0.127	0	2.0	0	7.50	14	<	0	<	0	0.012	0	55
MARWA	MR	N	3.60	64	0.037	64	<	20	<	9	12.62	18	<	9	11.540	100	0.276	9	6.5	0	8.00	0	<	18	<	0	0.029	0	80
EASTS	LC	O	3.30	80	0.074	100	<	29	<	20	11.40	40	<	0	0.137	0	0.235	20	4.3	20	6.96	0	<	0	<	0	0.012	0	50
LAKFN	SR	P	6.96	30	0.012	0	<	10	<	18	3.38	0	<	0	0.152	0	0.200	9	2.1	0	7.26	0	<	9	<	0	<	0	78
LAKFS	SR	Q	7.75	0	0.005	18	<	10	<	9	8.44	0	<	18	0.176	0	0.302	0	5.7	0	7.65	10	<	0	<	0	<	0	67
THORN	MR	R	8.30	0	0.025	45	<	30	0.019	27	23.50	73	<	0	0.347	0	0.392	9	21.0	27	8.10	0	<	0	<	0	<	0	0
TAOFF	LC	S	2.80	78	0.028	44	<	13	<	0	6.06	0	<	11	0.392	0	0.287	0	5.3	11	7.04	0	<	0	<	0	0.015	0	73
LIGHF	MR	T	8.00	0	0.041	64	<	20	0.026	45	11.56	22	<	9	0.255	0	0.232	0	7.7	0	7.90	11	<	18	<	13	0.012	0	55
GEORG	LC	U	5.10	45	0.021	27	<	20	<	18	3.46	0	<	9	0.259	0	0.210	0	3.9	0	6.98	0	<	0	<	0	0.012	0	100
TRICO	HC	V	3.28	73	0.019	9	<	0	<	0	5.00	0	<	9	0.143	0	0.220	0	3.1	0	6.68	0	<	0	<	0	0.011	0	73
LIWAF	HC	W	7.17	29	0.019	40	<	0	<	0	5.19	0	<	10	0.133	0	0.206	10	8.4	0	7.50	0	<	10	<	0	<	0	40
BLDAS	HC	X	3.14	91	0.016	27	<	10	<	9	3.87	0	<	9	0.225	0	0.260	0	3.8	0	7.00	0	<	0	<	0	0.019	0	89
Total %E				38.9		29.8		17.7		13.8		9.8		6.9		4.9		4.5		4.4		3.1		3.3		0.6		0.4	// 59
# Ponds				18		21		20		18		9		15		1		8		5		7		5		1		1	
Standard			5.0		0.03		0.0012		0.03		(f) 20.0		0.03		1.275		1.0		29.0		6.0 - 8.5		0.05		0.1		(g) 0.1		
DL (e)			----		0.005		0.002		0.01		----		0.01		----		0.02		----		----		0.01		0.01		0.01		

Legend: (a) Millimhos per centimeter @ 25 degrees Centigrade.
(b) Nephelometric turbidity unit(s).
(c) Standard unit(s).
(d) Discharge frequency (percent).
"." = Missing data.
SR = Single-family Residential
LC = Light Commercial
(e) DL = SWFWMD Lab Detection Limit.
(f) 20 mg/L TSS represents efficient 2' sewage treatment.
(g) Chapter 17-3 Class II standard for Mn.
"----" = Not established or not given.
"<" = Less than the Detection Limit (DL)
MR = Multi-family Residential
HC = Heavy Commercial

Ponds should be designed with aquatic macrophytes located so that a moderately deep (e.g, 1.5 m) sedimentation area is separated from a shallower, wider, open-water zone. In this manner, circulation and water movements will promote higher DO concentration through reduction of the effects of sedimentation. Pond maintenance to exclude undesirable plant abundance and species, to remove built-up organic matter and sediments, and to maintain the open-water zone directly in front of the control structure is desirable (SWFWMD, 1988 and Stahre and Urbonas, 1990). Because of the interaction of many physical, chemical, biological, temporal, and spatial factors affecting DO in surface waters, the specific pond size, shape, depth, and vegetation zones (i.e., stormwater system design) that will provide optimal DO concentrations should be determined on a site-by-site basis.

Zinc (Zn)

Zinc was probably the most problematic heavy metals measured during the study. None of the annual medians for zinc, and only 22 percent of the samples (55 of 248 samples) were less than the zinc detection limit (0.005 mg/L). A large number of outfall stations (21 of 24) (Table 3) and 30 percent (74 of 248 samples) of the grab samples exceeded the Class III zinc standard (0.03 mg/L); 34 percent exceeded the standard, if only samples when ponds were discharging are considered (Figure 2). Most of the outfall stations that exceeded the zinc standard did so more than once, except Pond D, Pond G, and Pond V. The annual mean zinc of 14 outfalls exceeded 0.03 mg/L (Kehoe, 1992), but there were no significant statistical correlations of annual mean zinc with other measured parameters for the outfalls. On five of eleven sample dates, 40% or more samples exceeded the zinc standards; on one date 75% of samples (18 of 24) exceeded the zinc standard. Seasonally, log mean zinc for the 11 sample dates was directly correlated with mean DO ($r=0.64$, $p=0.02$), but the reason is not clear. Higher means and more exceedences of the zinc standard on specific dates suggest an atmospheric source of zinc or a seasonal effect that may be related to changes in water levels, rainfall amounts and antecedent periods, wind-induced mixing, or internal cycling. At Pond S, the instantaneous zinc concentrations of 21 storms in 1990 ranged from 0.016 to 0.201 mg/L (mean = 0.045); 12 of the 22 storms exceeded 0.03 mg/L (Rushton, 1993). The data suggest that zinc should remain a source of concern in stormwater projects at the District.

Cadmium (Cd)

The bioavailability and toxicity of metals are affected by temperature, pH, organic and inorganic ligands, and hardness. Hardness, alkalinity, and pH may often be related through the carbonate-bicarbonate buffer system; hardness and alkalinity are reported in milligrams per liter as $CaCO_3$. In wet-detention ponds, hardness may depend on the regional surface geology and physiography (Canfield and Hoyer, 1988) and in-pond processes that may affect the concentrations of calcium and magnesium carbonates (Wetzel, 1975). Generally, higher background total hardness levels result in higher pH and alkalinity levels, especially if calcium and magnesium carbonates are the main source of hardness. Hardness indirectly affects metals toxicity by interfering with metals uptake by competing for uptake sites on cells of the gill membrane in fish (Davies, 1986). As hardness increases, metals toxicity decreases. For the 1989 Chapter 17-302, Class III cadmium standard, the hardness cutoff point is 150 mg/L (as $CaCO_3$). Above 150 mg/L, the cadmium standard is 0.0012 mg/L; however, below 150 mg/L, the cadmium standard is 0.0008 mg/L (Table 2). Hardness and alkalinity were not measured during the survey, and the higher standard for cadmium (Chapter 17-302, Class III = 0.0012 mg/L) was used. However, recent data from an ongoing stormwater survey indicates that hardness may be quite variable (0 to 200 mg/L as $CaCO_3$) within and between stormwater ponds in the Tampa Bay region (M. Kehoe, unpublished). Recently revised State of Florida water-quality standards base heavy metals standards on a calculation using the natural log of total hardness in a surface waters (Florida Administrative Code, Chapter 17-302, 1992).

Note that median cadmium values (23 of 24 outfalls were below the method detection limit (BDL or <) of the laboratory (0.002 mg/L) for every outfall except Pond L (Table 3). Eighty-two percent of grab samples (186 to 226 samples) were BDL for cadmium. Whether or not the cadmium standard was exceeded in samples when cadmium levels were BDL is unknown, because the standards (0.0008 and 0.0012 mg/L) are also less than 0.002 mg/L. Thus, cadmium was in exceedence of its standard whenever values occurred above its DL, and probably more often than reported here. The analysis would be equally vague (most data below DL) if wet-detention ponds were assumed to be softwater in character or of mixed hardness.

Eighteen percent of the grab samples (40 of 226 samples) originating from 20 or 24 outfall stations (Table 3 and Figure 2) exceeded the cadmium standard (0.0012 mg/L). Twelve outfall stations exceeded the cadmium standard two or more times. The total percent exceedence decreases by nearly half to 10 percent if only samples when outfalls discharged are considered (Figure 2). This may indicate that cadmium and water levels may be inversely related, a relationship that may involve bottom sediments. In fact, log mean annual cadmium for the 18 outfalls was inversely related to log mean annual bottom depth ($r=-0.52$, $p=0.03$), and to mean annual iron ($r=-0.57$, $p=0.01$). Rushton (1993) detected cadmium (thus exceeding the standard) in rainfall for only 7 of 22 storms in 1990. Most cadmium exceedences occurred on two sampling dates, one during a dry period (6/7/89) and one during a more rainy period (12/19/89), each with 13 of 24 outfalls exceeding the cadmium standard. Seasonally, log mean cadmium for the 11 sample dates was inversely related to water level -- discharge frequency ($r=-0.69$, $p=0.03$) and possibly mean bottom depth ($r=-0.55$, $p=0.10$). Because exceedence of the cadmium standard occurs for the majority of ponds, and Total %E is 18 percent for all samples (and is probably higher), cadmium should remain a source of concern and investigation in District stormwater projects.

Copper (Cu)

Only seven outfall stations had annual medians above the DL for copper, and 58 percent (142 of 246 samples) of grab samples were less than the copper detection limit (0.01 mg/L). All of the outfall stations had less than 50 %E of the copper standard (two had 45 %E); therefore, none had annual medians above the copper standards (0.03 mg/L total copper) (Table 3). Fourteen percent (34 of 246 samples) of grab samples originating from 18 of 24 outfall stations (Figure 2) exceeded 0.03 mg/L. Two outfalls exceeded the standard 5 times each (Ponds F and T), and one outfall had three exceedences (Pond V). The total percent exceedence of copper decreases slightly (to 12 percent) if only samples collected when outfalls discharged are considered (Figure 2).

Three of the six outfalls that never exceeded the standard were heavily vegetated (Ponds C, J, and V). While vegetation appeared to be impacted occasionally at Ponds F, T, and others, whether or not herbicides containing copper were used was unknown. However, an inverse correlation (r=-0.47, p=0.05) existed between the log annual mean copper concentrations and estimated percent of pond surface area occupied with aquatic macrophytes (percent cover). Log annual mean copper concentrations were also directly correlated with mean annual pH (r=0.58, p=0.01), mean annual DO (r=0.52, p=0.03), and log mean annual nickel (r=0.66, p=0.03) for outfalls (Kehoe, 1992). Reduced macrophyte coverage resulting from herbicides containing copper may result in higher algal production with higher mean DO and mean pH. The role of copper uptake by macrophytes in reducing copper concentrations and the use of herbicides during the survey are uncertain.

Exceedences of the copper standard occurred mostly on three dates, one in a dry month and two during rainy months. The mean copper from the one dry period sample date was about 33% and 47% higher than the means from two wet-period dates. Seasonally, log mean copper for 11 sample dates was inversely correlated with pH (r=-0.58, p=0.06) (opposite of the annual mean correlation for these variables), but directly related to log mean cadmium (r=0.66, p=0.03) and log mean lead (r=0.62, p=0.04). The general seasonal pattern in mean copper and number of exceedences of the copper standard may indicate an inverse water-level (seasonal) effect or increased herbicide treatments - especially on one dry period date. Recent rainfall data suggest that rainfall is a minor source of copper (only detected in 3 of 22 storms) (Rushton, 1993). Because most of the ponds and over 10 percent of samples exceeded the Chapter 17-302, Class III copper standard, copper should also remain a source of concern and investigation in District stormwater projects.

Total Suspended Solids (TSS)

There was no established Chapter 17-302 standard for TSS during the survey; therefore, TSS concentrations reported for efficient secondary sewage treatment plants (2' STP) (Randall, et al., 1982) were used for comparison (20 mg/L, Table 2). Only one outfall station had a %E of the 2' STP standard greater than 50 percent (73 percent, Pond R). Pond R was the only outfall station with an annual median greater than 20.0 mg/L (Figure 7). Roughly 10 percent (22 of 225 samples) of grab samples originating from just 9 of 24 outfall stations (Figure 2) exceeded 20.0 mg/L. Pond R, with 8 of 11 samples (73%) greater than 20.0 mg/L TSS standard, accounted for 36% (8 of 22) of all TSS exceedences. One other outfall (Pond H) exceeded 20.0 mg/L on 4 of 11 dates. The %E for TSS decreases by half to 5 percent if samples only when ponds discharged are considered. This may result from less bottom sediment in samples when high water levels exist and/or greater sedimentation and flushing rates during the wet season. Log mean annual TSS concentrations were inversely correlated with discharge frequency (r=-0.62, p=0.006) for the 18 outfall stations. Positive correlations with log mean annual TSS included log mean annual turbidity (r=0.82, p=0.0001), mean annul iron (r=0.60, p=0.009), and perhaps mean annual pH (r=0.41, p=0.09). Seasonally, log mean TSS for the 11 sample dates was directly correlated with log mean conductivity (r=0.65, p=0.03) (Kehoe, 1992). TSS concentrations in rainfall were minimal compared to infall and reduced outfall concentrations during 29 storms at Pond S in 1990 (Rushton, 1993).

Studies and reports suggest that the reduction of suspended solids improves with increasing detention times (Whipple, 1979; Whipple and Hunter, 1981). Pretreatment basins or deeper sedimentation zones are also recommended (SWFWMD, 1990). If properly maintained (e.g., located away from the outfall), vegetational zones can further reduce pollutant loads, including TSS (Fetter, et al., 1978; Shih, 1981). Data from the District's 24-pond survey, from the District's intensive studies, and from studies elsewhere indicate the general effectiveness of wet-detention systems as sedimentation basins. However, while TSS generally did not appear to be a problem in the twenty-four ponds during the survey, the relationships between TSS and other variables (e.g., heavy metals and nutrients) reported in the literature warrants continued measurement of TSS in District stormwater projects.

Lead (Pb)

All annual pond medians for lead were less than the detection limit for lead (0.01 mg/L, Table 3); therefore, none exceeded the lead standard (0.03 mg/L). Roughly 7 percent (17 of 246 samples) of the grab samples originating from 15 of 24 outfall stations (Figure 2) exceeded 0.03 mg/L. Thirteen outfalls exceeded the lead standards only once, and two outfalls had two exceedences (Pond E and Pond Q). The highest outfall station annual "mean" was 0.034 mg/L at Pond S, at least 10 times less than EMC values reported for lead in the NURP report (Whalen and Cullum, 1988). If only samples when outfalls discharged are considered, there is a slight increase in the %E of lead from 7 to 9 percent (Figure 2).

The general seasonal pattern for lead did not show an inverse water-level effect; however, log mean annual lead concentrations for the 18 outfall stations were inversely correlated (r=-0.42, p=0.08) with log mean annual bottom depth. Seasonally, log mean lead was directly correlated (r=0.62, p=0.04) with log mean copper for the 11 sample dates. Most lead exceedences occurred on three dates, two during wet periods and one during a dry period. Nearly half of all lead exceedences (7 of 17) occurred on 12/19/89. While rainfall did not appear to be a significant source of lead during the District's intensive study of a wetlands treatment system (B. Rushton, unpublished), an atmospheric source of lead in dryfall was not evaluated and may occur. Winter peaks (December, 1988 and 1989) in log mean lead that are nearly an order of magnitude higher than other sample dates (Figure 29) may be due to greater automotive activity and subsequent wet and dry fallout containing lead during the winter tourism period. Other possible causes for higher lead in winter months are turnover and mixing of the water column causing resuspension of sediment-bound lead, and winter die-off of aquatic macrophyte and subsequent release metals. Although most of the data (83% - 204 of 246 samples) are below the lead detection limit (0.01 mg/L) and less than 10 percent of samples exceed the Chapter 17-302, Class III standard for lead, over half the ponds (15) exceeded the standard at least once. Because of its bioaccumulative and extreme toxicity, lead should continue to be a subject of investigation in District stormwater projects.

Specific Conductance

Specific conductance levels in this survey were well below the Chapter 17-302 standard for freshwater (1.275 mmhos/cm, Table 2). All outfall stations were in compliance with the standards except one (Pond N, Table 3) which received tidal backflow on a regular basis and cannot be considered freshwater; therefore, Pond N was dropped from subsequent statistical analyses (Kehoe, 1992). The twenty-three freshwater outfall stations were one order of magnitude below the Chapter 17-302 standard (Table 3). The Total %E of 4.9% (11 of 226 samples) (Figure 2 and Table 3) was due entirely to Pond N. Log mean annual conductivity for outfalls was directly correlated with log treatment volume, although Pond A (large and located coastally) probably has a disproportional impact on the correlation. Also, mean specific conductance appears to demonstrate a strong seasonal pattern related to rainfall and surface hydrology. Log mean conductivity for the 11 sample dates was inversely correlated with discharge frequency ($r=-0.85$, $p<0.001$) and mean bottom depth ($r=-0.64$, $p=0.03$), and directly correlated with log mean TSS ($r=0.65$, $p=0.03$), log mean turbidity ($r=0.55$, $p=0.08$), and log mean cadmium ($r=0.65$, $p=0.04$).

Iron (Fe)

Only two of the grab bag samples (2 of 247 = 0.81%) and none of the outfall station annual median iron concentrations (Table 3) were less than the DL for iron (0.02 mg/L). Only 4.5 percent of the grab bag samples (11 of 247 samples) originating from 8 of 24 outfall stations (Figure 2) exceeded the Chapter 17-302 iron standard (1.0 mg/L, Table 2). The %E did not change if samples only when outfalls discharged were considered. Only one outfall station exceeded the iron standard more than once (4 times at Pond G). All the outfall stations had less than 50 %E; the largest %E was 44 percent at Pond G. Mean annual iron concentrations for outfall stations were directly correlated with log mean TSS ($r=0.60$, $p=0.009$) and log mean turbidity ($r=0.59$, $p=0.01$), but inversely correlated with log mean cadmium ($r=-0.57$, $p=0.01$). Mean iron in 22 of 24 1990 storms at Pond S was 0.056 mg/L. Seasonally, mean iron concentrations for the 11 sample dates were inversely correlated with log mean nickel ($r=-0.68$, $p=0.07$). Because less than half the ponds and only 11 samples exceeded the Class III iron standard, iron does not appear to be a problem in the wet-detention ponds included in the survey. Nevertheless, Fe-nutrient internal cycling (Armstrong, et al., 1987) and Fe-Mn distributions in relation to other trace metals (Murray, 1987) remain important areas of limnological investigation and potentially important mechanisms in stormwater ponds.

Turbidity

Turbidity is an expression of the light scattering and absorbing properties of water. Water clarity (the opposite of turbidity) is important for water's potability (A.P.H.A., 1989).

Only 4.4 percent (10 of 228 samples) of grab samples originating from 5 of 24 outfall stations (Figure 2) exceeded the Chapter 17-302 turbidity standard (29 NTU, Table 2). The %E decreased slightly to about 3 percent if samples only when ponds discharged were considered. One outfall exceeded the standard three times (Pond R), and one outfall exceeded the standard four times (Pond G). Overall, turbidity measurements did not appear to be excessively high. Of the five outfalls that exceeded the turbidity standard, shallow depth (Ponds O and S), lack of vegetation (Ponds G and R), clay soils (Ponds G and H), construction activity (Pond H), wind-blown particles from a road construction company (Pond H), and high levels of algal productivity (Ponds G, H, R, and S) were all probable factors that may have increased the turbidity of samples. Log mean annual turbidity for outfall stations was directly correlated with log mean TSS (r=0.82, p=0.0001), mean annual iron (r=0.59, p=0.01) and mean annual pH (r=0.45, p=0.05), and inversely correlated with discharge frequency (r=-0.44, p=0.07) and the square root of maximum pond depth (r=-0.42, p=0.09). Seasonally, log mean turbidity for the 11 sample dates was directly correlated with log mean nickel (r=0.66, p=0.07), log mean conductivity (r=0.55, p=0.08) and possible log mean cadmium (r=0.55, p=0.10); but inversely correlated with discharge frequency (r=-0.62, p=0.04). As a measure of fine suspended matter, turbidity measurements should continue to be analyzed during stormwater projects.

pH

pH is important in trace metals speciation and mobility (Murray, 1987). As pH values are lowered, the more acidic environment makes some metals more mobile and more toxic. Low pH has been correlated with elevated mercury accumulation in fish (Irwin, 1988). The effects of elevated pH should logically have the reverse effect. Mean annual pH for outfall stations was directly correlated with mean DO (r-0.82, p<0.001), log mean copper (r=0.52, p=0.03), log mean turbidity (r=0.45, p=0.06), mean temperature (r=0.45, p=0.06), log mean chromium (r=0.41, p=0.09), and log mean TSS (r=0.41, p=0.09), and inversely correlated (r=0.53, p=0.02) with estimated percent cover with aquatic macrophytes. Seasonally, mean pH was inversely correlated (r=0.58, p=0.06) with log mean copper for the 11 sample dates (Kehoe, 1992).

The evaluation of whether or not pH values in surface water of the state violates Chapter 17-302 standards requires knowledge of "natural background" pH in receiving waters. Median outfall pH ranged from 6.52 to 8.10 (Ponds J and P, respectively); the lowest outfall pH recorded during the survey was 5.75 (Pond J) and the highest was 9.25 (Pond M). Receiving water median pH ranged from 5.07 to 7.55 (at Ponds G and L, respectively); the lowest receiving water pH was 4.70 (Pond G) and the highest was 8.70 (Pond A). However, the data provides no evidence that changes in receiving water pH resulted from discharges of stormwater from the survey ponds. The natural variability of pH was evident from the data. Differences of 1 or 2 pH units were observed at 18 outfalls and 13 receiving waters

(Kehoe, 1992), and diurnal fluctuations of 0.5 pH units have been observed in stormwater ponds (Rushton, 1993 and unpublished).

Chromium (Cr)

Only 3.3 percent (8 of 246 samples) of grab samples originating from 6 of 24 outfalls (Table 3) exceeded the Chapter 17-302 chromium standard (0.05 mg/L). Percent exceedence was unchanged if samples only when outfalls discharged were considered. Two outfall stations exceeded the chromium standard two times (Pond N and Pond T) and four others one time each (total of eight exceedences). None of the annual chromium medians exceeded the standard, and 19 annual medians plus 82 percent of grab samples (201 of 246 samples) were less than the DL. Log mean annual chromium concentrations for the outfall stations were directly correlated with mean DO ($r=0.42$, $p=0.08$), mean temperature ($r=0.42$, $p=0.08$), and mean pH ($r=0.41$, $p=0.09$), and inversely correlated with log mean manganese ($r=-0.48$, $p=0.04$). There were no seasonal correlations with chromium that were significant at the $p=0.10$ level. No rainfall data were available for chromium in Rushton (1993). Although chromium levels do not appear to have posed a problem in the wet-detention ponds that were surveyed, periodic tests (e.g., quarterly) for metals, pesticides, and other substances would help establish a data base to document changes in levels of specific substances in stormwater ponds over time.

Nickel (Ni)

Nickel was detected at 18 of 24 survey outfall situations, and only once at a concentration exceeding the Chapter 17-302 standard (0.1 mg/L) at four times the standard (Pond T outfall station, 1 of 180 grab samples = 0.6 Total %E). This concentration was probably in error, considering levels at Pond T and other ponds during the survey. Nickel levels were less than the detection limit (0.01 mg/L) in 88 percent of the grab samples (158 of 180 samples). Data were missing for three sample dates because of analysis errors. Log mean annual nickel for the outfall stations was directly correlated with log mean annual copper ($r=0.67$, $p=0.002$). Seasonally, log mean nickel for the 8 sample dates was directly correlated with log mean cadmium ($r=0.94$, $p=0.002$) and log mean turbidity ($r=0.66$, $p=0.07$), and inversely correlated with mean iron ($r=-0.68$, $p=0.07$). No rainfall data were available in Rushton (1993). Nickel levels did not appear to be problematic for the wet-detention ponds in this survey. Like chromium, nickel levels should continue to be measured periodically in order to maintain current information regarding patterns or any changes in nickel levels in regional stormwater ponds.

Manganese (Mn)

Manganese is an essential micronutrient that poses less toxicity problems in natural waters than most other contaminants (Irwin, 1988). Association of Mn with Fe in redox cycles at aerobic/anaerobic interfaces may be important for the internal cycling of nutrients and trace metals (Murray, 1987). Manganese was detected in rainfall during 6 of 32 storms in 1991, during the intensive study of a wetland-treatment system (B. Rushton, unpublished); therefore, rainfall may be an occasional source of manganese.

Manganese was detected at least twice at every outfall station, except Pond H (only once). Manganese concentrations exceeded the Chapter 17-302 Class II standard (0.1 mg/L) (no Chapter 17-302, Class III standard) only once, at the Pond C outfall station (1 of 225 samples = 0.44 Total %E) (Table 3, Figure 2). Log mean annual manganese for the outfall stations was directly correlated (r=0.50, p=0.03) with estimated pond area coverage with aquatic macrophytes, and inversely correlated with log mean chromium (r=-0.48, p=0.04). Manganese was less than the detection limit (0.01 mg/L) in only 50 percent of the grab samples (112 of 225 samples), and does not appear to be problematic in the wet-detention ponds in this survey. Like iron, the potential role of manganese for internal cycling of metals and nutrients in ponds needs investigation.

Variable Clusters

Multivariate factor, cluster, correlation, and regression analyses were used to explore empirical relationships among annual means of study variables. Results from each analysis provided supportive evidence for some significant relationships among variables for the survey ponds. Factor analysis and cluster analysis provided an interpretation of how certain variables were related through hydrologic conditions (suspended particles), through primary productivity (including aquatic macrophytes) and through pond dimensions. Many of the relationships among variables within clusters and factors were explored and supported by correlation and regression analyses. The results are detailed in Kehoe (1992).

Pond Relationships

Two methods were employed to compare water-quality variables among ponds: 1.) grouping ponds by land uses and comparing water-quality means; and, 2.) multivariate cluster analysis using water-quality variable means, and identifying ponds in clusters. The land-use evaluation provided evidence that of the four land uses, the multi-family residential ponds were statistically among the ponds with highest mean concentrations for TSS, turbidity, and three heavy metals (i.e., Zn, Fe, and Cu) (Duncan multiple range test, alpha=0.05), thus having poorest water quality. This was probably due to a higher population density and more impervious surfaces compared to other land uses. Cluster analysis using SAS Proc Cluster formed two or three clusters of ponds depending on the variables included. Again, multi-family residential ponds were grouped together in clusters with the highest mean TSS, mean turbidity, mean zinc, mean copper, and mean iron (Duncan multiple range test, alpha=0.05) (Kehoe, 1992).

CONCLUSIONS

When wet-detention is the type of stormwater management system to be built, design recommendations that optimize treatment of pollutants are well-documented, and have been used by agencies such as the Southwest Florida Water Management District as guidelines for wet-detention system regulations. The recommendations assume that the best available information has been considered. Furthermore, it is assumed that site-specific characteristics are to be considered when designing wet-detention basins. Pond design characteristics and stormwater management practices that are being utilized in the state of Florida are often based on the following broad recommendations found in the literature (SWFWMD, 1988; Stahre and Urbanos, 1990; Livingston and McCarron, 1992) and elsewhere.

1. Maximize sedimentation with pretreatment basins.

2. Lessen the effects of sediment accumulation at the outfall with a deep sedimentation area separated by a shallow aerobic area free of aquatic macrophytes at the outfall.

3. Maximize treatment time with the greatest possible separation between the inflow and the outfall.

4. Design ponds for longer detention times.

5. Prevent short-circuiting of flow with a length to width ratio of 2 to 1, and by spreading inflows with a baffle structure.

6. Prevent dead zones and stagnant water with a gradual widening of the pond from the inflow to the outfall.

7. Maximize sedimentation and biological treatment with littoral zones and shallower, open-water areas between deeper zones at the inflow and outfall.

8. Littoral zones should occupy roughly 30% of the pond, with no steeper than 4:1 slopes and preferably with 10:1.

ACKNOWLEDGEMENTS

The survey was originated as part of the Southwest Florida Water Management District's (SWFWMD) Stormwater Research Program. Invaluable editorial guidance was provided by Dr. Betty T. Rushton, Craig Dye, Clark Hull, and Dr. Ted Rochow. The author is grateful for the contributions of the District's Environmental and Laboratory staff. Sincere thanks also go to Herb Bryant, Kelly Goshorn, and Linda Eichhorn or the SWFWMD. Additionally, the author would like to thank Dr. Jerome V. Shireman, Dr. Charles E. Cichra, and Dr. Frank G. Nordlie of the University of Florida for their inspiration.

LITERATURE CITED

A.P.H.A. 1989. Standard Methods for the Examination of Water and Wastewater. Franson, M.A.H. (Editor). American Public Health Association. Washington, D.C.

Arimoto, R. and R. A. Duce. 1987. Air-Sea transfer of trace elements. Chapter 5 In: R. A. Hites and S. J. Eisenreich (Editors), Sources and Fates of Aquatic Pollutants. American Chemical Society. Washington, D.C. 558 pp.

Armstrong, D. E., J. P. Hurley, D. L. Swackhamer, and M. M. Shafer. 1987. Cycles of nutrient elements, hydrophobic organic compounds, and metals in Crystal Lake: Role of particle-mediated processes in regulation. Chapter 15 In: R. A. Hites and S. J. Eisenreich (Editors), Sources and Fates of Aquatic Pollutants. American Chemical Society. Washington, D.C. 558 pp.

Bastian, R. K. 1986. Potential Impacts of Receiving Waters. Pages 157-168 In: B. Urbonas and L. A. Roesner (Editors): Urban Runoff Quality - Impact and Quality Enhancement Technology (Proceedings). American Society of Civil Engineers. New York, New York. 477 pp.

Canfield, D. E. and M. V. Hoyer. 1988. Regional geology and the chemical and trophic state characteristics of Florida lakes. Lake and Reservoir Management (4)1:21-31.

Davies, P. H. 1986. Toxicology and chemistry of metals in urban runoff. Pages 60-78 In: B. Urbonas and L. A. Roesner (Editors): Urban Runoff Quality - Impact and Quality Enhancement Technology (Proceedings). American Society of Civil Engineers. New York, New York. 477 pp.

FAC. 1990. Chapters 17-3, 17-4, 17-25, 17-40, 17-302, 17-550, 40D-4, and 40D-40. Florida Administrative Code.

FAC. 1992. Chapters 17-3, 17-4, 17-25, 17-40, 17-302, 17-550, 40D-4, and 40D-40. Florida Administrative Guide.

Fetter, C. W., W. E. Sloey, and F. L. Spangler. 1978. Use of a natural marsh for wastewater polishing. Journal of the Water Pollution Control Federation 50:290-307.

Gilliom, R. J. and D. R. Helsel. 1986. Estimation of distributional parameters for censored trace level water quality data, 1, Estimation Techniques. Water Resources Research. 22(2):135-146.

Irwin, R. J. 1988. Impacts of toxic chemicals on Trinity River Fish and Wildlife. United States Fish and Wildlife Service. Contaminants Report. November, 1988. 82 pp.

Kehoe, M. J. 1992. Water-Quality Survey of Twenty-four Stormwater Wet-Detention Ponds (Final Report). Southwest Florida Water Management District. Brooksville, Florida. (Revised, May 1993).

Livingston, E. H. 1986. Stormwater regulatory program in Florida. Pages 249-256 In: B. Urbonas and L. A. Roesner (Editors): Urban Runoff Quality - Impact and Quality Enhancement Technology (Proceedings). American Society of Civil Engineers. New York, New York. 477 pp.

Livingston, E. H. and E. McCarron. 1992. Stormwater Management - a guide for Floridians. Florida Department of Environmental Regulation. Tallahassee, Florida. 72 pp.

Lotus Development Corporation. Lotus 1-2-3 Release 2.01. Cambridge, MA., 1986.

Marsalek, J. 1986. Toxic contaminants in urban runoff: A case study. Pages 39-58 In: H.C. Torns, J. Marsalek and M. Desbordes (Editors); Proceedings of a NATO workshop on Urban Runoff Pollution. Springer-Verlag, Berlin. 842+ pp.

McClave, J. T. and F. H. Dietrich. 1982. Statistics (2nd Edition). Dellen Publishing Company. Santa Clara, California. 766 pp.

Murray, J. W. 1987. Mechanisms controlling the distributions of trace elements in oceans and lakes. Chapter 6 In: R. A. Hites and S. J. Eisenreich (Editors), Sources and Fates of Aquatic Pollutants. American Chemical Society. Washington, D.C. 558 pp.

Randall, C. W., K. Ellis, T. J. Grizzard, and W. R. Knocke. 1982. Urban Runoff pollutant removal by sedimentation; Pp. 205-218 In: W. DeGroot (Editor), Proceedings of the Conference on Stormwater Detention Facilities. American Society of Civil Engineers. New York, New York.

Rushton, B. T., H. C. Hull, Jr., and C. W. Dye. 1989. Discharge water quality of permitted wet-detention systems. Pages 8A-1 to 8A-13 in Water: Laws and Management, Symposium. American Water Resources Association. September, 1989.

Rushton, B. T., and C. W. Dye. 1993. An In-depth analysis of a wet-detention stormwater system. Southwest Florida Water Management District. Brooksville, Florida. (In print).

SAS Institute, Inc. 1988. SAS/STAT User's Guide. Release 6.03 Edition. Cary, North Carolina. 1028 pp.

Shih, S. F. 1981. Upland marsh for water quality control. Proceedings of the American Society of Civil Engineers: 107 (EE4). August, 1981. American Society of Civil Engineers. New York, New York.

Southwest Florida Water Management District. 1988. Management and storage of surface waters, permit information manual - Volume I. Southwest Florida Water Management District. Brooksville, Florida.

Stahre, P. and B. Urbonas. 1990. Stormwater detention for drainage, water quality, and CSO management. Prentice-Hall. Englewood Cliffs, New Jersey. 338 pp.

Terstriep, M. L., D. C. Noel, and G. M. Bender. 1986. Sources of urban pollutantants - Do we know enough? Pp. 107-121 In: B. Urbonas and L. A. Roesner (Editors): Urban Runoff Quality - Impact and Quality Enhancement Technology (Proceedings). American Society of Civil Engineers. New York, New York. 477 pp.

United States Environmental Protection Agency. 1979. Methods for Chemical Analysis of Water and Wastes. USEPA Report (EPA 600/4-79-020). Washington, D.C.

United States Environmental Protection Agency. 1983. Results of the National Urban Runoff Program (NURP). Executive Summary; Volume I - Final Report; Volume II - Appendices; Volume III - Data Appendix. USEPA, Washington, D.C.

Wanielista, M. P. 1978. Stormwater Management: Quantity and Quality. Ann Arbor Science Publishers, Inc. Ann Arbor, Michigan. 383 pp.

Wetzel, R. G. 1975. Limnology. W. B. Saunders Company. Philadelphia, Pennsylvania. 743 pp.

Whalen, P. J. and M. G. Cullum. 1988. An assessment of Urban land use/Stormwater runoff quality relationships and treatment efficiencies of selected stormwater management systems. Technical Publication 88-9. South Florida Water Management District, Water Quality Division. West Palm Beach, Florida. 56 pp.

Whipple, W., Jr. 1979. Dual-purpose detention basins. Journal of the Water Resources Planning and Management Division; Proceedings of the American Society of Civil Engineers; v. 105, No. WR2. American Society of Civil Engineers. New York, New York.

Whipple, W., Jr. and J. V. Hunter. 1981. Settleability of urban runoff pollution. Journal of the Water Pollution Control Federation. 53(12):1726-1731.

WETLAND CREATION: A VIABLE MITIGATION OPTION

Glenn Lowe
Chief Environmental Specialist
Robert J. Epting, Ph.D.
Lisa M. Grant
Environmental Specialist
ST. JOHNS RIVER WATER MANAGEMENT DISTRICT
Department of Resource Management
P.O. Box 1429
Highway 100 West
Palatka, Florida 32178-1429

ABSTRACT

An assessment of compliance and success of permitted projects with mitigation suggests that wetland creation can be a viable mitigation option when properly designed and implemented. Saltwater creation sites were found to have the least problems, followed by freshwater herbaceous, and then freshwater forested creation. Although on-site creation will continue to be used as a mitigation option, there are probably many instances where regional mitigation banking can be a more effective mitigation option.

INTRODUCTION

In the St. Johns River Water Management District (District) wetland creation is one of the mitigation alternatives available as part of the Management and Storage of Surface Waters (MSSW) Rule, Chapter 40 C-4, F.A.C. and also as a part of the Wetland Resource Management (WRM) Program, Chapter 17-312, F.A.C. Other mitigation measures include wetland enhancement or restoration, or the preservation by conservation easement or fee simple title transfer of both wetlands and uplands.

Between October, 1984 and April, 1993 the District issued 5098 MSSW permits and 788 WRM permits. Of these permits, 973 MSSW and 213 WRM required mitigation for wetland impacts. The majority of permits which required mitigation involved wetland creation either as the sole mitigation measure or in combination with other alternatives. Wetland "creation" has been so prevalent as a method to offset wetland impacts that the phrase has often been used synonymously with the term "mitigation".

The District has conducted compliance reviews of projects involving mitigation since inception of the current MSSW program in 1984. Staff has additionally conducted more detailed assessments of wetland creation (Lowe, et al., 1988) which laid the framework for standardized data forms and design of a mitigation database for assessment and tracking purposes.

This report summarizes the results of the information currently contained in the mitigation database (i.e., 374 creation sites representing 193 permits); some permits entailed the creation of more than one site. Although the summary does not yet include all projects for which compliance reviews have been conducted, it is likely to be representative of the overall program. The results suggest that wetland creation, in conjunction with a good project design, can be successfully employed as a viable mitigation option.

METHODS AND ASSESSMENT PARAMETERS

The assessment begins with a review of the administrative requirements of the issued permit. These requirements include the recording of legal documents such as conservation easements, and the submittal of monitoring reports, completion reports, and as-built surveys.

The administrative review is followed by a site inspection. The sites were compared to the permitted plans regarding size, location, and species composition. The percent survival of planted species, growth rates and vigor, invasion by nuisance species, and other factors which make up the success criteria of the selected permit, were also evaluated.

Also included, is an evaluation of the likelihood of achieving the success criteria specified in the permit, both with and without any additional enforcement action by the District, and an assessment of similarity to reference or adjacent wetlands is included for those projects where relevant.

A wildlife section on the data forms includes information on current and long-term wildlife value, both with and without consideration of surrounding land use; and, an assessment of the current and long-term similarity of the constructed wildlife habitat to the type of habitat anticipated when the permit was issued. (see Appendix A for data forms).

RESULTS

Between 1984 and April, 1993, the District issued 5,098 Management and Storage of Surface Waters (MSSW) permits and 788 Wetland Resource Management (WRM) permits. Of these 1,086 required mitigation for wetland impacts; 873 MSSW (17%) and 213 WRM (27%). Staff has administratively reviewed and is tracking 384 (35%) of these permits in a mitigation database and has inspected 374 wetland creation sites for 193 of these permits. Some permits entailed more than one wetland creation site; therefore, the analysis will always indicate a greater number of individual creation sites than permits which involved wetland creation. The 374 wetland creation sites inspected included freshwater herbaceous (37%, 91 ha), freshwater forested (57%, 238 ha), saltwater herbaceous (4%, 20 ha), and saltwater forested (1%, 4 ha).

The vast majority of freshwater creation sites were constructed in accordance with permitted plans and conditions (Table 1). Only 8 sites (2%) were found not to have been constructed. The data shows that the earlier issued permits had more problems than those issued more recently (Figure 1).

The assessment includes an evaluation of other site characteristics, which may be useful for long-term evaluation of individual sites (Table 2).

One-half of creation sites was in compliance with permitted plans and conditions at the time of inspection and two-thirds had at least a moderate likelihood of achieving the success criteria specified in the permit without enforcement action (Table 3). While likelihood of achieving the success criteria specified in the permit is higher for saltwater systems, the frequency of compliance is similar for all types of creation. The current and potential similarity of wildlife value of the created habitat to that intended in the permit, was at least moderate for a majority of sites (Table 4). For all types of creation sites combined, 58% currently have at least moderate wildlife value and 72% are expected to have moderate value in the long term.

Corrective actions required by permit condition, or considered necessary to meet the intent of the permit, are more frequent for forested than herbaceous freshwater creation (Table 5). Regrading the site was necessary more than twice as often for forested than herbaceous systems. Replanting and exotic/nuisance species removal were the most common site problems.

```
      |
1984  |  ****************************************        100.0
      |
1985  |  ***********************************              85.7
      |
1986  |  ****************************                     66.6
      |
1987  |  *********************************                80.0
      |
1988  |  *******************************                  75.0
      |
1989  |  ************************                         58.3
      |
1990  |  *******************                              44.1
      |
1991  |  *******************                              46.1
      |
1992  |  ***********                                      27.2
      |
       ----|---|---|---|---|---|---|---|---|---|--
          10  20  30  40  50  60  70  80  90  100

               Frequency Percent
```

Figure 1. Percent by year with mitigation judged to have had a problem with mitigation plans or conditions at the time the permit was issued.

Table 1. Freshwater creation site construction parameters (percent of sites constructed correctly).

PARAMETER	HERBACEOUS	FORESTED
Size (Acres)	90	91
Excavation Grade	86	71
Type of Wetland	96	89
Species Planted	88	83
Planting Elevation	84	73
Site Stabilized	85	81

Table 2. Freshwater creation site characteristics (percent of sites exhibiting characteristics).

CHARACTERISTIC	HERBACEOUS	FORESTED
Evidence of Normal Growth	85	68
Evidence of Woody Recolonization	23	35
Exotic/Nuisance Coverage >10%	25	37
Evidence of Colonization by "Upland" Species	42	54
Algae Cover >10%	27	22
Hydrologic Problems	21	38

Table 3. Wetland creation site compliance (percent in compliance with permitted plans and conditions) and likelihood of achieving success criteria without enforcement (percent with moderate to high evaluation scores)

TYPE OF CREATION	% COMPLIANCE	% SUCCESS
Freshwater Herbaceous	59	66
Freshwater Forested	37	66
Saltwater Herbaceous	43	80
Saltwater Forested	50	100
All Types	46	
Permit Compliance	49	

Table 4. Similarity of wildlife value of created habitat to that intended in the permit (percent of sites with moderate to high evaluation scores).

	HERBACEOUS	FORESTED
Current	75	53
Potential	77	66

Table 5. Corrective actions considered necessary for freshwater creation sites to meet the intent of the permitted mitigation (percent of sites).

ACTION	HERBACEOUS	FORESTED
Regarding	10	27
Replanting	26	53
Remove exotic or nuisance species	34	49
Control access	8	17
Stabilize site	15	23

DISCUSSION

The compliance, success and wildlife results for wetland creation reported here are more positive than those of earlier studies (FDER, 1991; Erwin, 1991). The higher compliance and success rates within our District are thought to be related primarily to early recognition of the need for sufficient compliance staffing to provide a level of oversight and assistance necessary to gain the cooperation of permitees. This interpretation is supported by the observation that the vast majority of creation sites were actually constructed, whereas earlier reports noted lack of construction as a fundamental compliance problem. Important secondary factors are generally fewer exotic species, due to a more temperate geographic location, and less hydrologic alteration of the landscape than in south Florida.

When a project is not in compliance, there is often a combination of administrative and site problems. The enforcement effort required to achieve compliance varies considerably, depending on the severity of the problem. Many permits require relatively minor action, such as a reminder phone call of a past-due monitoring report. Compliance with administrative reporting is expected to continue to improve as a result of the mitigation tracking database. Older projects with more serious site problems may require extensive follow-up over many months for resolution. Earlier site follow-up of new projects will identify problems early rather than late in the monitoring time frame, minimizing compliance cost and effort for both the District and permittee.

As anticipated, saltwater creation sites appear inherently more successful than freshwater sites. The contrast between freshwater and saltwater systems in achieving a proper grade may be attributed to data available in the design of the system. Saltwater system design is based upon established and recorded tidal ranges, usually from gauging stations in the immediate area. The design grade of freshwater systems is usually based on adjacent wetland grades without extensive knowledge of existing hydroperiods. Only occasionally have freshwater levels been recorded over sufficient time that design can be based on historic data.

In recent years, additional consideration has been given to design of forested systems. Some permits have included detailed evaluations of pre and post-construction hydrology, others have included phase planting plans to allow for stabilization of water tables and soils.

Drought conditions over much of the District often made a field determination of whether or not the grade and planting elevation of a site was correct for freshwater systems, particularly when there was not an adjacent natural wetland for comparison. Sites that were indicated as having incorrect planting elevation, either had species clearly planted at elevations which would not support their continued survival and growth, or the reviewer was unable to determine if the plants were planted at the correct elevations. It should also be noted that the "correct" hydroperiod of created freshwater systems can be difficult to determine during a one-day site inspection, as most freshwater system water levels fluctuate seasonally and following storm events. During a field inspection in saltwater systems, the evidence of tidal fluctuations is readily observable and "correct grading" is a more direct determination.

Requiring planting inventories, in addition to a mitigation as-built as a standard permit condition, has improved the evaluation process. Results of an earlier mitigation assessment indicated the need for submittal of a mitigation-specific as-built survey in order to determine whether the size, grading and planting elevations were in accordance with the permitted design. This requirement has been included on most permits issued since 1989. In the absence of an as-built survey, grading, like size, was not a measurement, but rather an estimate. In this case, the estimate was largely based upon whether the hydrology was correct. If the hydrology appeared correct, the assumption was made that the grading was correct.

Wetland creation is a relatively new science. Some of the early creation efforts which were "successful" were due mostly to luck and natural ecological succession. Many sites were not "successful", resulting in the conclusion that creation doesn't work. It is recognized that the current quantitatively based success criteria do not allow an assessment of the variability inherent in natural systems, nor do they allow for a broader functional assessment, based on professional ecological judgement. Wildlife value results, although subjective, can be useful for comparing the actual and potential result of the creation effort with that anticipated at realistic time frames during the permit review process. Preliminary analysis of the wildlife value results, which factor in the surrounding land use, indicate that location of the creation site relative to developed areas is an important determinant of wildlife use. (This is equally true for natural wetlands). This may lead to an acknowledgement that wetland acreage comparisons (lost vs. created) are not a representative measure of the impact of a project, or of the effectiveness of a regulatory program.

The nature of a regulatory program requires a short-term assessment of impacts and compensatory mitigation. This has caused the cost of creation to increase in an effort to accelerate natural succession, in order to quickly reach a point at which the permittee can be released from responsibility for the creation effort. Additional research is needed to evaluate the ecological succession of man-made systems. With proper planning and reasonable expectations this, can occur as a part of the regulatory process.

Mitigation and permitting requires a regional perspective to ensure that cumulative and secondary impacts to an ecosystem will be capable of being offset in the long term. There are probably many instances where regional creation, enhancement or restoration efforts (mitigation banks) between a number of patients can be effectively used to ensure a greater likelihood of long-term success, and to address certain impacts associated with development of the surrounding uplands. Development of mitigation banking would also allow the use of on-site creation in those situations where it is the most appropriate option, rather than the "only" option.

Long-term assessment of creation will likely show that if properly designed and implemented it can be used in combination with other mitigation alternatives, and a responsible site plan to effectively offset many of the adverse effects of impacts to wetlands.

ACKNOWLEDGEMENTS

We thank the staff of the Department of Resources Management for its ongoing support and assistance.

LITERATURE CITED

Erwin, Kevin L. 1991. An evaluation of wetland mitigation in the South Florida Water Management District, Volume 1. West Palm Beach, Florida.

Florida Department of Environmental Regulation (FDER). 1991. Report on the effectiveness of permitted mitigation. Tallahassee, Florida.

Lowe, Glenn, David Walker, and Barbara Hatchitt. 1989. Evaluating manmade wetlands as compensation for the loss of existing wetlands in the St. Johns River Water Management District. Proceedings of the 16th Annual Conference on Wetlands Restoration and Creation. Tampa, Florida. pp. 109-118.

Appendix A.1 ADMINISTRATIVE REVIEW DATA FORM

PERMIT Mitigation

Project: ______________________________
Form: date ___/___/___ by _ _ _
See back [ ]
Data keyed ___/___/___ by _______

Permit # ____ - ________ - ___________ (no "A")
Check if related #/initials are new or update [ ].
Related # ____ - ________ - _____A_____ (w/ "A")
Compliance by _ _ _ (use three initials)

Mitigation	# of sites if yes
MI CReation	Y N
MICR Freshwater Woody	_____
MICR Freshwater Herbaceous	_____
MICR Saltwater Woody	_____
MICR Saltwater Herbaceous	_____
MI REstoration	Y N
MIRE Freshwater Woody	_____
MIRE Freshwater Herbaceous	_____
MIRE Saltwater Woody	_____
MIRE Saltwater Herbaceous	_____
MI ENhancement	Y N
MIEN HYdrologic improvement	_____
MIEN EXotic Removal	_____
MIEN PLanting	_____
MIEN OTher (specify on back)	_____
MI Preservation oN-site	Y N
MIPN WEtland preservation	_____
MIPN UPland preservation	_____
MI Preservation oFf-site	Y N
MIPF WEtland preservation	_____
MIPF UPland preservation	_____

DEsign (plans and permit)	
DE mitigation PLan w/permit	Y N
DE Soils Infor. with plan	Y N
DE Planting Scheme w/plan	Y N
DE MOnitoring reports req	AN BI QU MO N
DE Success Criteria specified	Y N
DE Exotic Control specified	Y N
DE other MAint. required	Y N
DE Correct Measures specified	Y N
DE AS-Built required	Y N
DE permit Condition Problems	Y N
DE MUlch required	Y N
DE Hydrologic Monitoring req	Y N
DE Planting Required	Y N
DE Plant Inventory required	Y N
DE no. of MOnitoring YeaRs	___
DE Deed Restrictions	Y N
DE Conservation Easement	Y N
DE COnstruction start DaTe:	___/___/___

cc: permit file

permit COmpliance (administrative)	
CO MOnitoring reports	
COMO REceived	Y N NR
COMO DUe DaTe	/ /
COMO REceived DaTe	/ /
CO Easement Restrictions	
COER REceived	Y N NR
COER DUe DaTe	/ /
COER EXhibits received DaTe	/ /
COER Legal Review	Y N
COER APproved DaTe	/ /
COER Certified copy Due DaTe	/ /
COER Cert. copy Received DaTe	/ /
CO AS-built	
COAS REceived	Y N NR
COAS DUe DaTe	/ /
COAS REcevied DaTe	/ /
CO Hydrologic Monitoring	
COHM report REceived	Y N NR
COHM DUe DaTe	/ /
COHM REceived DaTe	/ /
CO Plant Inventory REceived	Y N NR
COPIRE DaTe	/ /
CO PErmit compliance	
COPE In Compliance	Y N
COPE measures NEcessary	ADMIN SITE BOTH N
COPE Schedule Revisit	Y N
COPESR DaTe	/ /
ACtion measures	
AC Permittee Notified	Y N
ACPN DaTe	/ /
ACPN BY	CALL MTG LTR
AC DUe DaTe	/ /
AC Response Received	Y N
ACRR DaTe	/ /
AC Problems Corrected	Y N SOME
ACPC DaTe	/ /
AC EXtension letter	Y N
ACEX DaTe	/ /
TIckle reminder	
TI ACtion COMO COER COAS COHM COPESR AC	
TI DaTe	/ /

Appendix A.2 SITE EVALUATION DATA FORM

SITE mitigation

Date Inspected ___/___/___
Inspected by __ __ __ (use three initials)

MItigation
MItigation TYpe CR RE EN
MITY CReation FW FH SW SH
MITY REstoration FW FH SW SH
MITY ENhancement HY EX PL OT

DEsign (plans and permit)
DE Contig to exist Wetland Y N
DE STormwater treat. area Y N
DE Littoral Zone Y N

permit COmpliance (site)
CO MItigation construction
COMI STarted Y N
COMI STart DaTe: _/_/_
COMI COmpleted Y N
COMI COmpleted DaTe: _/_/_

CO As Designed Y N
COAD correct GRade Y N
COAD correct SIze (acres) Y N
COAD correct TYpe Y N
COAD correct SPecies Y N
COADSP SIze Y N
COADSP NUmbers Y N
COADSP SPacing Y N
COAD correct Planting Elev Y N
COAD site STabilized Y N

Site Information

SI PLanting survival
SIPL HErbaceous
0-50%: L 50-80%: M >80%: H NP NR
SIPL WOody
0-50%: L 50-80%: M >80%: H NP NR

SIPL evidence of Normal Growth
SIPLNG HErbaceous plantings Y N NA
SIPLNG WOody plantings Y N NA

SI COverage (recolonization or total)
SICO HErbaceous
0-10%: L 10-50%: M 50-80%: H >80%: XH NA
SICOHE TYpe REColonization TOTal
SICO WOody
0-10%: L 10-50%: M 50-80%: H >80%: XH NA
SICOWO TYpe REColonization TOTal

SICO evidence of Woody Recolonization Y N
SICO evidence of UPland invasion Y N

Permit # ____ - ________ - ________ (no "A")
Site ID __ __
Data entered ___/___/___ by ____________

SI coverage by ALgae
none:N <5%:L 5-10%:M 10-50%:H >50%:XH
SI coverage by EXotics/nuisance species
none:N <5%:L 5-10%:M 10-50%:H >50%:XH

SI Mulch Used Y N
SI HYdrologic problems Y N

SI similar to ReFerence wetland
SIRF GRade H MH M ML L NA
SIRF HYdroperiod H MH M ML L NA
SIRF HErb spp H MH M ML L NA
SIRF WOody spp H MH M ML L NA

SI similar to ADjacent wetland
SIAD GRade H MH M ML L NA
SIAD HYdroperiod H MH M ML L NA
SIAD HErb spp H MH M ML L NA
SIAD WOody spp H MH M ML L NA

SI mitigation SUccess
SISU CRiteria met Y N NA NS
SISU POtential to meet success criteria
SISUPO W/O action H MH M ML L NS
SISUPO W/ Action H MH M ML L NS

SI wildlife VAlue & similarity
SIVA CUrrent value & similarity
SIVACU with LandUse H MH M ML L
SIVACU in a BoX (w/o lu) H MH M ML L
SIVACU SImilarity H MH M ML L
SIVA 20-yr POtential value & similarity
SIVAPO with LandUse H MH M ML L
SIVAPO in a BoX (w/o lu) H MH M ML L
SIVAPO SImilarity H MH M ML L

ACtion measures (see back)
AC RePlanting Y N R
AC ReGrading Y N R
AC REmove exotics Y N R
AC Control Access Y N R
AC STabilize Y N R
AC other Maint. Necessary Y N R

CO SIte compliance
COSI Maint. Performed Y N
COSIMP DaTe _/_/_
COSI In Compliance Y N
COSI Schedule Revisit Y N
COSISR DaTe _/_/_

BIOGENIC WETLANDS AND INADEQUATE UNDERSTANDING OF INDICATOR PLANT SPECIES POSE PROBLEMS FOR LAND MANAGEMENT AND USE IN COOL CLIMATES

Jay D. McKendrick
UNIVERSITY OF ALASKA FAIRBANKS
Alaska Agricultural and Forestry Experiment Station
533 East Fireweed, Palmer, Alaska 99645
(907) 746-9463

ABSTRACT

Successfully managing and restoring wetlands in cool climates depends on understanding of the genesis processes responsible for these features of the landscape. A national guideline for classifying wetlands was created to permit decisions on classification to be made by non-technical individuals. Where conditions are such that the judgement call is obvious, that is probably a workable approach. For some locations in Alaska, this is creating management problems, because the distinction between wetlands and non-wetlands is nebulous. The national standards for classification do not take into account the natural formation and disappearances of wetland soils developed as a result of plant succession and permafrost. Through this natural process, a given location on the landscape can technically be a wetland at advanced stages of plant succession, and a non-wetland after wildfire destroys the vegetation. Another aspect of the classification system relies on plant indicators. The current listing of plant species can cause misclassification on sites. If the federal management system relied more on installing technical experts at the local level and giving them the responsibility for wetland classification than on a universal and simplified classification system a more reasonable process that meets the needs of regional and local human populations might result. Attempting to establish a simplified system to cover a diversity of land and climate types across the U.S. from the subtropics to the Arctic is theoretically appealing, but laden with impracticality.

INTRODUCTION

The federal policy regarding development and use of wetlands, coupled with the 1989 "no net loss" policy from the Bush Administration, has created confusion for wetland management of much of Alaska. If the federal criteria (hydric soil characteristics, hydrophytic vegetation, and hydrology) are used, most of Alaska can be classified as wetland. In an attempt to simplify the process so that non-technical individuals could classify as sites either wetland or non-wetland, has resulted in an inferior method for much of Alaska. In the temperate zone states, wetlands are perhaps more distinct from non-wetlands because hydrology, soils, and vegetation features contrast with the adjacent non-wetlands. That is not true in Alaska, where wetlands and adjacent non-wetlands often have plant species in common, and the natural process of plant succession can create wetlands. These circumstances are creating costly problems with the use of lands in Alaska.

In this paper, the aspect of biogenic wetland phenomenon associated with permafrost and soil wetness is described to illustrate the difficulty of the federal wetland classification for this state. A few examples of indicator plant species that should be reevaluated, with regard to their relationships with wetlands, are also given.

In 1990, the Domestic Policy Council Task Force on Wetlands solicited information and opinion regarding the implementation of wetland policies (Federal Register, 1990). To assist that investigation, a technical panel in Anchorage, Alaska provided the committee on September 7, 1990 statements on the cyclical nature of vegetation and soils in cold regions. Subsequently, others have published similar information for Alaska (Ping, et al., 1992; Knight, 1993).

VEGETATION AND FIRE AFFECT SOIL DEVELOPMENT

In cold climates, particularly those with permafrost, plant succession can be the determining factor that affects hydric soil formation and hydrology. Wildfire periodically resets plant succession to early seres, erasing the hydric characteristics of soils and altering the hydrology of sites.

Management of vegetational resources in the Alaskan territory focused early on the influences for fires (Heintzleman, 1936). Nearly 20 years ago, the effects from fire on cold-region soils were described (Pettapiece, 1974). More recently, these factors have been reiterated with emphasis on how fire and plant succession cycles alter soil properties relative to hydric conditions and, hence, wetland classifications (Ping, et al., 1992; Knight, 1993).

The role of living organisms on the soil forming process becomes significant as a factor controlling soil temperature and hydrologic conditions. The process involves the development of thick layers of moss and vegetation litter (peat) on the surface of mineral soil. This mat insulates the soil profile, reducing the annual depth of thaw, and the permafrost table rises near the soil surface.

Permafrost impedes drainage, often perching water in the moss layer itself. Periodically, natural fire from lightning strikes burn the vegetation, removing the insulation and causing soil profiles to drain.

The repeating vegetation sequence from bare soil to the climax muskeg bog is illustrated in the schematic diagram (Figure 1). Initial plant communities usually consist of grasses and forbs. Shrubs and hardwood forest invade, followed by a mixed forest of hard and softwood species. Ultimately, the hardwoods are replaced, and the softwood forest becomes predominantly open stands of black spruce (Picea mariana), or a muskeg bog. When fire destroys this community, it may remove the insulating moss and litter layer and cause the soil to drain.

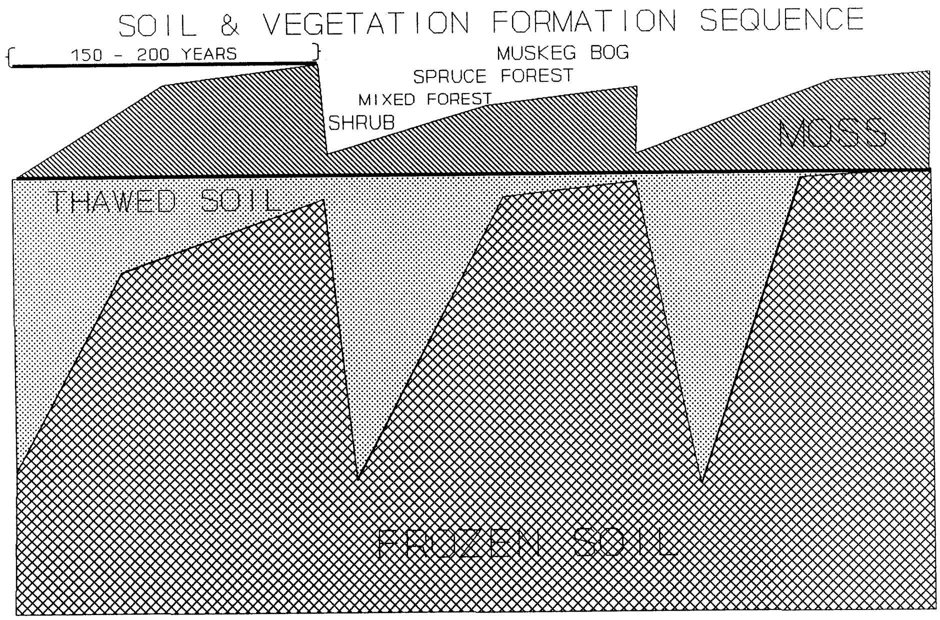


Figure 1. Schematic diagram of plant successional and permafrost sequence for interior Alaska.

In the drained state, soils lose their hydric characteristics, and hydrophytic vegetation may, or may not, disappear from the location. The site would not be classified as a wetland at that stage of succession. At the ultimate end of plant succession, the soil is hydric and supports hydrophytic vegetation. The period of this fire cycle in Alaska typically ranges from 50 to 200 years. Interior Alaskan forests are seldom more than 150-200 years in age.

This natural process is analogous to draining swamps and/or creating impoundments through management to either eliminate or create wetlands. The same laws of nature are in force, and it should come as no surprise to land managers that natural shifting between wetland and non-wetland at a given site can occur naturally.

The accumulation of sphagnum and other mosses also forms administratively-definable wetlands on slopes exceeding 100% in parts of Alaska. Such moss layers absorb moisture. The trapped water supports hydric species on such slopes, which in temperate zones, would be well-drained. Disturbances of the moss layer can result in a disappearance of the wetland features.

EXAMPLES OF PLANT SPECIES FOR WHICH EXCEPTIONS TO THEIR OBLIGATE INDICATOR STATUS HAVE BEEN OBSERVED

Observations of some indicator species listed in the accompanying table show that refinement of the standards are in order. Of the nine species contained in this list, the observations indicate the designation currently used is weighted too much in favor of the plant as an indicator of wetness. One exception is for the species Oplopanax horridum, for which a higher status may be appropriate for the regional category.

In most instances, the differences are related to the function of plant species during secondary plant succession. Such would be expected, if the criteria were developed with only the climax communities in mind. However, serial communities are to be expected in the vicinity of major human activities and as a response to natural disturbances (i.e. fire, floods, land slides, etc.). Therefore, the secondary successional stages and functions of plants under those conditions should be taken into account when determining wetland status for sites in question.

PRACTICAL IMPLICATIONS

If the land is indeed a wetland having particular value as wildlife habitat, watershed, etc., then developments conflicting with those values is a matter for concern, and that should be obvious. If the wetland status is not readily apparent, and the determination hinges on conditions that are either obscure or subject to change by natural ecological variations, then conditions are ripe for land use disputes. Streams coursing across deserts are readily recognized for their high water and habitat values. Boundaries between the desert vegetation and the riparian habitats along streams are usually distinct. But, hundreds of miles of either forests, shrublands, or tundra, with only subtle variations in soils and vegetation, are difficult to categorize. With those conditions, the ecotones between wetlands and non-wetlands are broad and vague, and the room for error large. Furthermore, the special attraction of indigenous wildlife to one part or another of such an ecosystem may be governed by the lack of wetness. Often it is the driest sites that are seasonally most attractive to wildlife in the boreal and tundra regions, as they escape from either the wet or seek diversion from that most constant.

It is difficult for committees to develop standard criteria for regions and the nation as a whole. Initially, there were undoubtedly provisions for refining these plant indicator criteria as new information became available. However, the danger with such guidelines is they often become unalterable once the information becomes published. Funding to revise manuals is usually difficult to obtain, and the course of least resistance is to let conditions remain stationary. Unfortunately, for many citizens of Alaska, this approach has economic implications when landowners suddenly discover they cannot improve lands purchased in good faith for agriculture, homesites, businesses, etc., because such would violate wetland regulations.

Listing of plants observed to differ from the published list (Reed, 1988) of wetland species.			
Species Name	Regional Wetland Category	National Wetland Category	Observations
Agropyron macrourum	Facultative	Facultative	Commonly occurs only on driest sites in Arctic.
Agropyron violaceum	Facultative	Facultative	Commonly occurs only on driest sites in Arctic.
Alopecurus aequalis	Obligate	Obligate	Colonizes well-drained sites in the vicinity of wetlands.
Beckmannia syzigachne	Obligate	Obligate	Colonizes well-drained sites (gravel) in Arctic.
Braya purpurascens	Facultative	Facultative	High fidelity to colonizing well-drained, rocky soils in Arctic.
Carex rotundata	Obligate	Obligate	Colonizes well-drained soil and gravel in vicinity of wetlands.
Lupinus nootkatensis	Facultative	Facultative	Occurs on drained soils and slopes along arctic streams.
Oplopanax horridum	Facultative Wet	Facultative Wet	High fidelity to wet soils. Building on these sites will result in wet basements and frost heaved roads.
Puccinellia langeana	Obligate	Facultative Wet - Obligate	Consistently colonizes well-drained gravel and saline mesic soils in the Arctic.

CONCLUSIONS

Developing a single designation standard for the entire U.S. is illogical, either from an ecological or a habitat value perspective. A general policy statement indicating goals for management is more realistic. Federal criteria should be general and flexible enough to take into account the cyclical nature of cold-climate environments which naturally exhibit alternating characteristics of hydric soils and hydrophytic vegetation and non-hydrophytic vegetation. If such a policy were written with understanding of the plant ecology and soil forming processes, it should be feasible to permit utilization of natural resources to occur on such sites, with the proviso that soils and topographic integrity be replaced after the development or extraction is completed; thus, allowing natural processes to operate as they did prior to the development.

Designing the national wetland classification process to be used by non-technical people requires rigidity in the criteria and risks introducing errors and unsound conclusions. It is an error in judgement to attempt such rigidity in land management policy where diversity is great. With the current policy, technical judgement is confined to the highest levels perhaps in an attempt to eliminate need for expertise at local levels. A much better management is achieved by placing the professional expertise at the local levels and delegating technical decisions to those individuals, as opposed to placing the expertise at the national level and diffusing technical judgement via guidelines and technical manuals for non-technical personnel. For land resources in a nation as diverse as the U.S., top-down management results in poorer management, unworkable systems, local and regional dissatisfaction, in addition to wasting human resources on litigation and related processes, as opposed to on-the-ground achievements. The imprudence of that was clearly demonstrated in the former U.S.S.R. as a major handicap for the people, their economy and the environment.

To many Alaskans, it appears as if unfortunate errors in more densely-populated locations of the U.S. are over compensated for in regulations with respect to the sparsely-populated regions of this state. This is not to say land management in Alaska is without fault. It is intended to emphasize the significance of a management policy that reflects national goals tempered by local environmental conditions and values and human needs. The implementation of management should be a product of sound judgement and based on substantial experience. The attempt to protect the environment for the betterment and enjoyment of mankind often becomes so complicated that mankind has no place in the environment.

ACKNOWLEDGEMENTS

Funding for this paper was provided by the Alaska Agricultural Experiment Station and BP Exploration (Alaska), Inc. Opinions expressed are those of the author and may, or may not, reflect that of the sponsors of this work. Gratitude is expressed to Robert Sener and Peg Banks for reviewing the text.

LITERATURE CITED

Federal Register. 1990. 55(143):30279-30280

Heintzleman, B. F. 1936. Reindeer grazing. Senate Document, 74th Congress 2nd Session 199:(1936) pp. 581-598.

Knight, Charles W. 1993. Wetlands: an editorial perspective. Agroborealis 25(1):13-16.

Pettapiece, W. W. 1974. A hummocky permafrost soil from the subarctic of northwestern Canada and some influences of fire. Canadian Journal of Soil Science. 54(4):343-355.

Ping, C. L., J. P. Moore, and M. H. Clark. 1992. Wetland properties of permafrost soils in Alaska. In: Kimble, J. M. 1992. Proceedings of the Eighth International Soil Correlation Meeting (VIII ISCOM): Characterization, classification, and utilization of wet soils. USDA, Soil Conservation Service, National Soil Survey Center, Lincoln, NE. pp. 198-205.

Reed, Porter B. 1988. National list of plant species that occur in wetlands: Alaska (Region A). U.S. Fish and Wildlife Service, U.S. Department of the Interior, Biological Report 88(26.11). 86 pp.

A COMPARISON OF FLORISTIC COMPOSITION AND SPECIES RICHNESS WITHIN AND BETWEEN CREATED AND NATURAL WETLANDS OF SOUTHEASTERN NEW HAMPSHIRE

Donald J. Padgett and **Garrett E. Crow**
Department of Plant Biology
UNIVERSITY OF NEW HAMPSHIRE
Durham, New Hampshire 03824

ABSTRACT

The vegetation of six natural wetlands and six created wetlands of southeastern New Hampshire was analyzed. The presence of each plant species within 0.01 hectare quadrats in each of twelve wetland sites was recorded to estimate species richness. The study revealed the composition of each wetland plant community varied, but the relative number of species within each wetland did not differ greatly from site to site. Interestingly, species similarity indices between sites were low within natural sites and created sites, as well as between natural and created sites. Problems and implications of introducing rare and non-native wetland species as part of a mitigation project are discussed.

INTRODUCTION

Plant communities of wetlands are important and vital components of overall productive ecosystems. Plant community composition and species richness of natural wetland ecosystems may involve several factors. Two primary factors providing for wetland plant communities are the hydrological conditions and the associated soils or substrate of an area (Niering, 1989; Jacobson and Jacobson, 1989; Larson, 1988). Other important factors include competition, dispersal resources and existing seed banks (Parker and Leck, 1985; van der Valk and Davis, 1978).

In the context of wetland mitigation, additional factors can be attributed to the presence of particular species, community composition and species richness of the created wetland plant communities. These factors include the age of the created site (Hammer, 1992; Niering, 1990; Odum, 1988), methods of revegetation (Shisler, 1990; Brooks, 1990; Lowry, 1990, Jarman, et al., 1991), proximity of nearby wetlands (Confer, 1990; Jarman, et al., 1991), and the impact of wildlife (Hammer, 1992).

The establishment of diverse, functional plant communities is a crucial and challenging part of a wetland creation project. Because presence and diversity of vegetation cover in created wetlands are typically used as indicators of success, vegetation analyses comparing diversity in created marshes to that of natural marshes would seem appropriate.

STUDY SITES

All of the sites studied are inland freshwater wetland ecosystems located in southeastern New Hampshire. The vegetation consisted primarily of herbaceous macrophytes; however, some of the natural study sites showed signs of disturbance, with some vegetation indicative of past forested communities.

Created Wetlands

A created wetland in Salem, New Hampshire (Rockingham County) (hereafter referred to as GARAB), was constructed during the fall of 1989 to compensate for illegal fill activity performed at another area. The mitigation plant for this site allowed for the creation of approximately six acres of freshwater wetlands. The planting strategy for this wetland consisted only of some shrub plantings along the outer margins of the site. Some minor seeding was performed, but was reported as a failed effort (R. Prokop, pers. comm.). Natural revegetation was the primary strategy for establishing wetland vegetation at this site.

A created wetland in Portsmouth, New Hampshire (Rockingham County) was included in the study. The construction of this one-acre wetland (hereafter referred to as HSCHL) was made necessary by the expansion of a high school ball park and subsequent fill of existing forested wetlands. Construction was completed in late summer, 1989. The mitigation design allowed for the creation of wet meadow and shall marsh-type wetlands. The planting strategy employed in the HSCHL project consisted of direct planting and "seeding" of commercially-grown species.

There were three artificially-created wetland systems constructed within the confines of a shopping mall in Salem, New Hampshire (Rockingham County). Each of these wetlands represented an independent study site (hereafter referred to as MALL1, MALL2, and MALL3). Collectively, these wetlands compensated for six acres of wetland fill required for the construction of the mall. The mitigation design allowed for the wetlands to detain stormwater generated from the surrounding parking lots and enhance water quality (New England Development, 1989). Initial construction began through the winter of 1989 and 1990, and each basin was completed by early summer, 1990. There was an aggressive planting effort to vegetate the three mall wetlands rapidly in order to stabilize the sediments for functional purposes.

A created wetland in Portsmouth, New Hampshire (hereafter referred to as PORTS) was also studied. This 13-acre wetland was constructed during the winter of 1985-86, in order to compensate for the loss of wetlands due to the construction of a nearby hospital (Michener, et al., 1986). Wetland muck and soils were transplanted from the impacted wetlands into the created wetland basins. Minor seeding of Reed Canary Grass (Phalarisarundinacea) was performed in order to stabilize the soils. Some Cattail (Typha latifolia) colonies were also transplanted into the site during the early summer after construction.

Natural Wetlands

A natural wetland was studied in Dover, New Hampshire (Strafford County) associated with Indian Brook, a small tributary running southwest into the Cocheco River. The study site (hereafter referred to as DOVER) was surrounded by meadows and forest areas, and appeared undisturbed.

A second wetland was studied in Salem, New Hampshire (Rockingham County). This site (hereafter referred to as APART) is associated with Policy Brook, a small tributary of the Spicket River running through southern Salem. The study area is surrounded by forested areas; however, it appears to be disturbed on one side, as evidenced by the presence of dead standing trees in the wetland areas.

In Durham, New Hampshire (Strafford County) two sites were sampled. The first was a natural wetland (hereafter referred to as DAME1) located on the south side of Dame Road. This wetland is a large open water area with heavily-vegetated margins and is surrounded by forested areas. Because the open water area was somewhat shallow, portions were colonized by emergent aquatic plants. The presence of large numbers of dead trees indicates the site has experienced some disturbance.

A second natural wetland studied in Durham, New Hampshire (Strafford County) (hereafter referred to as DAME2), was located on the south side of Dame Road. The wetland consisted of shallow marsh and pools, with bordering forested areas.

One natural wetland was sampled in Portsmouth, New Hampshire (Rockingham County). This wetland (hereafter referred to as HOSPT) is located across from a regional hospital. The northern boundary of the site borders a roadway and a large parking lot, and the remainder of the area is surrounded by forested land. The wetland system appears to be heavily disturbed, as indicated by a large number of dead standing trees.

A natural wetland associated with a small stream was sampled in Rye, New Hampshire (Rockingham County). The study site (hereafter referred to as LJOHN) consisted of shallow areas parallel with a deeper central channel. Because a road bisects the stream, the channel starts from an underpass crossing the road and allows for the flow of slow-moving water. The area appeared to be somewhat disturbed, as there are a number of standing dead trees.

MATERIALS AND METHODS

Plant biodiversity data were obtained from six artificially-created wetlands and six natural wetlands between late July and early October, 1992. Within each wetland, two 0.01 hectare quadrats (modified as 5 meter and 20 meter plots) were inventoried to obtain an estimate of species richness. Because the analysis was concerned with the diversity of each wetland area, each quadrat was placed in a location that represented the most diverse vegetation and potentially maximum species richness. Quadrat placement was also arranged to best incorporate varying habitat types, therefore, avoiding the undersampling of species restricted to certain habitat types (i.e., open water or shores). This method was aimed at obtaining an estimate of relative species diversity for each wetland.

In addition to obtaining a total species inventory for each plot, plant biodiversity data were analyzed using the Sorensen index of community similarities (Mueller-Dombois and Ellenberg, 1974). The similarity of species composition between the created wetlands and the similarity of species composition between the natural wetlands were compared. Similarity indices between created and natural wetlands were also compared.

RESULTS AND DISCUSSION

Species Richness

Overall, the six created wetlands, while variable, are similar in species richness (Table 1). However, the number of species at the PORTS site is markedly greater than at the other created sites, with 49 species. The PORTS site is the oldest of the created sites studied, with its first growing season beginning in the spring of 1986. One factor contributing to the high diversity of this 7-year-old site is its age. Revegetation of the site involved both the importation of hydric soils, with a seed bank of many propagules, and natural revegetation. It is also adjacent to a forested wetland site. The full impact of a wildlife on the site is not known, but there is evidence of beaver greatly reducing the Cattail population in one area of the wetland (Garlo, 1993).

In contrast, the HSCHL site had the lowest diversity, with only 22 species. Completed in late summer of 1989, the HSCHL site had over three growing seasons at the time of sampling. This was an artificially-planted site. There has been some evidence of muskrat impact at the site (R. Prokop, pers. comm.), but the effects are unknown.

A site of approximately equal age to HSCHL site is the GARAB site. Sampling there yielded a total of 12 additional species. However, this site was allowed to be naturally colonized by wetland plants, instead of direct plantings. This wetland was constructed proximal to the Spicket River.

Table 1. Number of species sampled from created and natural wetlands.

CREATED		NATURAL	
SITE	SPECIES	SITE	SPECIES
MALL1	32	DOVER	29
MALL2	28	APART	28
MALL3	30	DAME1	35
GARAB	34	DAME2	49
HSCHL	22	HSOPT	28
PORTS	49	LJOHN	32
Mean	32.5		33.5

The youngest (three years old) of the created wetlands are the MALL sites (Table 1). Of the three sites, MALL1 had the highest number of species and MALL3 had the lowest. However, the differences in the number of species between the three sites are minor. The MALL sites were heavily planted, collectively with ca. 44,500 tubers and 255 pounds of seeds of herbaceous species. While this was an extensive and obviously expensive effort, the sampled species diversity only ranged by 28-32 among the sites. The MALL created wetlands are not near any natural wetland areas.

It is interesting that the two created wetland sites with the highest levels of diversity are PORTS (49 species) and GARAB (34 species), the only two sites that were not directly planted with wetland plants. The construction of the PORTS site did, however, employ transplanted hydric soils.

Compared to created sites, the number of species found in each of the natural wetlands does not vary much between sites (Table 1). However, the total of 49 species from the DAME2 plots in markedly greater than the total from the other natural sites. This site is the shallowest of the natural wetlands studied, with a mosaic of pools intermixed with emergent areas. This mosaic habitat most likely accounts for a greater diversity of wetland species. At the other natural wetlands, the majority of the areas consisted of deeper waters, with only limited emergent and/or shore areas. The APART and HOSPT had the lowest diversity of the natural sites, with a total of only 28 species in the plots sampled.

The mean number of species found among the created sites is 32.5 (Table 1), while the mean number of species found among the natural sites is greater (33.5). While few similar studies had been conducted, these data contrast with data from a study comparing created wetlands to natural wetlands in Connecticut. There species richness was found to be slightly greater at the created sites (Confer, 1990). However, the diversity means and ranges for both created and natural wetlands of that study are similar to those of the present investigation. Among the Connecticut wetlands, the mean species richness reported was 33 (range ca. 22-45 species) for the created wetlands and 30 (range ca. 20-39 species) for the natural wetlands.

The closeness of the mean diversity values for the created and natural wetland sites indicates that wetland creation practices and techniques used to construct wetland plant communities are adequate to establish plant diversities similar to those of natural wetlands. However, this may be true of species richness numbers only; it is not necessarily true with respect to community composition, an important concern of some mitigation projects.

There were 94 different species observed in the created sites as a whole, while within the natural sites a total of 100 species were observed (Table 2). The greater diversity was observed in the natural wetlands; the difference recorded, however, is minor.

Characteristic Species

In comparing the species composition of created wetlands to those regarded as natural, certain species were found to be characteristic of the created wetlands, yet this was not as common among the natural wetlands (Table 2). For instance, Alisma triviale, Carex scoparia, Eleocharis acicularis and E. obtusa were found in all six created wetlands, while their presence among the natural wetlands was disparate. Carex lurida is present in all but one created wetland. Juncus effusus was also found to be a common component of all the created wetland floras. Confer (1990) reports that J. effusus is common among created wetlands in Connecticut. Willard, et al., (1990) report that the colonization of Carex spp., Eleocharis spp., and Juncus spp can be expected in created wetlands. Because these genera frequently dominate natural wetlands, their dispersal into nearby created sites would be inevitable (Anderson and Brown, 1991).

Two plants regarded as weedy and aggressive, and common to all the created wetlands, are Typha latifolia and Lythrum salicaria. The presence of these "disturbance" species is not uncommon among wetland creation projects; however, the dominance of these two species, in particular, is not desirable (Odum, 1988; Shisler, 1990; Confer, 1990; Hammer, 1992).

It is important to reiterate that the presence of certain species can be due either to natural colonization or to artificial plantings, or both. Although the specific components of large-scale plantings employed in some sites are unclear, some of the above species common among the artificially-constructed wetlands are typically selected by wetland planners because of their high rate of successful establishment and because of their availability from wetland nurseries in large quantities. For example, both Typha latifolia and Juncus effusus were planted in all the MALL sites. Typha latifolia and "Carex spp." were planted at the HSCHL site.

The species occupying the natural wetlands are more scattered among the individual sites (Table 2). Three species (Lemna minor, Scirpus cyperinus, and Sparganium americanum) were found common to five of the six natural wetlands. Only Typha latifolia is common to all of the natural wetlands (as well as all of the study sites).

Table 2. Species composition within 0.01 hectare plots in each study site. Site abbreviations are as follows: 1=MALL1, 2=MALL2, 3=MALL3, 4=GARAB, 5=HSCHL, 6=PORTS, 7=DOVER, 8=APART, 9=DAME1, 10=DAME2, 11=HOSPT, and 12=LJOHN.

Taxa	Created						Natural					
	1	2	3	4	5	6	7	8	9	10	11	12
Acer rubrum	X			X		X			X		X	X
Acorus americanus	X		X									
Agalinis purpurea	X			X								
Agrostis stolonifera var. *stolonifera*			X									
Alisma triviale	X	X	X	X	X	X		X		X		
Alnus incana subsp. *rugosa*						X				X		X
Ambrosia artemisiifolia	X											
Asclepias incarnata subsp. *pulchra*						X					X	
Aster novi-belgii												X
A. praealtus										X		
A. lanceolatus var. *simplex*							X					
Betula lenta									X			
Bidens cernua					X					X		X
B. connata				X		X			X	X	X	X
B. discoidea									X	X		
B. vulgata			X									
Boehmeria cylindrica										X		
Brasenia schreberi		X							X			
Calamagrostis canadensis						X		X	X		X	X
Callitriche sp. (sterile)					X			X				
Carex comosa									X		X	X
C. lurida	X		X	X	X	X	X		X	X		
C. pseudocyperus												X
C. scoparia	X	X	X	X	X	X	X			X		
C. stricta						X		X		X	X	X
C. vulpinoidea						X						
Cephalanthus occidentalis							X	X				
Ceratophyllum echinatum									X	X		
Cicuta bulbifera						X			X	X	X	
Cinna arundinacea										X		
Clethra alnifolia												X
Cornus amomum subsp. *obliqua*										X		
Cuscuta gronovii										X		
Cyperus strigosus	X		X	X								
Dead trees								X	X		X	X
Decodon verticillata									X			
Drosera intermedia						X						
Dulichium arundinaceum									X			
Echinochloa crusgalli			X			X						
Eleocharis acicularis	X	X	X	X	X	X	X					
E. obtusa	X	X	X	X	X	X		X		X		
E. smallii		X				X	X	X				
Epilobium ciliatum subsp. *ciliatum*		X	X							X		X

Table 2 (Continued)

Taxa	Created						Natural					
	1	2	3	4	5	6	7	8	9	10	11	12
Equisetum arvense				X								
Eupatorium dubium			X									
E. perfoliatum							X					
Euthamia graminifolia							X					
Galium palustre						X						
G. tinctorium				X			X	X		X		X
G. trifidum	X		X		X						X	
Glyceria borealis									X			
G. canadense							X		X	X		
G. striata var. *striata*											X	
Grass (sterile)					X							
Hypericum boreale				X			X					
H. canadense	X			X		X						
H. mutilum			X									
Hypericum sp. (sterile)						X						
Ilex verticillata											X	
Impatiens capensis					X	X				X	X	X
Juncus brevicaudatus				X								
J. bufonius				X								
J. canadensis	X		X			X	X	X	X			
J. effusus	X	X	X	X	X	X	X		X	X		X
J. marginatus			X									
Leersia oryzoides	X	X	X			X			X	X		
Lemna minor	X	X			X	X		X	X	X	X	X
Lindernia dubia	X											
Ludwigia palustra	X			X	X	X		X		X		
Lycopus americanus										X		
L. uniflorus						X	X		X	X	X	
L. virginicus												X
Lysimachia terrestris						X	X		X			X
Lysimachia thyrsiflora											X	
Lythrum salicaria	X	X	X	X	X	X		X			X	
Mentha arvensis										X		
Mimulus ringens					X					X		
Myrica gale								X				
Myriophyllum humile				X								
Najas flexilis		X		X								
N. minor						X						
Nuphar variegata				X			X	X		X		
Nymphaea odorata		X			X							
Onoclea sensibilis	X		X	X		X				X	X	X
Oxalis sp. (sterile)				X								
Panicum rigidulum						X						
Penthorum sedoides										X		
Phalaris arundinacea	X					X			X	X		
Poa palustris				X		X	X	X		X		
Polygonum amphibium var. *emersum*						X			X			
P. arifolium										X		X
P. hydropiperoides var. *hydropiperoides*	X		X		X							
P. punctatum var. *confertiflorum*	X	X	X							X		
P. sagittatum			X	X						X		

Table 2 (Continued)

Taxa	Created						Natural					
	1	2	3	4	5	6	7	8	9	10	11	12
Pontederia cordata	X	X		X		X						
Potamogeton epihydrus		X					X	X		X		
P. foliosus	X											
P. natans						X						
P. pusillus												
var. *tenuissimus*		X			X	X	X	X			X	
Potamogeton spirillus				X								
Proserpinaca palustris						X		X				
Rhamnus frangula											X	
Sagittaria latifolia	X	X		X	X		X	X		X		X
Salix nigra						X		X				
S. sericea								X				
Scirpus cyperinus	X	X	X			X	X		X	X	X	X
S. fluviatilis		X										
S. hattorianus			X			X						
S. maritimus		X										
S. pungens	X	X	X	X								
S. tabernaemontanii	X	X	X		X		X	X			X	
Scutellaria galericulata										X		
S. latifolia												X
Sium suave				X		X	X					X
Solanum dulcamara											X	X
Solidago rugosa												
subsp. *rugosa*										X		X
Solidago sp. (sterile)											X	
Solidago uliginosa							X					
Sparganium sp. (sterile)					X							
S. americanum		X		X			X	X	X	X		X
S. eurycarpum	X	X				X				X		
Spiraea tomentosa	X					X	X	X	X			
S. latifolia							X			X	X	
Spirodela polyrhiza									X	X		
Thalictrum pubescens										X		
Thelypteris palustris						X	X			X	X	
Toxicodendron radicans												X
Triadenum fraseri						X						
T. virginicum									X	X		X
Typha angustifolia			X		X	X						
T. latifolia	X	X	X	X	X	X	X	X	X	X	X	X
Unknown seedling				X								
Utricularia gibba		X										
U. minor						X						
U. vulgaris		X					X		X	X		
Vaccinium corymbosum									X			
V. macrocarpon						X		X				
Verbena hastata			X									
Vicia tetrasperma												X
Wolffia borealis									X		X	X
W. brasiliensis									X			
W. columbiana									X		X	

Community Similarities

The similarities in species composition between the created wetland sites are presented in Table 3. Similarity indices range from 0-1.0, corresponding to a 0-100% similarity between any two plant communities. Interpretation of similarity indices is largely subject to the investigator (Mueller-Dombois and Ellenberg, 1974). Barbour, et al. (1987) states that as a "rule of thumb" any two plant communities with a similarity index >0.5 (i.e., >50% similarity) represent the same association. However, this appears to be a rather low cut-off for "similar" plant associations. Nevertheless, the overall similarities of the representative vegetation communities appear low among the created wetlands. The average index among the created sites is 0.44, with a range of 0.31-0.61, indicating that overall, the plant community of any one created site hardly resembles the plant community of any other created site.

The MALL1 and MALL3 sites had the greatest species similarity (0.61) of the created sites. Higher similarity indices between the three MALL sites would be expected because of their close proximity and similar plantings of the sites (the three were part of a single plan). The 0.57 similarity index between the MALL1 and MALL2 sites is the second highest index overall for the created wetlands. Of the MALL sites, MALL3 and MALL2 resemble each other the least. This can be attributed to the larger open water areas of MALL2, a feature that is limited in MALL3.

The least similar created wetlands are the PORTS and MALL2 sites. The PORTS site experienced natural revegetation, while the MALL2 site was directly planted.

The average similarity index among the natural wetlands is 0.34, indicating an even lower similarity between the natural sites than between the created sites (Table 4). Plant species compositions of natural wetlands are known to vary greatly from site to site. A vegetation study of 18 salt marshes along the coast of Maine found that the associated vegetation communities are site-specific in regard to species composition and richness (Jacobson and Jacobson, 1989). Only four species were found common to all 18 marshes.

The most similar sites among the natural wetlands are the HOSPT and LJOHN sites (Table 4). However, their respective vegetation communities share only 43% of the species between the two wetlands. Both sites are disturbed, deep marsha wetlands with limited emergent areas. The least similar sites among the natural wetlands are shared between DAME1 and APART, and between HOSPT and DOVER, both with similarity indices of 0.25. Very few species are common between these sites.

In comparing created and natural wetlands, the average similarity of species compositions between the two groups is 0.29 (Table 5). This number is lower than the average similarity index within the created wetlands (0.44) and within the natural wetlands (0.34), indicating that the species compositions between the created and natural sites resemble each other less than do the species compositions within wetland types (created and natural).

Table 3. Similarity in species composition between created wetlands.

SITE	MALL1	MALL2	MALL3	GARAB	HSCHL	PORTS
MALL1	1.0					
MALL2	.57	1.0				
MALL3	.61	.45	1.0			
GARAB	.52	.42	.38	1.0		
HSCHL	.52	.44	.46	.36	1.0	
PORTS	.49	.31	.38	.39	.34	1.0

Table 4. Similarity in species composition between natural wetlands.

SITE	DOVER	APART	DAME1	DAME2	HOSPT	LJOHN
DOVER	1.0					
APART	.42	1.0				
DAME1	.34	.25	1.0			
DAME2	.41	.31	.40	1.0		
HOSPT	.25	.29	.38	.40	1.0	
LJOHN	.26	.27	.39	.38	.43	1.0

With respect to floristic diversity, the PORTS created wetland and the natural DAME2 wetland are the most similar. Yet, these two sites have only 41% of the species in common. It might have been expected that these two sites would exhibit greater floristic similarity. These two sites had the highest species richness (49 species) among the created and the natural wetlands.

The HSCHL created wetland and the natural DAME1 wetland are the least similar of the wetlands.

Of the created wetlands, the PORTS is, on average, most similar to the natural wetlands. The mean similarity index of the PORTS side among the natural wetlands is 0.37, ranging from 0.30-0.41 (Table 5). This similarity is most likely the result of the site's maturity (seven growing seasons), utilizations of hydric soil with seed bank, and high species richness (49 species). The MALL3 created wetland is the least similar to the natural wetlands. The mean similarity index of the MALL3 site among the natural wetlands is 0.22, ranging from 0.16-0.30 (Table 5).

Of the natural wetlands, the DAME2 natural wetland is most similar floristically to the vegetation of the created wetlands. Similarity indices range from 0.30-0.41,

Table 5. Similarity in species composition between created and natural wetlands.

SITE	MALL1	MALL2	MALL3	GARAB	HSCHL	PORTS
DOVER	.30	.39	.27	.38	.31	.36
APART	.33	.36	.21	.32	.40	.39
DAME1	.30	.25	.18	.17	.14	.38
DAME2	.35	.36	.30	.36	.34	.41
HOSPT	.27	.21	.21	.16	.28	.36
LJOHN	.22	.23	.16	.27	.22	.30

with a mean of 0.35 (Table 5). The high species richness (49 species) and floristic diversity of DAME2 account for closer similarities. On the other hand, the DAME1 wetland least resembled the created wetlands, having a mean similarity index of 0.23 and a range of 0.14-0.38.

The results of the plant biodiversity study are comparable to the results from the relatively few other studies comparing created and natural wetlands. The vegetation component of a study of five pairs of created and natural wetlands in the Connecticut River Valley shows similar results in regard to average species richness and characteristic species (Confer, 1990). In a study analyzing the plant community composition of created wetlands in Massachusetts, six created wetlands were compared to six adjacent natural wetlands (Jarman, et al., 1991). Species richness did not vary significantly between the sites, however, the created wetlands were dominated by species different from the dominant species of the natural wetlands. On average, 52% of the species were common to both natural and created wetlands.

CONCERNS FOR WETLAND PLANT ESTABLISHMENT

Since the beginning of the "mitigation revolution", there has been some worry about introducing the non-native species (Hedgpeth, 1980; Padgett and Crow, 1994). Introduction of non-native species or "foreign stock" can involve common species, species rate to the region but more common elsewhere, or new introductions to a given flora. In the mitigation context, the phrase "introduced species" can no longer be restricted to new, deliberate additions to a particular flora. "Introduced species" must now be expanded to include species that possess a genetic composition differing from local, natural populations. For example, if plants or propagules of Sagittaria latifolia are acquired from a Michigan supplier and planted in a created wetland in New Hampshire, these plants should be considered "non-native" or "introduced", even though S. latifolia naturally occurs in New Hampshire. Many states have not faced this issue. On the other hand, Florida requires mitigation plant stock to come from within a 50-miles radius of the mitigation site.

Concerns for Ecological Compatibility

A major issue which needs to be addressed by those involved in approving mitigation plans is the introduction of foreign genetic material into an area where native populations exist. Through cross-pollination, there is the potential of altering the genetic make-up of native plant populations, which may influence behavior at the individual population or community level. The consequences of genetic "contamination" are unknown but may, in some cases, be very serious.

In some instances, the consequence of the introduction of (genetically) foreign plants could be that the introduced stock might die immediately, lacking the ability to cope with the environmental conditions present at the new site (Hammer, 1992, 1989). An underlying factor for this phenomenon may relate to the genetic composition of the transplanted species; it may not be ecologically compatible to a new site merely because of its genetics (Millar and Libby, 1989; Hammer, 1992). Another consequence may be a delayed death syndrome. Delayed vegetation failure may occur within non-native planted populations because they lack the genetic composition required to sustain longevity of the population over short and long-term events (droughts, extreme temperature fluctuations, etc.). Poor vigor among introduced populations may also occur. This may lead to the inability of that particular species to perform its needed function in the created ecosystem (e.g., produce seed to attract wildlife).

In contrast to failure, another ecological concern for introduced species is that of the potential competitive replacement of native flora. An introduced foreign species may competitively exclude less aggressive native species.

Ethical Concerns

The ethical concern surrounding introduced species must also be addressed. This concern evokes the question of whether or not introduced species, or even reintroduced species, should be considered in a mitigation plan. For example, in the MALL2 site, <u>Scirpus fluviatilis</u> represented a new addition to the flora of New Hampshire (Padgett and Crow, 1993). In wetland mitigation, the goal is to establish ecosystems which, as nearly as possible, simulate natural ecosystems which are therefore likely to interact with natural systems.

Another concern is that if a species is deliberately planted as part of a mitigation, could that species be legally recognized as rare, threatened, or endangered? Is it ethically correct to introduce such species (Falk and Olwell, 1992)? For instance, <u>Sparganium eurycarpum</u>, a wetland species protected by New Hampshire state law (Storks and Crow, 1978; DRED, 1987), was planted in great numbers in the MALL1 and MALL2 created wetlands.

ACKNOWLEDGEMENTS

We would like to thank A. Linn Bogle and Thomas D. Lee, University of New Hampshire, for their helpful comments on the manuscript. Albert Garlo, Normandeau Associates, Inc., kindly provided mitigation information on the PORTS site. Robert Prokop, Fugro-McClellan East, Inc., provided mitigation information on the MALL1, MALL2, MALL3, HSCHL, and GARAB sites. We are grateful to Kathleen M. McCauley for her assistance in the field.

This paper is Scientific Contribution Number 1822 from the New Hampshire Agricultural Experiment Station.

LITERATURE SITED

Anderson, R. C., and L. E. Brown, 1991. Establishment of a wetland plant community in a limestone quarry. Castanea 56(3): 168-175.

Barbour, M. G., J. H. Burk, and W. D. Pitts. 1987. Terrestrial plant ecology. 2nd ed. The Benjamin/Cummings Publishing Company, Inc., Reading, Maine.

Brooks, R. P. 1990. Wetland and waterbody restoration and creation associated with mining. Pp. 529-548. In: J. A. Kusler and M. E. Kentula (Editors). Wetland creation and restoration: the status of the science. Island Press. Washington, D. C.

Confer, S. R. 1990. Comparison of created and natural freshwater palustrine-emergent wetlands in Connecticut. M. A. thesis. Connecticut College. New London, Connecticut.

DRED, 1987. Protected plants of New Hampshire. Natural Heritage Inventory. Res-N 306 plant listing. Department of Resources and Economic Development. Concord, New Hampshire.

Falk, D. A. and P. Olwell. 1992. Scientific and policy considerations in restoration and reintroduction of endangered species. Rhodora 94: 287-315.

Garlo, A. S. 1993. Wetland creation/restoration in gravel pits in New Hampshire. In: F. J. Webb, Jr. (Editor). Proceedings of the 19th Annual Conference on Wetlands Restoration and Creation. May, 1992. Hillsborough Community College. Tampa, Florida.

Hammer, D. A. (Editor). 1989. Constructed wetlands for wastewater treatment; municipal, industrial, and agricultural. Lewis Publishers, Inc. Chelsea, Michigan.

Hammer, D. A. 1992. Creating freshwater wetlands. Lewis Publishers, Inc. Ann Arbor, Michigan.

Hedgpeth, J. W. 1980. The problem of introduced species in management and mitigation. Helgolander Wiss. Meeresuntersuch. 33:662-673.

Jacobson, H. A. and G. L. Jacobson. 1989. Variability of vegetation in tidal marshes of Maine, U.S.A. Canad. J. Bot. 67:230-238.

Jarman, N. M., R.A. Dobberteen, B. Windmiller, and P. R. Lelito. 1991. Evaluation of created freshwater wetlands in Massachusetts, Rest. and Manag. Notes 9(1):26-29.

Larson, J. S. 1988. Wetland creation and restoration: an outline of the scientific perspective. Pp. 73-79. In: J. Zelazny and J. S. Feierabend (Editors). Proceedings of a conference: increasing our wetland resources. National Wildlife Federation, Washington, D. C.

Lowry, D. J. 1990. Restoration and creation of palustrine wetlands associated with riverine systems of the glaciated northeast. Pp. 267-280. In: J. A. Kusler and M. E. Kentula (Editors). Wetland creation and restoration: the status of the science. Island Press. Washington, D. C.

Michener, M. C., N. Hansen, and J. Bridges. 1986. Final plan for mitigating the loss of wildlife habitat on the site of the proposed hospital construction in Portsmouth, New Hampshire, Hospital Corporation of America. Normandeau Associates, Inc., Bedford, New Hampshire.

Millar, C. I. and W. J. Libby. 1989. Disneyland or native ecosystem: genetics and the restorationist. Rest. and Manag. Notes 7(1):18-24.

Mueller-Dombois, D. and H. Ellenberg. 1974. Aims and methods of vegetation ecology. John Wiley and Sons. New York, New York.

Niering, W. A. 1988. Wetlands hydrology and vegetation dynamics. Pp. 320-322. In: J. A. Kusler, M. L. Quammen and G. Brooks (Editors). Proceedings of the national wetland symposium: mitigation of impacts and losses. October 8-10, 1986. New Orleans, Louisiana. Assoc. of State Wetland Managers Tech. Report No. 3 Berne, New York.

Niering, W. A. 1990. Vegetation dynamics in relation to wetland creation. Pp. 479-488. In: J. A. Kusler and M. E. Kentula (Editors). Wetland creation and restoration: the status of the science. Island Press. Washington, D. C.

Niering, W. A. 1989. Wetland vegetation development. Pp. 103-113. In: S. K. Majumdar, R. P. Brooks, F. J. Brenner, and R. W. Tiner (Editors). Wetlands ecology and conservation: emphasis in Pennsylvania. Pennsylvania Academy of Science.

New England Development. 1989. Impact assessment, Vol I. Natural resource protection and construction management: The Mall at Rockingham Park, Salem, New Hampshire. New England Development, Inc. Newton, Maine.

Odum, W. E. 1988. Predicting ecosystem development following creation and restoration of wetlands. Pp. 67-70. In: J. Zelazny and J. S. Feierabend (Editors). Proceedings of a conference: increasing our wetland resources. National Wildlife Federation, Washington, D. C.

Padgett, D. J. and G. E. Crow. 1993. Some unwelcome additions to the flora of New Hampshire. Rhodora 95:348-351.

Padgett, D. J. and G. E. Crow. 1994. Foreign plant stock: concerns for wetland mitigation. Restoration and Management Notes. In press.

Parker, V. T. and M. A. Leck. 1985. Relationships of seed banks to plant distribution patterns in a freshwater tidal wetland. Amer. J. Bot. 72(2):161-174.

Shisler, J. K. 1990. Creation and restoration of coastal wetlands of the northeastern United States. Pp. 143-170. In: J. A. Kusler and M. E. Kentula (Editors). Wetland creation and restoration: the status of the science. Island Press. Washington, D. C.

Storks, I. M. and G. E. Crow. 1978. Rare and endangered vascular plant species in New Hampshire. The New England Botanical Club in cooperation with the U. S. Fish and Wildlife Service. Newton Corner, Maine.

van der Valk, A. G. and C. B. Davis. 1978. The role of seed banks in the vegetation dynamics of prairie glacial marshes. Ecology 59(2):322-335.

Willard, D. E., V. M. Finn, D. A. Levine, and J. E. Klarquist. 1990. Creation and restoration of riparian wetlands in the agricultural midwest. Pp. 327-338. In: J. A. Kusler and M. E. Kentula (Editors). Wetland creation and restoration: the status of science. Island Press. Washington, D. C.

EVALUATION OF WETLAND PLANTINGS IN THREE STORMWATER RETENTION PONDS IN MARYLAND

Jeffrey Shenot and **Patrick Kangas**
Marine, Estuarine and Environmental Sciences Program
0218 Symons Hall
UNIVERSITY OF MARYLAND
College Park, Maryland 20742

ABSTRACT

Wetland plantings at three stormwater retention ponds in Montgomery County, Maryland were evaluated during the summer of 1992. Plantings were coordinated by staff of the Metropolitan Washington Council of Governments (WASHCOG) and involved eight obligate wetland species (Acorus calamus, Peltandra virginica, Pontederia cordata, Sagittaria latifolia, Saururus cernuus, Scirpus americanus, Scirpus validus, and Zizania acquatica). Stem densities of all emergent plant species found in the littoral zone of the retention ponds were sampled with quadrats. Densities of the eight planted species were compared with information on original densities established by WASHCOG. All but one species exhibited significant survival rates and several species expanded beyond their original planting locations. However, more than 100 other species volunteered at the sites and the original planted species were not numerically dominant in relation to the larger wetland plant community.

INTRODUCTION

Water quality improvement is one of the most important services a wetland can provide. Residential, industrial, agricultural and urban storm runoff wastewaters have been tested for treatment in wetlands with generally successful results over the last 20 years. Natural wetlands were first utilized for water quality improvement, but recently, artificial wetlands have been created for this and other purposes. The creation of wetlands is a major challenge involving both engineering design and ecological considerations (Mitsch and Cronk, 1992). In this paper, one aspect of wetland creation is evaluated: the success of a wetland planting program in existing stormwater retention ponds.

Wetlands have been found to improve water quality of urban storm runoff in a number of cases (Meiorin, 1989, Palmer and Hunt, 1989, Brown, 1985). Lakatos and McNemar (1988) cite six mechanisms involved in stormwater pollution removal in wetlands: sedimentation, absorption, filtration, biological assimilation, microbial decomposition, and chemical decomposition. The wetland plant community plays important roles in almost all of these mechanisms. Thus, the processes of choosing and planting the species that will make up the plant community are an important component of wetland creation for stormwater improvement. In the cases studied here, existing stormwater retention ponds were planted with species to create a wetland community that would improve their performance. The ponds were not specifically designed to be wetlands, but appropriate conditions exist along their margins for wetland plant community development.

The wetland planting program was coordinated by the Metropolitan Washington Council of Governments (WASHCOG). Staff from WASHCOG chose the species to be planted, and trained and supervised volunteers who did the actual plantings. Existing recommended species lists were followed for species selection (Livingston, 1989 and Athanas, 1987). The purpose of this study was to evaluate the retention pond plantings. Success of the plantings was determined by assessing the relative contribution of the planted species to the community that developed at the sties and by the degree of spreading or occurrence beyond stations originally planted. Successful species, therefore, would be those that make relatively large contribution to their community and/or those that have spread widely beyond where they were originally planted.

STUDY SITES

The study sties are stormwater management projects located in the Anacostia River watershed in Montgomery County, Maryland. Table 1 lists characteristics for each site. All of the sites are found in suburban settings, with mixtures of residential and low-density commercial developments.

At each site, the water level of the pond was drawn down for planting along the slope by groups of volunteer workers. The water level was returned after the plantings were completed. Plantings generally were done in the spring with distances of approximately 46 cm (18 inches) on center between plants and with two or more rows planted in plots, depending on the angle of the slope. Single species were planted in separate plots along the slope in all but one case where two species were planted together in a plot. Plantings were carried out primarily in 1989 and 1990.

Table 1. Characteristics of the stormwater management projects used as study sites. All sites are in Montgomery County, Maryland. Data are from the Metropolitan Washington Council of Governments.

	COUNTRYSIDE	WESTFARM	WHEATON BRANCH
Drainage Area (ha)	67	174	304
Average percent imperviousness of drainage area	12	50	55
Pond surface area (ha)	0.6	1.3	0.4

METHODS

Maps of the ponds which identified the locations of the plantings were obtained from WASHCOG. These maps showed the plots for each species planted at the sites, along with the number of individuals that were planted in each plot. Plots were located in the field with the maps and sampled with transects. All plants were counted in 0.25 square meter quadrats placed at three-meter intervals along a transect through the planting location. At least two transects were sampled in each plot and transect length varied, depending on the size of the plot. Further details of the sampling design are given in Shenot (1993).

Sampling was conducted during the 1992 growing season (May through September). In order to minimize disturbance to the plant community, only basal stem counts were performed. Thus, data are reported as stem densities. This measure is objective, but is not the best indicator of the contribution or importance of an individual plant to the community since all plants (small and large) are counted equally. Biomass would have been a better indicator, but one intent of the study was to not disturb the community with harvesting.

RESULTS

A total of eight emergent species were planted at the study sites, though not all species were planted at each site (Table 2). All of these species are classified as obligate wetland species by Reed (1988). It was originally hoped that numbers of individuals could be compared between the original plantings from the WASHCOG data and what was found during the present study. However, it was not feasible to count individual plants in the field since morphologies were variable and a large number of plants volunteered onto the sites. Thus, the data columns for a site in Table 2 are not directly comparable.

The contributions of the planted species to the plant communities that developed at the sites are given in Tables 3 through 5. In each of these tables, contributions are assessed relative to three different groupings: the total plant community found in all samples at a site; the total numbers of obligate and facultative wetland species, according to Reed (1988), found in all samples at a site; and the plant community found growing within the individual plots that were planted with one of the eight species. These groupings represent different community comparisons for the species. A planted species should make the highest contribution within the community at the planted stations (i.e., plots), since it was the only species planted in the plots and should make up 100 percent of the community. Its contribution should be reduced in the other comparisons because they include species from other plots.

In general, the planted species made only minor contributions to either the total community (Column 1 in Tables 3-5) or to the community of obligate and facultative wetland species (Column 2 in Tables 3-5). No species made up more than 6 percent of either of these community representations at any of the sites. Greater contributions are found within the planted stations (Column 3 in Tables 3-5). Three species stand out: Pontederia cordata made up 15 percent of the community where it was planted at Wheaton Branch and 23 percent at Countryside; Scirpus

Table 2. Comparison of number of individuals originally planted with number of individuals found in transect samples.

NAMES	WHEATON BRANCH		COUNTRYSIDE		WESTFARM	
	Planted Individuals	Sampled Stems	Planted Individuals	Sampled Stems	Planted Individuals	Sampled Stems
Acorus calamus	0	0	82	163	0	302
Peltandra virginica	101	1	0	0	0	6
Pontederia cordata	60	222	84	483	200	0
Sagittaria latifolia	300	521	130	128	250	80
Saururus cernuus	0	0	0	0	50	0
Scirpus americanus	130	174	152	786	225	0
Scirpus validus	125	954	230	758	225	530
Zizania aquatica	0	0	105	349	100	934

Table 3. Contribution of planted species to the plant community at Wheaton Branch.

NAME	Percent of Total Community	Percent of Wetland Species	Percent of Community at Planted Stations
Acorus calamus	Not Planted		
Peltandra virginica	< 1	< 1	< 1
Pontederia cordata	1	1	15
Sagittaria latifolia	3	3	9
Saururus cernuus	Not Planted		
Scirpus americanus	1	1	4
Scirpus validus	6	6	47
Zizania aquatica	Not Planted		

Table 4. Contribution of planted species to the plant community at Countryside.

NAME	Percent of Total Community	Percent of Wetland Species	Percent of Community at Planted Stations
Acorus calamus	1	1	6
Peltandra virginica	Not Planted		
Pontederia cordata	2	4	23
Sagittaria latifolia	1	1	4
Saururus cernuus	Not Planted		
Scirpus americanus	3	6	9
Scirpus validus	3	6	10
Zizania aquatica	1	3	6

Table 5. Contribution of planted species to the plant community at Westfarm.

NAME	Percent of Total Community	Percent of Wetland Species	Percent of Community at Planted Stations
Acorus calamus	1	2	Not Planted
Peltandra virginica	< 1	< 1	Not Planted
Pontederia cordata	0	0	0
Sagittaria latifolia	< 1	1	1
Saururus cernuus	0	0	0
Scirpus americanus	0	0	0
Scirpus validus	2	4	12
Zizania aquatica	4	6	96

validus made up 47 percent of the community where it was planted at Wheaton Branch, 10 percent at Countryside, and 12 percent at Westfarm; and Zizania aquatica made up 96 percent of the community where it was planted at Westfarm. However, except for Zizania aquatica at Westfarm, even these values are not large since, if a species were successful, it should make up 100 percent of the community in which it was the only planted species.

The degree of spreading of the planted species is shown in Table 6. These data are ratios of the number of plots in which a species was found in the present study over the number of plots in which the species was originally planted for each site. Totals are shown in the column to the right. Ratios greater than one suggest that the species has spread from its originally planted plots to other plots that were planted with other species. A successful species should have a ratio greater than one. A ratio less than one indicates that a species died out in some of the plots in which it was originally planted. Five of the eight species in Table 6 have ratios greater than one (Acorus calamus, Peltandra virginica, Sagittaria latifolia, Scirpus americanus, and Scirpus validus). However, Acorus calamus and Peltandra virginica at Countryside were found in the present study, but were not originally planted. Obviously, this indicates natural dispersal from outside the site, rather than spreading from within the site. Ratios greater than one, therefore, include spreading and/or natural dispersal. Even with this addition, though, these data indicate successful species and Acorus calamus and Sagittaria latifolia seem to have significant spreading ability. Saururus cernuus and Pontederia cordata have ratios less than one, indicating problems with their plantings.

DISCUSSION AND CONCLUSIONS

The overall conclusion of the study is that the planting program was not particularly successful in terms of the analyses described in this paper. None of the species that were planted made significant contributions to their total plant communities. Also, there were only a few cases in which planted species dominated the plots where they were planted or spread by some mechanism to other plots. However, many other species volunteered or dispersed naturally to the sites from outside (more than 100 across the three sites) and wetland communities did develop (Table 7). Between 26 and 53 obligate and facultative wetland species were found at the sites, including the planted species, and these species contributed more than 50 percent of the diversity (total number of species) at all of the sites, and more than 50 percent of the total stem density at two of the three sites (and nearly so at the other site: 46 percent at Wheaton Branch). Thus, through natural dispersal, wetlands did develop, apparently, with only small overall contributions from the planting program.

A number of factors may have contributed to the poor development of some of the planted species. Probably the most significant factor was competition with species that volunteered onto the sites. A large number of species appeared and, through competition, they probably impacted and planted species. Additionally, it was observed that some planted species were heavily grazed by water fowl which probably reduced their growth and vigor. Pollution, inadequate substrate, or extreme hydrology may also have contributed. Responses for species varied with site, so

Table 6. Spreading of planted species within the sites. Data are ratios of number of stations where the species was found, divided by the number of stations planted with the species (total stations/stations planted).

NAMES	COUNTRYSIDE	WESTFARM	WHEATON BRANCH	TOTAL
Acorus calamus	1/1	3/0	0/0	4/1
Peltandra virginica	0/0	2/0	1/2	3/2
Pontederia cordata	1/2	0/3	4/2	5/7
Sagittaria latifolia	6/2	8/3	9/3	23/8
Saururus cernuus	0/0	0/1	0/0	0/1
Scirpus americanus	6/3	0/4	6/3	12/10
Scirpus validus	3/2	5/4	4/3	12/9
Zizania aquatica	2/2	2/2	0/0	4/4

Table 7. Overview of plant communities, emphasizing the contribution of wetland species to the total communities. Sample sizes (1/4 m^2 quadrats) were as follows: Wheaton Branch n=135 (33.75 m^2), Countryside n=143 (35.75 m^2), and Westfarm n=168 (42 m^2).

PARAMETER	COUNTRYSIDE	WESTFARM	WHEATON BRANCH
Total number of plant species	85	62	42
Number of obligate and facultative wetland species	53	32	26
Percentage wetland species	62%	52%	62%
Total stem density	798/m^2	549/m^2	508/m^2
Obligate and facultative wetland stem density	369/m^2	351/m^2	461/m^2
Percentage wetland stem density	46%	64%	91%

site-specific studies of causal factors would be needed to understand the details of planting success.

The conclusions given above must be qualified because of the short time frame of the assessments (ponds were planted 2-3 years before this study). Given this qualification, the results indicate the planting program was only marginally successful and not a good economic investment. Although labor was basically free in the planting program, the plants cost between $500.00 and $1,000.00 for each site. These monies are important amounts since funding for conservation efforts, such as wetland planting programs, is limited. More analyses are needed to evaluate wetland planting programs and other aspects of wetland creation in order to make the best decisions with conservation funding.

Perhaps the best decision would be to manage the natural succession process that resulted in the diverse, wetland species-dominated communities summarized in Table 7. If certain species are desired, topographic diversity in the pond form (depressions and mounds) could encourage different species in the natural succession process. Also, the community could be allowed to develop for several years and then be cut back and/or treated with herbicides to give certain species competitive advantage. These kinds of management would require a monitoring program, but they should be economical and should achieve the objective of creation of a wetland plant community to help filter urban storm runoff waters.

ACKNOWLEDGEMENTS

John Galli of the Metropolitan Washington Council of Governments provided the maps and advice to initiate the study. Irwin Forseth and Adel Shirmohammadi provided help throughout the project at the University of Maryland.

LITERATURE CITED

Athanas, C. 1987. Guidelines for Constructing Wetland Stormwater Basins. Maryland Department of Natural Resources. Annapolis, Maryland.

Brown, R. G. 1985. Effects of an urban wetland on sediment and nutrient loads in runoff. Wetlands 4:147-158.

Lakatos, D. F. and L. J. McNemar. 1988. Wetlands and stormwater pollution management. Pp 214-223. In: Wetland Hydrology, Proceedings of the National Wetland Symposium. J. A. Kusler and G. Brooks (Editors). Association of State Wetland Managers. Berne, New York.

Livingston, E. H. 1989. Use of wetlands for urban stormwater management. Pp. 253-262. In: Constructed Wetlands for Wastewater Treatment. D. A. Hammer (Editor). Lewis Publishers. Chelsea, Michigan.

Meiorin, E. C. 1989. Urban runoff treatment in a fresh/brackish water marsh in Fremont, California. Pp. 677-685. In: Constructed Wetlands for Wastewater Treatment. D. A. Hammer (Editor). Lewis Publishers. Chelsea, Michigan.

Mitsch, W. J. and J. K. Cronk. 1992. Creation and restoration of wetlands: some design consideration for ecological engineering. Advances in Soil Science 17:217-259.

Palmer, C. N. and J. D. Hunt. 1989. Greenwood urban wetland, a man-made stormwater treatment facility. Pp. 205-214. In: Wetlands: Concerns and Successes. D. W. Fisk (Editor). American Water Resources Association. Bethesda, Maryland.

Reed, P. B. 1988. National List of Plant Species That Occur in Wetlands: 1988 National Summary. Biological Report 88(24), U.S. Fish and Wildlife Service. St. Petersburg, Florida.

Shenot, J. A. 1993. An Analysis of Wetland Planting Success at Three Stormwater Management Ponds in Montgomery County, Maryland. M. S. Thesis, University of Maryland. College Park, Maryland.

RESTORATION OF DEGRADED SALT MARSHES IN CONNECTICUT

Ron Rozsa
Department of Environmental Protection
LONG ISLAND SOUND PROGRAMS
79 Elm Street - P.O. Box 5066
Hartford, Connecticut 06102-5066
and
Richard A. Orson
NAJARIAN ASSOCIATES, INC.
1 Industrial Way West
Eatontown, New Jersey 07724

INTRODUCTION

During the last 300 years, Connecticut's tidal marshes have been subjected to a variety of culturally-induced modifications. This has led to the loss of an estimated 28 to 35 percent since the turn of the century, as a result of dredging and filling activities. Presently, the degradation of tidal wetlands results primarily from indirect impacts, such as stormwater runoff (e.g., dilution effects and sedimentation) and direct impacts from structures which reduce or eliminate tidal flows. Since hydrology is the single most important factor controlling wetland development, its modification has the greatest potential to impact wetlands. Some structures, such as tide gates and undersized culverts, have the greatest impact on wetland hydrology, while others such as bridges associated with transportation corridors, appear to have a minimal, if any, impact on tidal flows. Therefore, it is important to understand the type of modification and its potential impacts on tidal wetlands.

When the natural tidal cycle is interrupted and the marsh system is drained by structures such as tide gates, soil salinities are significantly reduced (Parrondo, et al., 1978) and water tables (Roman, et al., 1984) are lowered. The lower water tables and decreased soil salinities increase rate of decomposition (Roman, et al., 1984) and compaction (Dent, et al., 1976) causing a subsidence of the marsh peat and a corresponding decrease of surface elevations upstream of the restriction (Roman, et al., 1984). Further, the draining of peat has been shown to impact water quality through the creation of an acid-sulphate soil type. The oxidation of pyrite in relic salt marsh soils can lead to increased acid runoff and a subsequent mobilization of aluminum (Dent, 1986), and the oxidation of organic matter can create anoxic conditions in the surrounding water column (Portnoy, 1991). Therefore, restricting tidal flows and draining marsh areas can have a significant impact on the health and continued development of the coastal zone.

Reduced salinities and mesic soil moisture favor the replacement of salt marsh vegetation by brackish and upland plant species depending on the degree of tidal flow reduction. Roman, et al. (1984) noted that initial vegetation changes may vary geographically within the State of Connecticut. In Fairfield County, at the western end of the state, early colonizers to newly restricted marshes included species of

Solidago, Iva, and Erechtites with Phragmites australis becoming important over time. Along the central and eastern portions of the State, Phragmites was the colonizer in tidally restricted sites. Thus, a visual inspection of the vegetation provides a tool for qualitatively assessing the health of a marsh and discerning restoration needs.

Once Phragmites becomes established, it is a strong competitor. Phragmites grow rapidly, often attaining heights in excess of 1.5 m in one growing season (Haslam, 1973). This height shades other species to the point that they can no longer survive (Buttery and Lambert, 1965). Thus, it is common that Phragmites form monocultures at these sites. Indeed, Niering and Warren (1974) estimated that in 1970, 10% of Connecticut's tidal salt marshes were dominated by reedgrass.

As reed expands into former salt marsh, there is a concurrent decline in wildlife utilization and plant and animal diversity. Much of the decline can be related to a reduction in quality and variety of food sources (Larsson, 1976), reduced salinities (Daiber, 1963) and a decline in water quality (Dent, 1986; Portnoy, 1991). However, some of the decline can be attributed to the physical barriers created by monocultures of Phragmites which limit the migration of certain animal species onto the marsh (Alls, 1969). With the increased cover of a tall reed, predators are also able to gain greater access onto the marsh surface (Moller, 1975), and further limit breeding sites for waterfowl (Larsson, 1976).

Reedgrass also produces large amounts of detritus. Because flushing rates have already been reduced due to tidal restrictions, the culms of many of these reeds accumulate at culvert or tide gate entrances, further reducing flows. The reduction in flows and a subsequent decrease in flushing rates, cause a decline in water quality and an increase in mosquito breeding sites (Ferrigno, 1959; Tindall, 1961). Since the culms of Phragmites degrade very slowly, large amounts of combustible material accumulate quickly. Thus, an otherwise fire-free ecosystem is subjected to regular burns. The fires burn very fast and reduce the plant material to ashes (Steinke, 1974; Roman, et al., 1984). Thus, monocultures of Phragmites are less desirable than the salt marsh which existed previously.

With the spread of Phragmites and the concurrent decline in habitat and water quality, Connecticut has instituted a program of restoration for its coastal marshes. Although Connecticut has been involved in marsh restoration for the last 20 years, it is only within the last ten years that a coastwide restoration program has been in place. This paper presents some of the results of the restoration projects conducted in Connecticut. Two sites have been chosen to represent some of the more salient problems and dynamic solutions encountered thus far during the program.

REPRESENTATIVE STUDY SITES

Two representative sites were chosen to illustrate some of the problems associated with restoration. At both sites, different approaches to restoration had to be considered because of the specific conditions and limitations of each individual marsh and their surrounding communities. Data collected at each site includes soil salinities, vegetation analysis, a qualitative assessment of macroinvertebrate populations, land use history, and tidal circulation patterns. The restoration programs were cognizant of both costs and safety problems (i.e. flooding, fire).

Hammock River, Clinton

The Hammock River tidal marshes are located in Clinton, Connecticut. These marshes are microtidal (neap - 1.49 m; spring - 1.71 m) and drained by the Hammock River directly into Clinton Harbor. The system contains approximately 100 hectares of marsh area, of which about 70% is upstream of tidal gates (Figure 1).

During the late 1800's, an earthen road causeway was installed across the marsh surface (Beach Park Road). In 1913, this roadway was fitted with wooden flapper gates for mosquito control purposes. As beach communities grew, the need for additional roads increased and the marsh was further divided by the installation of Meadow Road., Causeway Road. and improvements to Route 1, each drained through a culvert opening. The combined effects of tide gates and road crossings reduce the tidal prism and lower the groundwater table (Areas II, III, IV). For instance, at high tide, water levels between the unrestricted marsh (Area I) and the restricted marsh (Area II) can vary as much as 0.6 m to 1.2 m.

Tidal circulation patterns in Hammock River have been significantly altered by the tidal restrictions. At its mouth, Hammock River is about 61 m (200 ft) wide. At Beach Park Road., the area of flow is reduced by over 68% of the natural creek width, as four 1.2 x 1.2 m (4x4 ft) tide gates control tidal exchange. The tide gates eliminated salt water flow into the system. A second opening through Beach Park Road. consists of a 0.6 m diameter metal culvert. This culvert feeds a smaller relic channel which historically drained into Hammock River, but today is connected directly to Clinton Harbor through a linear ditch. At the upper road crossings, cement box culverts allow for drainage from these areas. These box culverts are almost as wide as the natural creek opening and, due to increased flow velocities, the reductions in channel flow are probably only minimal, at most. The causeways do eliminate some surface sheet flow, but the extent of reductions and the impacts on marsh hydrology, if any, are unknown.

With reductions in tidal flow volumes and freshwater inflow from ground water and runoff during storm events, salinities in the restricted marshes (Area II, III, IV) are significantly lower than those reported in the unrestricted marsh (Area I)(Table 1). It would seem, therefore, that the tide gates have significantly reduced salinity values.

Subsidence has occurred upstream of Beach Park Road. in response to restricted tidal flow. Relative elevation surveys conducted by Roman, et al. (1984) noted that the surface of the restricted Hammock River marsh (Area II) was as much as 40 cm below the levels of the unrestricted marsh surface (Area I) located 10 m away.

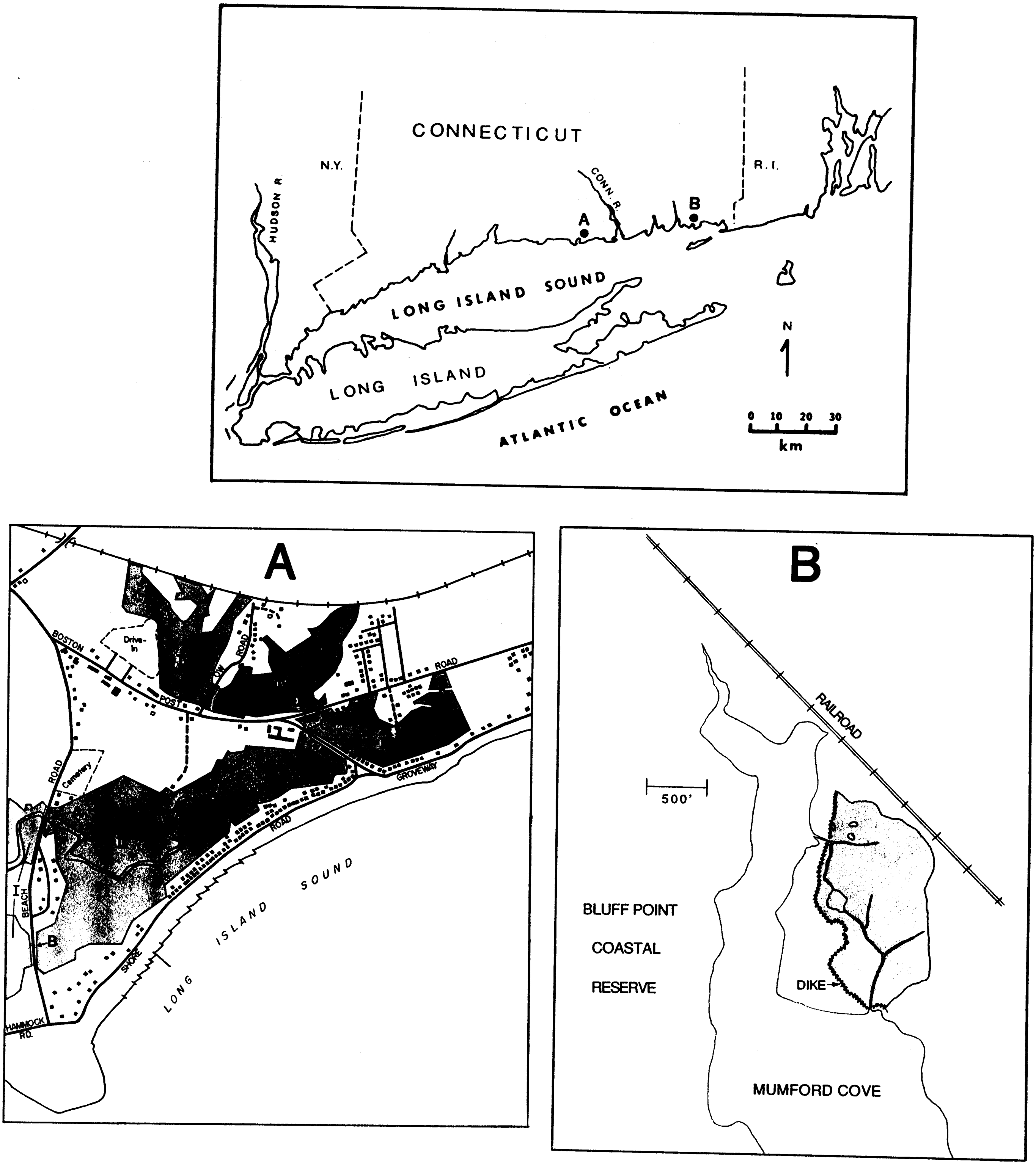


Figure 1. Site location map - Hammock River (A) and Mumford Cove (B).

Table 1. Results of salinity surveys for Hammock River.

LOCATION	WATER (ppt)		SOIL (ppt)	
	1982	1993	1982	1993
Area I	27	26	27	27
Area II	10	25	20	24
Area III	8	18	16	19
Area IV	4	8	12	15

The vegetation in the unrestricted marsh (Area I) is typical of New England type marshes (Niering and Warren, 1980). Spartina alterniflora dominates the low marsh and pannes, while Spartina patens is important in high marsh zones. In the restricted marsh areas, Phragmites australis (reedgrass) replaces salt marsh vegetation as the community dominant, forming dense monocultures in Areas III and IV and becoming a co-dominant in Area II. The ability of Spartina to persist in Area II was probably due to direct tidal exchange through the lower culvert across Beach Park Road.

The vegetation history, as determined through peat core analysis, reveals that reedgrass has existed in the system for many centuries as an upper-border species. It is only within the last 10 cm (ca. 50 years) that reed has become a low and high marsh community dominant above Beach Park Road.

With changes in plant community structure, there are corresponding changes in animal populations. In the unrestricted portions of the marsh, macroinvertebrates such as the mud fiddler crab (Uca pugnax), the ribbed mussel (Guekensia) and the salt marsh snail (Melampus) were abundant on the marsh surface while killifish (Fundulus) were abundant in the tide channels. Just above Beach Park Road., mud fiddler crabs and killifish decline to present while ribbed mussels and salt marsh snail fall to sparse to absent. In Areas III and IV, snails and mussels are absent, killifish are sparse and the less salt-tolerant red fiddler crab (Uca minax) replaces the mud fiddler crab, although its populations are sparse, as well. Based on the studies by Dent (1986) and Portnoy (1991) it is not known whether the reductions in aquatic organisms is due to changes in salinity, physical barriers to migration or degradation of water quality, specifically acidity, alumina toxicity or anoxia.

To restore the Hammock River salt marshes and control mosquito breeding, the Department of Environmental Protection and Mosquito Control Section of the State Department of Health Services implemented a tide gate management plan in 1985 to increase tidal flow. Recognizing that subsidence had occurred and a goal of maximizing the area occupied by emergent vegetation, only one of the four gates were opened. Within two years, it became evident that the increased tidal flushing was having a significant impact on the extent and density of reedgrass in all areas.

The initial success of a single gate opening in restoring the marsh at Hammock River was due to more regular flooding of the marsh surface. This would not have been possible without subsidence of the peat while tidal flows were restricted. This observation has been made at other sites and is important in restoration efforts. For example, at Sybil Creek in nearby Branford, a detailed study of the marsh surface found that by removing one tide gate and partially opening a second one, full restoration could be achieved in the marsh which now lies 40 to 60 cm below its unrestricted counterpart (Milone and MacBroom Engineers, Inc., 1987). At Hammock River, therefore, the tidal flows through a single gate may be sufficient to restore salt marsh habitat.

Since restoration began in 1985, Phragmites has been reduced in area and height. In Area II, approximately 10% of the reedgrass has been eliminated and replaced with Spartina spp. In Areas III and IV the height of the reedgrass has been reduced by as much as 50% and Spartina is expanding along the creek banks. With the opening of the gates, macroinvertebrate populations have increased as well. In Area II, the mud fiddler crabs, the ribbed mussel and marsh snail are present to abundant. In Area III, snails, mussels and the mud fiddler crab have moved back into the site. In Area IV, both fiddler crab species can now be found and mussels have begun to colonize the site. Killifish are now common in the creeks and channels throughout the system.

Mumford Cove, Groton Long Point

Mumford Cove is an embayment located between Groton Long Point and Bluff Point Coastal Reserve (Figure 1). The marsh in question is a 6-hectare site located on the eastern shore of Mumford Cove. The mean title range is 0.8 m.

During the 1950's, Mumford Cove marsh was used as a disposal site for sediments dredged from the cove. A low dike was built around the perimeter and the sediments were hydraulically pumped into the southern section of the marsh. The effluent structure, an adjustable weir, was located in the dike where the dike crossed the historic path of the tidal creek (Figure 1). The dredged sediments were deposited over a salt marsh dominated by high marsh vegetation with accumulations ranging from a high 1.2 m in the southern portion, to a low of 0.3 to 0.6 m in the northern areas.

In time, Phragmites colonized the site and formed a monoculture. At Mumford Cove, restoration could not be achieved through hydrologic manipulations because the marsh surface was positioned above spring-high water.

Rainwater would pond on the site and create an uncontrollable mosquito breeding problem. The Department of Environmental Protection and Mosquito Control Section developed a cooperative program to restore the site to salt marsh and, subsequently, implement open marsh water management (OMWM) techniques as necessary for mosquito control. It was envisioned that complete restoration would take about ten years. In the fall of 1988, the Mosquito Control Division began excavating the dredged sediment and disposing this material on the uplands. The following spring, the Fish and Wildlife Service became a cooperator through their Partners for Wildlife program. Target elevations were based upon the elevation of healthy low and high marsh adjacent to the degraded wetland. As the digging began, the textural differences between the relic peat and the sandy dredged sediments enabled the bulldozers to follow the tidal marsh's original grade, thus eliminating the need for constantly surveying elevations. Through the use of historic aerial photographs, tidal channels were located and restored to their original configuration. Much of the dredged sediment was disposed on uplands off site. However, some of the material was not able to be removed and this was used to construct upland islands on the marsh to increase habitat diversity. The removal of dredged sediment took four years to complete.

Within the first growing season, Juncus gerardi (blackgrass) recolonized the relic salt marsh areas and freshwater marsh species began growing along the upper borders. Rapid regeneration of marsh vegetation suggests that the seed bank may have remained viable while buried under the dredged sediments. Regeneration may have also been aided by a source of seeds and/or propagules available in Mumford Cove. It appears that restoration may proceed without the need for additional plantings.

DISCUSSION

As shown by the above examples, marsh restoration is a complex issue. It requires a combination of skills to achieve any measure of success. Site inventories are very important and without the proper baseline ecological data, it is difficult to judge the success or failure of a restoration project. Because of the complexity of the issue, restoration programs must be custom designed for each individual site. However, it is equally important that general guidelines be established for setting priorities and goals of the restoration program at the state or regional level.

In Connecticut, tidal marsh restoration begins with the premise that partial or complete tidal flushing will be reintroduced into a system that has been degraded by tidal restriction. Before this can be attempted, a knowledge of surface elevations and system hydrology is required. This is of particular importance when considering flood hazards of the surrounding communities. The reduction in surface elevations due to oxidation of the peat, is an important consideration in restoration plans. Indeed, past experience[1] has shown that, in systems such as Hammock River or Sybil Creek, if full tidal flushing is restored, the lowered surface elevations may submerge emergent halophytes beyond their flood tolerance and result in waterlogging of the peat and the creation of open water habitats. In contrast, just reintroducing full tidal flushing at Mumford Cover would not achieve success in restoration because the surface elevations are above mean high water. Therefore, careful and controlled flooding is important and must be considered for each individual site. Tidal flushing rate will have to be adjusted to existing elevations, rather than historic flow conditions. In marshes where subsidence has occurred, full

[1]During the 1950's, a hurricane removed tide gates from a system in Guilford, Connecticut. Since subsidence had already occurred at the site, the sudden reintroduction of full tidal flushing drowned the marsh existing at the time, and created an area now known as Lost Lake.

restoration may be accomplished with the reintroduction of only partial flooding. At these sites, the restricting structure will have to remain while the system readjusts to the flooding regime, a consideration that saves money and protects surrounding communities from flood damage.

Once the baseline data has been collected and the proper flooding regime has been established, tidal flushing can be reintroduced into the system. With the reintroduction of tides, there are corresponding increases in soil and water salinities and the duration and extent of flooding. Eventually, the height and vigor of the Phragmites is reduced and salt marsh species can begin to effectively compete to displace the reed. Displacement of reed first occurs along creek banks as increased flooding and salts allow for the lateral expansion of halophytes. Lateral expansion of salt marsh also occurs in pockets where high salinities have maintained salt-tolerant species through the restricted period. Eventually, reedgrass is displaced towards the upper border and halophytic garminoids, once again dominate the marsh plant community. On Connecticut marshes, full restoration takes about five to fifteen years to complete, depending on the amount of time that the system was restricted and the size and complexity of the wetland area.

It has been Connecticut's experience that salt marsh restoration will proceed spontaneously without the expense and failures associated with planting programs. This allows the plants and associated fauna time to adjust to restoration. Unfortunately, restoration projects are often constrained beyond the needs of the marsh (i.e., flooding of low-lying homes or roadways). When the proper flooding levels cannot be maintained to achieve restoration, then additional techniques may be required. Mechanically reducing the height of the reed (mowing) will hasten restoration. This must be done during the mid-summer and, if necessary, again in the early fall to deplete the reed of its growth reserves [Note: If the cutting is completed after the growing season, the plants will increase in vigor the following year due to reductions in light attenuation during the early spring.]. Chemical controls can also be used to reduce reed; however, the use of herbicides is only recommended if all other approaches cannot be employed.

Our experience in Connecticut has taught us that restoration must be given the proper amount of time to complete. This realization has enabled projects to continue with minimal expenditures, enabling more restoration projects to be started. Another important aspect of the restoration program has been to educate and include surrounding communities in the implementation of the plans, whenever possible. When this is pursued, opposition is limited and success rates are higher. Combining common sense with basic marsh ecology and allowing marsh restoration to proceed as naturally as possible, has led to a successful program of tidal marsh restoration in the State of Connecticut. Under the direction of the Long Island Sound Programs of the DEP, Connecticut has successfully restored over 400 hectare (1,000 acres) of tidal marsh habitat.

LITERATURE CITED

Alls, R. T., 1969. Killifish predation of mosquitos in low-level impounded Delaware salt marshes. Unpublished Masters Thesis, University of Delaware. 73 pp.

Buttery, B. R. and J. M. Lambert. 1965. Competition between Glyceria maxima and Phragmites communis in the region of Surlingham Broard: I: The competition mechanism. J. of Ecology 53:163-181.

Daiber, F. C. 1963. Tidal creeks and fish eggs. Est. Bull. 7:6-13.

Dent, D. L., E. J. Downing, and H. R. Rogaar. 1976. Changes in the structure of marsh soil following drainage and arable cultivation. J. Soil Sci. 27:250-265.

Dent, D. L. 1986. Acid sulphate soils: a baseline for research and development. Publication No. 39, International Institute for Land Reclamation and Improvement. The Netherlands.

Ferrigno, F. 1959. Further study on mosquito production on the newly acquired Caldwalder tract. Proc. N. J. Mosq. Ext. Assoc. 46:95-102.

Haslam, S. M. 1973. Some aspects of the life history and autecology of Phragmites communis: A review. Pol. Arch. Hydrobiol. 20:79-100.

Larsson, T. 1976. Competition and density of the bird fauna in Swedish shore meadows. Ornis Scand. 7:1-12.

Milone and MacBroom Engineering, Inc. 1987. Sybil Creek salt marsh study. Branford, Connecticut. Report to the State of Connecticut's DEP, Office of Planning and Coordination/Coastal Management. Hartford, Connecticut. 60 pp.

Moller, H. S. 1975. Danish salt marsh communities of breeding birds in relation to different types of management. Ornis Scand. 6:125-133.

Niering, W. A. and R. S. Warren. 1974. Tidal wetlands of Connecticut: Vegetation and associated animal populations. Volume I. Report to the State of Connecticut's DEP. Hartford, Connecticut. 46 pp.

Niering, W. A. and R. S. Warren. 1980. Vegetation patterns and processes in New England salt marshes. Bioscience 30:301-307.

Parrondo, R. T., J. G. Gosselink, and C. S. Hopkinson. 1978. Effects of salinity and drainage on the growth of three marsh grasses. Bot. Gaz. 139:102-107.

Portnoy, J. W. 1991. Summer oxygen depletion in a diked New England estuary. Estuaries 14:122-129.

Roman, C. T., W. A. Niering and R. S. Warren. 1984. Salt marsh vegetation change in response to tidal restoration. Environ. Manag. 8:141-150.

Steinke, T. 1974. A proposal concerning the restoration of the Pine Creek Estuary. Fairfield, Connecticut. Fairfield County Conservation Commission. Fairfield, Connecticut.

Tindall, E. E. 1961. A two-year study of mosquito breeding and wildlife usage in the Little Creek impounded salt marsh, Little Creek Wildlife Area, Delaware. 1959-1960. Proc. N. J. Mosq. Ext. Assoc. 48:100-105.

WETLAND CONSERVATION TECHNIQUES ON PRIVATE LANDS: LANDOWNER PERCEPTIONS OF STREAMLAND MANAGEMENT

Charles C. Schrader
Assistant Professor
Department of Landscape Architecture
KANSAS STATE UNIVERSITY
215 Seaton Hall
Manhattan, Kansas 66506

ABSTRACT

Woodland and floodplain wetlands along stream corridors play a critical role in protecting water quality in agricultural landscapes. Restoration of existing wetland areas and the creation of new systems will require the cooperation of private landowners. The study described in this paper examines agricultural landowners' perceptions of land management issues and program approaches which directly affect wetland protection and restoration. Issues studied include land-use control, environmental adequacy of present methods, and public concerns about water quality. Program approaches explored include conservation easements, public use/access, and landowner participation in program development. Results indicate that land-use control is extremely important to this group of landowners with tax relief, cooperative agreements and landowner participation in program development being the most preferred program approaches. Somewhat surprisingly, existing land-use methods are not thought to be highly effective for preserving water quality. Landowners also acknowledge public concerns about land-use decisions as an important issue in streamland management. Directions for further research suggest information-education programs designed and developed for both private landowners and the public are critical to successfully restoring and protecting wetlands within agricultural lands in the future.

INTRODUCTION

This research study focuses on stream corridor management issues, issues which are directly affected by individual actions and community attitudes and values. The role of the individual landowner must be studied in order to successfully implement wetland protection and restoration efforts in agricultural areas. The first step in promoting wetland protection and restoration practices is to identify landowners' attitudes and values.

Agricultural landowners' attitudes, which reflect environmental concern, land-use planning and management decisions, and conservation adoption behaviors, are examined in this study. Riparian land management issues and practices, which include restricting and changing landowner patterns of land use, are also studied. By focusing on resource management issues at the local level, we are more likely to identify a community's true environmental collective attitude (deHaven-Smith, 1988). One goal of this study is to identify private landowners' concerns and characteristics to aid in developing information/education programs which target riparian land protection and water quality enhancement. In order to facilitate the development of successful voluntary conservation and restoration programs, it is important to identify the types of alternative land management approaches that are preferred by landowners. These approaches are designed to encourage and promote wetland and streamland restoration and protection through voluntary, rather than mandated, adoption of changes in land-use.

Farmland protection programs have been adopted in all fifty states. Local governments have employed such techniques as zoning and transfer of development rights through alternative programs as a means to protect rural landscapes (Coen, Nassauer, and Tuttle, 1987). Generally, as the federal government becomes less involved in environmental regulation, state agencies have assumed an increased role in land protection (DeGrove, 1984). Cutbacks in federal spending and an increased emphasis on state and local management have transferred much of the conservation responsibility to local institutions. Public opinion and the present political climate ever increases the threat of regulatory actions to pressure farmers into changing streamland practices impacting water quality.

Riparian land conservation strategies are designed to protect land and water quality by restricting or changing land-use activities. The consequences of such changes impact the issues of economics, control of private property, the future needs of the community, and the ecological integrity of the environmental system. Strategies which take agricultural lands out of production directly effect: 1.) land and water quality as well as visual and biological character; 2.) individual landowners and farm operators; and, 3.) rural communities. The seriousness of local environmental problems and the prevailing political climate will dictate whether or not these methods will be voluntarily adopted or require enforcement.

The most successful local farmland protection programs characteristically are those in which the farming community participates throughout the process (Toner, 1978). Strategies which allow landowners to participate in planning and management at a local level may have a greater chance of long-term success. Landowners in this study respond to questions pertaining to their participation in conservation program planning, as well as their acceptance of alternative riparian land management strategies.

STUDY SITE

The study area for this research is in Dickinson County, Kansas, an area of predominantly agricultural land uses. Dickinson County is located in central Kansas, approximately 150 miles west of Kansas City, Kansas (Figure 1). Water quality assessment studies show streams, rivers, and lakes in this area are moderately to severely impacted by agricultural runoff. Excessive sediment loads and high levels of pesticides and fertilizers are present in the streams which drain the study watersheds (Kansas Department of Health and Environment, 1989). Primary threats to Kansas floodplain wetlands in this region are: 1) land conversion to agricultural cropland and livestock production; 2.) stream channel alterations; and 3.) irrigation activities which deplete ground and surface waters (Kansas Fish and Game

Commission, 1985). Historically, Kansas landowners have drained and developed streamlands for cropland production. This practice has resulted in loss of wetlands and increased non-point pollutants in surface water and groundwater.

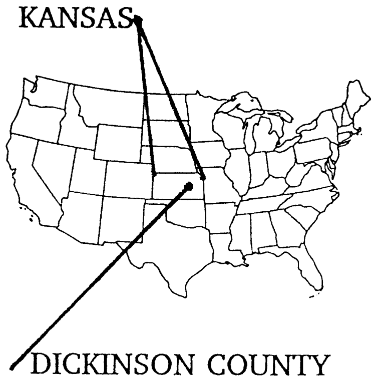


Figure 1. Study region and watershed location.

STUDY METHODS

A survey questionnaire was developed and distributed to a sample of landowners in Dickinson County, Kansas. The questionnaire was designed to collect information and function as an educational tool to introduce information which addressed conservation issues. Information dealing with the concepts of conservation easements and stream corridor management practices were included in the survey. An existing land ownership data base from the Soil Conservation Service in Dickinson County provided the names of owners and operators within the county. Surveys were sent to 500 randomly-selected farmland owners and operators in southern Dickinson County. The final return rate was 182/482 (38%). The majority of the respondents (77%) were farm operators with 23% being non-farm operators who owned farmland, but did not directly participate in farming operations. Over 90% of the participants reported a secondary or post-secondary education with 34% having some college experience. A diversity of off-farm income characteristics was reported with 48 participants (26%) having no off-farm income, and 117 (64%) having an off-farm income of less than 50 percent.

Participants responded to a variety of statements and questions based on a five-point Likert scale with a "1" on the scale indicating a low or negative assessment ("not at all") while a rating of "5" indicated the highest or most positive response ("very much"). In addition, a category of "don't know" was included to gather additional information on the research questions posited.

RESULTS AND DISCUSSION

Voluntary adoption of conservation practices requires landowners to balance private costs of time, labor, and capital investment with the expected benefits of economic return and resource protection. Only when landowners are willing and able to modify practices which currently degrade land and water quality, will it be possible to protect and restore vulnerable wetland and riparian resources. The importance of maintaining control over long-term use of their land is reflected in private landowners' attitudes toward land management approaches and practices. Control over the situations which produce these costs and benefits is vitally important to landowners who are directly impacted by the uncontrollable forces of nature and the free-market system. Increasingly, resource production activities such as farming, must also consider social and regulatory costs of public opinion and government regulation. The benefits of a wetland restoration project may directly affect community-wide environmental quality, as well as that of the individual property owner implementing such land improvements.

Land Management Philosophies

Because agricultural landowners are dependent on the land for their economic livelihood, private property rights can be expected to be extremely important to them. This is confirmed by the strong endorsement of the landowner's right to determine land-use (Table 1). Linked to the issue of land control is the perceived condition of the natural resource base which is impacted by each landowner. Existing land-use methods are not thought to be highly effective for preserving stream water quality (mean score of 3.39). Only 13% feel strongly that present methods are adequate. This would suggest that there are a substantial number of landowners who can see room for improvements in local land management as it affects water quality conditions.

An increasingly important issue in the management of agricultural land-use is the degree of control exercised by the public, be it urban dwellers or rural, non-farm operators. The right to control land-use is a controversial subject in rural areas where people are inherently suspicious of outside control. In this study, respondents are asked whether or not it is important to consider non-landowners' (public) concerns when making land-use decisions. Responses fall into the extremes at either end of the scale - extreme agreement (21%) and extreme disagreement (29%). It is interesting to note that 1 out of 5 Dickinson County landowners feel that non-landowner (public) concerns about water quality should be included in local land-use decisions.

Table 1. Agreement with land management philosophies.

Description	Mean	Std. Dev.	Don't Know	Agree Much	Not Agree
*The use of the land should be determined by the person who owns it.	4.08	1.12	00%	48%	04%
*Present methods of land use are adequate for preserving stream water quality in our community.	3.39	1.01	06%	13%	04%
*Land-use decisions which affect stream water quality should include the concerns of local non-landowners.	2.85	1.53	03%	21%	29%

Public concerns and demands are a driving force in efforts to protect and restore private wetlands and streamlands which impact public "quality of life". How much control, if any, should the general public have over the land-use methods practiced by private landowners, practices which directly impact community environmental quality? Who will pay for the conservation measures which must be taken by private landowners to protect water quality? Implementing wetland restoration and protection programs on private lands involves trade-offs in land-use control and financial assistance. Identifying compromise solutions which are feasible and acceptable to both public and private interests is a major issue in future environmental protection efforts.

The results of this study emphasize the important role landowner awareness of existing environmental conditions plays in influencing their attitudes toward land management. Alternative land management approaches are more preferred by landowners who feel that agricultural production is a serious cause of local water quality problems (Schrader, 1993). An important link between landowner characteristics, the perception of water pollution, and program approach acceptance, can be seen in Figure 2.

* High off-farm income * Age (over 59 yrs. old) * Non-feedlot operator * Farm operations border streamland * Small farm ownership/ high off-farm income	-->	Ag Production as Cause of Pollution	-->	Program Approaches

Figure 2. Factors influencing perceptions of program approaches.

It is important for landowners to recognize their influence on and the interconnections between land-use, environmental quality, and the community-wide quality of life. Results from the Dickinson County study indicate that there are a number of landowners who are not aware of the severity of water pollution in local watercourses. It has been shown that study participants who are younger and more dependent on farm income perceive agricultural pollution to be less of a threat to local water resources than do landowners who are older (over 59 years of age), or who have more off-farm income (Schrader, 1993). An increased awareness of environmental conditions by both private landowners and the general public is necessary to move in the direction of increased water quality protection. Information-education programs must provide landowners with information which increases their awareness of community problems, identifies possible solutions, and provides incentives for action (Osterman, 1988).

Participation Opportunities in Land Management Programs

Landowners in this study expressed a moderate interest in being given the opportunity to participate in planning future conservation programs. Nearly 1/3 of the landowners feel that their participation in conservation program planning is a very important component of any proposed water quality protection program (Table 2). The scores for the importance of future participation are substantially higher than scores for opportunities for present participation. Landowners indicate that they are seldom given the opportunity to participate in local land-use planning and decision making. These findings indicate an interest in becoming more involved in future conservation program planning. As cited earlier in this paper, the most successful farmland protection programs include the participation of the local farming community.

Table 2. Importance of landowner participation.

Description	Mean	Std. Dev.	Don't Know	Agree Much	Not Agree
*Programs that allow you to participate in developing guidelines for local conditions and situations.	3.49	1.38	07%	29%	11%
*There are ways available for me to voice my concerns and participate in local land-use decisions.	2.75	1.11	13%	07%	10%
*I would like to participate in planning and developing programs for solving local conservation issues.	2.72	1.40	12%	13%	22%

Alternative Program Approaches

Opinions varied widely about the use of cropland set-aside programs, programs which take land out of production in exchange for per acre payments. A precedent for this type of program has been set with the CRP which includes riparian restoration assistance and streamland conservation easements. The success of the CRP program has been largely due to the amount of the annual acreage payments

Table 3. Importance of land management approaches.

Description	Mean	Std. Dev.	Don't Know	Agree Much	Not Agree
ALTERNATIVE PROGRAM Approaches					
* Lower property, income, and estate taxes on your land (due to restrictions on development and certain land-uses).	3.64	1.46	08%	37%	13%
* A cooperative agreement that is non-binding and is reversible (1-5 year trial period).	3.12	1.47	12%	20%	18%
* Public assistance (labor) for stream and bank cleanup, livestock fencing, and tree planting along streambanks.	3.06	1.61	08%	25%	24%
* I would be willing to retire streambank areas from crop production in exchange for acreage payments.	2.81	1.59	21%	19%	24%
* Financial assistance for landowner in exchange for public recreational access to stream (canoeing, fishing, etc.) with no public access to adjoining land.	2.58	1.56	05%	19%	32%
* Land-use easements and managed by local Watershed District Board.	2.29	1.35	07%	08%	36%

which are given for more highly erodible and less productive lands. Taking streamlands and wetland areas out of production in order to restore natural ecosystems often involves taking the most productive land out of production, land which provides the greatest amount of income per acre. The uncertainty landowners feel about giving up control of these lands, even with acreage payments as an inducement, is seen in the results for alternative program approaches. A substantial number of landowners feel strongly about this issue. The results show that 19% are strongly for and 24% strongly against taking these lands out of production (Table 3). In addition, a number of individuals (21%) indicate an uncertainty about their feelings on this item.

Landowners in this study perceive tax relief to be the most important aspect of a conservation program. Strongest support is given to programs which lower taxes on agricultural lands in exchange for provisions restricting urban development and agricultural land use practices. Programs which contain property tax relief are considered beneficial, particularly to landowners who own and are taxed on larger tracts of land. Giving property tax benefits in exchange for modifying land-use is a strategy that has been included in conservation easement programs and should be examined as a possible means of encouraging wetland protection.

The program approaches dealing with cooperative agreements and public labor assistance for landowners are of slightly less importance to the respondents. Public assistance in the form of labor for projects involving streamland restoration is indicated to be of moderate importance. This issue brought out strong views on both ends of the scale, with 25% of the landowners in agreement and 24% in disagreement with the importance of public labor as a form of private assistance.

Public action and involvement as a means of assistance to landowners (stream and bank clean-up, livestock fencing, and tree planting) is a relatively new concept in the study area. This concept is included as a possible method for restoring and preserving stream and wetland sites. The notion of public/private cooperation and participation in the restoration process is one which deserves continued study.

Public recreational access to private lands is a controversial factor in land-use management. In this sample of Dickinson County landowners, financial assistance in exchange for public recreational access is not highly preferred (mean score of 2.58 and extreme disagreement of 32%). Receiving financial support in exchange for public access was not perceived to be worth the infringement on privacy and private property rights.

The surveyed landowners are not in favor of local watershed board management of conservation easements. This item is included in the survey in order to evaluate landowner attitudes toward local water resource management as a possible alternative. Dickinson County has a recent history of watershed lake construction. The land areas flooded by these lakes are managed as easements by a locally-elected watershed board. Local opposition to the development of these lakes (especially by landowners who have had farmland flooded by lake waters) has at times been vocal and strong. The low rating given for this question could be a reflection of the past problems associated with the planning, construction, and management of these watershed lakes.

CONCLUSIONS

Restoring original wetlands on agricultural lands will require land management changes which remove productive land from production. Unpredictable changes in economic, environmental, or social conditions can limit a landowner's ability to plan and implement changes in land-use management. Landowners in this study express strong interest in retaining control over resulting land management decisions, and indicate some concern over the effectiveness of present methods of land management for preserving water quality.

The results of this study indicate that the landowners sampled favor programs, such as conservation easements, which reduce property taxes. Property tax reduction is an issue of importance to Kansas property owners (Coulson, Peterson, and Murray, 1991). Future studies need to focus on the economic impact of land management alternatives; alternatives which may include using tax incentives as a means of encouraging voluntary adoption of conservation practices.

A wide diversity of individual ownership scales and economic livelihood is characteristically found in agricultural communities. Today, income from off-farm sources is a necessity for many agricultural landowners. In this study, off-farm income is a significant factor in landowner acceptance of alternative programs. Differences in landowner characteristics and situations can play a major role in how individuals perceive the importance of wetlands in protecting environmental quality.

Results indicate that landowners would like to participate in the planning of new conservation programs. Participation in the planning process would give landowners a greater sense of control over changes in land use and, as a result, their future economic situations. Landowners gain a sense of control over their future through control of personal property (Edney, 1976). Through active participation in planning and continued maintenance and management of land use, landowners feel they are "working the land" rather than being "paid to do nothing". In the past, government programs and policies which target farm operations have rarely included public forums or feedback from landowners. Researchers, private consultants, extension personnel, and farmers should all participate in examining problems and formulating solutions.

Public/community action as a means of assistance to landowners is an important area for further study. This can involve issues of public control such as recreational access. Public access to private lands is a controversial issue in riparian land management. In this study, the incentive of financial support in concert with public recreational access is not perceived by landowners to be worth infringement on their privacy and private property rights. Determining the trade-offs between public assistance, public benefits, and private property rights is an important step in promoting wetland restoration on private lands.

The local community needs to be involved in educational efforts which illustrate the relationship between individual action and community impact. The linear, dynamic and far-reaching nature of watersheds creates a system of environmental and cultural linkages which cross legal, political, and physical boundaries.

Encouraging private and public cooperation can increase community awareness of existing local problems and appropriate solutions.

Increased public concern toward environmental and social issues is placing more demand on private landowners to provide non-productive land-uses. In the past, most land-use changes have been a result of societal demands. Unfortunately, many of these demands have been made and land-use changes have been carried out without the knowledge required to minimize negative environmental impacts (Snyder, 1991). In order to achieve conservation goals and provide a wider range of "public" benefits, landowners must be informed about acceptable alternatives and practices. In order for landowners to implement changes necessary to satisfy societal demands, multi-disciplinary, technical assistance given through information-education programs will be needed. Future conservation programs must build upon established land management philosophies and practices in order to link "what is" to "what can be". Information should be disseminated in a circular process involving all participants rather than a vertical, hierarchical structure (Ness, 1989). The future success of wetlands projects on private lands depends on the interaction and sharing of information between public and private sectors and between landowners and the local community.

The process of information flow and transfer is not a one-way path, but instead requires two-way linkages between the parties involved, be it a land/person or person/person relationship. To quote Kentucky farmer and author Wendell Berry:

> "An adequate relationship between people and the land is not a monologue but a conservation. If the land has something to say back, the good farmer hears it."
> (Ehrenfeld, 1987)

LITERATURE CITED

Coen, D., J. I. Nassauer, and R. Tuttle. 1987. Landscape Architecture in the Rural Landscape. American Society of Landscape Architects. Washington, D.C.

Coulson, L. A., L. R. Peterson, and J. P. Murray. 1992. The 1992 Future of Kansas Survey. College of Human Ecology. Kansas State University. Manhattan, Kansas.

DeGrove, J. M. 1984. Land Growth & Politics. Planners Press, American Planning Association. Chicago, Illinois.

deHaven-Smith, L. 1988. "Environmental Belief Systems: Public Opinion on Land-use Regulation in Florida." Environment and Behavior. Vol. 20, No. 3:276-299.

Edney, J. J. 1976. "The Psychological Role of Property Rights in Human Behavior." Environment and Planning. Vol. 8:811-822.

Ehrenfeld, D. 1987. "Sustainable Agriculture and the Challenge of Place." American Journal of Alternative Agriculture. Vol. II, No. 4:184-187.

Kansas Department of Health and Environment. 1989. "Non-point Source Assessment Report."

Kansas Fish and Game Commission. 1985. "Riparian Resources and Their Relationships to Wildlife in Kansas." Background Paper, No. 19, Prepared for the Kansas Water Office.

Ness, R. "A Paradigm Shift for Extension: On-Farm Research in Sustainable Farming Practices." Land Stewardship Letter. Vol. 7, No. 4:12-14.

Osterman, D. A. 1988. The Missouri Flat Creek Watershed: The Natural and Social Systems, Erosion, and Water Quality Problems, Farmers' Perception of Highly Erodible Land, and Water Quality Management Plan. State of Washington, Department of Ecology, Referendum 39 Report. Pullman, WA.

Schrader, C. C. 1993. Streamland Management in Agricultural Landscapes: Landowner Perceptions of Conservation Approaches. Unpublished doctoral dissertation, University of Michigan. Ann Arbor, Michigan.

Snyder, R. M. 1991. Project Lakes - Design and Maintenance. In: F. J. Webb, Jr. (Editor), Proceedings of the Eighteenth Annual Conference on Wetlands Restoration and Creation. Hillsborough Community College. Tampa, Florida.

Toner, W. 1978. Saving Farms and Farmlands: A Community Planning Guide. Planning Advisory Service Report No. 333. American Planning Association, July.

CAS: A NEW SOFTWARE PACKAGE FOR ANALYZING SPECIES ABUNDANCE DATA

W. J. Streever and T. L. Crisman
Department of Environmental Engineering Sciences
UNIVERSITY OF FLORIDA
Gainesville, Florida 32611

ABSTRACT

Monitoring of the abundance and species composition of flora and fauna allows comparisons of constructed and natural wetlands. However, large species lists and extremely variable abundances complicate data management and comparative analyses. The Community Analysis System (CAS), a recently developed software package, simplifies these tasks. CAS facilitates data tabulation and report generation, multivariate analysis, and plot generation. Data tabulation and report generation allow creation of tables of raw data and accompanying statistics, such as diversity index values. Multivariate analysis capabilities allow classification (cluster analysis) and ordination. Plotting capabilities allow construction of dendrograms, three-dimensional ordination plots, and species area curves. In addition, recovery analysis, inversion of samples, and exclusion and pooling of taxonomic groups are readily performed. Although SAS and SPSS are capable of performing similar data manipulations and analyses, CAS offers an easily learned, user-friendly format that may allow for more effective interpretation of data sets assembled during mitigation monitoring.

INTRODUCTION

Sampling of restored and created wetlands frequently results in large matrices of samples and taxa. Community Analysis System (CAS) software provides a convenient method of processing taxa-abundance data to produce summary reports, diversity measurements, species-area plots, similarity matrices, classification (cluster) dendrograms, and ordination plots. The purpose of this paper is to describe the capabilities of CAS and to discuss various potential applications of CAS in the wetlands creation and restoration field. Major sections of the program's primary menu (Figure 1) will be discussed separately. It is important to note that not all of the functions available in CAS will be discussed.

TAXONOMIC CATALOG UTILITIES

Taxonomic catalog utilities allow creation of lists of taxa contained in samples and accompanying identification codes which will be associated with each taxon. Entries may be made at any taxonomic level (Figure 2), and subsequent analyses may be made at any of the taxonomic levels in the catalog. For example, an initial analysis may indicate that certain plant species are absent from a particular site, while a secondary analysis may show that entire families are missing. Taxonomic catalogs may be printed as a summary report.

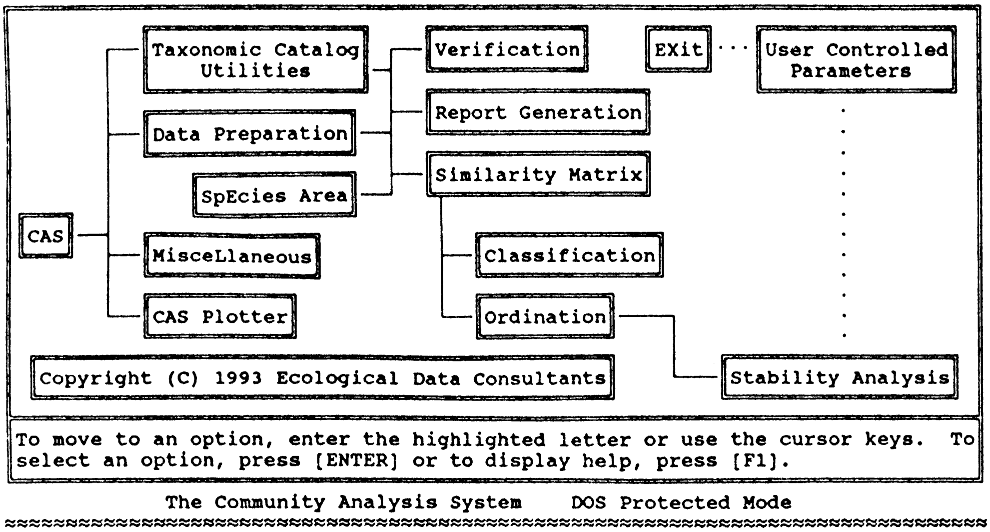


Figure 1. Primary CAS menu.

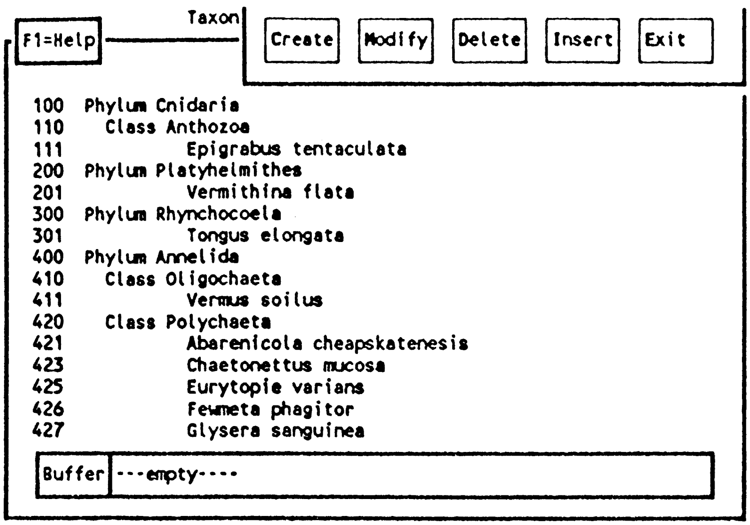


Figure 2. Taxonomic catalog.

DATA PREPARATION

Obviously, all data from a study must at some point be entered into the computer. By taking a "fill-in-the-blank" approach, the data preparation screens in CAS tend to minimize the chances of incorrectly entering data. The sample header screen (Figure 3) allows entry of data specific to each sample, while the data-entry screen (Figure 4) allows entry of taxon names and abundances. When the name of a taxon is keyed into the data entry screen, the identification code associated with that taxon automatically appears in the "ID" window. The abundance of the taxon is keyed into the "Value" window.

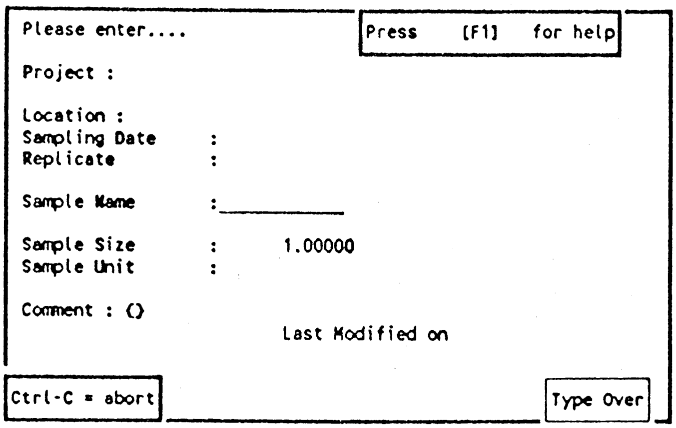


Figure 3. Sample header screen for data entry.

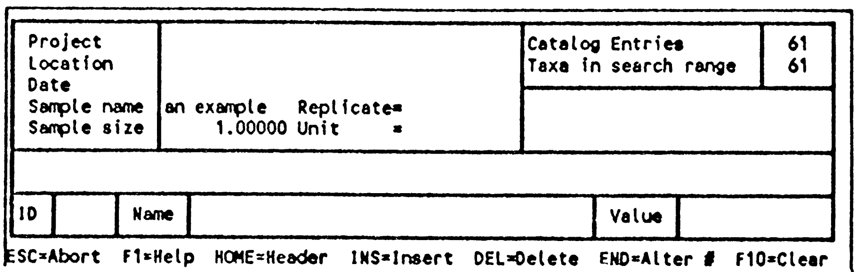


Figure 4. Sample entry screen for data entry.

VERIFICATION

Every data set should be checked for errors prior to analysis. In general, this involves checking data sheets against data entered into the computer. Although this can be accomplished by comparing data sheets to entries as they appear on the screen, it is generally more effective to use print-outs of information which appears on the screen. The verification menu allows generation of printed reports which can be easily compared to data sheets.

REPORT GENERATION

The report generation menu may be used to create summary reports (Figure 5) containing information on taxa richness, total number of specimens collected, specimen densities, diversity, and evenness. The reports also summarize the absolute and relative contributions of each taxa present in the data set. Reports may be generated for individual samples or for collections of samples. Thus, reports could be generated for each of the plant quadrats collected in a wetland and for the pooled plant quadrats of the wetland. In addition, a matrix can be created to allow quick comparisons of various summary statistics between samples (Figure 6), and a report of the mean and standard deviation for each taxa in a number of samples can be created (Figure 7). Reports generated by CAS are formatted to allow direct insertion into a document with little or no revision. Pielou (1977) and May (1975) provide reviews of the mathematics of diversity and evenness.

Page 1

Cheapskate Bay

Project	=Cheapskate Bay Faunal Survey (CAS Example Dataset)
Location	=Station # 1
Date	=July 23,1981
Name	=St#1 Jul81
Replicate	=1
Size	= 0.09375
Sample Unit	=m-2
Catalog	=Cheapskt.TXC
Last Modified	=Sep 21,1990

Sample Characteristics		
Number of Species	:	11
Density of All Species	:	1664.00000 /m-2
Total numbers of specimens counted	:	156.00000
Diversity (Hprime -log base e)	:	1.69375
Diversity (Hprime -log base 10)	:	0.73559
Evenness (Hprime/Hmax)	:	0.70635
Evenness (Scaled)	:	0.65006
Simpson's Index	:	0.75087

Taxa	Observed	Percentage
Phylum Annelida		
Class Oligochaeta		
Vermus soilus	533.33333	32.05128
Class Polychaeta		
Eurytopie varians	608.00000	36.53846
Fewmeta phagitor	32.00000	1.92308
Phylum Mollusca		
Class Bivalvia		
Nucula interferrous	53.33333	3.20513
Peeropig peeropig	10.66667	0.64103
Uno donu	74.66667	4.48718
Class Gastropoda		
Nastie vibovex	96.00000	5.76923
Turida predaceous	21.33333	1.28205
Phylum Arthropoda		
Class Crustacea		
Order Amphipoda		
Copralithia phagus	64.00000	3.84615
Gamma rush	10.66667	0.64103
Class Insecta		
Chironona extendor	160.00000	9.61538

Figure 5. Summary Report.

Cheapskate Bay		Page 1
Sample Characteristics	St#1 Jul81	St#2 Jul81
Number of Species :	11	14
Density of All Species :	1664.00000	1845.33333
Total numbers of specimens counted :	156.00000	173.00000
Diversity (Hprime -log base e) :	1.69375	1.55312
Diversity (Hprime -log base 10) :	0.73559	0.67451
Evenness (Hprime/Hmax) :	0.70635	0.58851
Evenness (Scaled) :	0.65006	0.50176
Simpson's Index :	0.75087	0.63611
Densities based on following units :	m-2	m-2

Figure 6. Data report for comparison between samples.

Page 1

Cheapskate Bay

Taxa	Mean	Std.Dev. (Divisor = N)	M / N
Phylum Platyhelmithes			
Vermithina flata	5.33333	6.15840	2/ 4
Phylum Rhynchocoela			
Tongus elongata	18.66667	13.42193	3/ 4
Phylum Annelida			
Class Oligochaeta			
Vermus soilus	738.66667	549.26429	4/ 4
Class Polychaeta			
Abarenicola cheapskatenesis	13.33333	10.21256	3/ 4
Eurytopie varians	152.00000	----	1/ 4
Fewmeta phagitor	8.00000	----	1/ 4
Pseudosludgia indicatus	45.33333	52.52724	2/ 4
Sabellaria podunka	40.00000	----	1/ 4
Phylum Mollusca			
Class Bivalvia			
Geoductus improbabilus	2.66667	----	1/ 4
Nucula interferrous	29.33333	28.05286	3/ 4
Peeropig peeropig	98.66667	103.92020	4/ 4
Tellina confusus	5.33333	----	1/ 4
Uno donu	40.00000	34.15000	4/ 4
Class Gastropoda			
Nastie vibovex	120.00000	30.63767	4/ 4
Polinices lunaformes	5.33333	6.15840	2/ 4
Turida predaceous	24.00000	5.33333	4/ 4
Phylum Arthropoda			
Class Crustacea			
Order Amphipoda			
Copralithia phagus	618.66667	651.91865	4/ 4
Gamma rush	16.00000	6.15840	4/ 4
Class Insecta			
Chiromona extendor	72.00000	84.15858	2/ 4
Phylum Echinodermata			
Class Holothuroidea			
Stychopus disgustus	32.00000	----	1/ 4
Phylum Hemichordata			
Ptycho daru	45.33333	52.52724	2/ 4

Figure 7. Summary reports giving means and standard deviations for each taxon.

SPECIES AREA

Curves comparing sampling effort to species richness indicate the level of sampling required to characterize a community qualitatively (Southwood, 1991). The concept may be extended to include quantitative information by replacing species richness with diversity. Further insight may be gained by considering only the evenness component of diversity. In each case, the shape of the curve will be affected by the order in which samples are drawn from the data. By chance, the first sample might contain an unusually large number of species, or it might contain only a few.

The species area menu in CAS readily computes species richness, diversity, and evenness curves. Samples are randomly drawn from the data set and richness, diversity, and evenness are computed. Problems with the effect of chance events on the shape of the curve are minimized by performing numerous iterations at each level of sampling effort, then reporting a mean and standard deviation. Data can be represented in a table or a figure (Figure 8).

SIMILARITY MATRIX

Similarity indices are used to compute a value which represents the degree to which two data sets are similar. In general, two identical data sets would be assigned a value of one, while two data sets with no species in common would be assigned a value of zero.

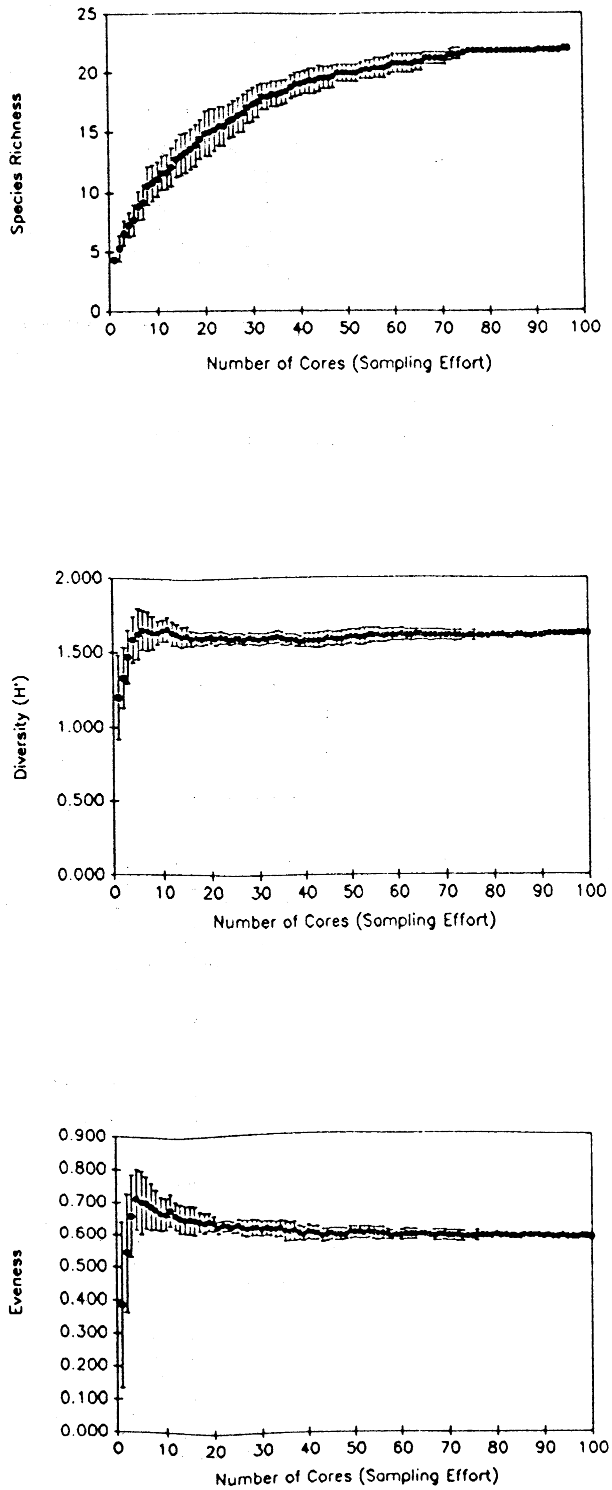


Figure 8. Curves for determining effect of increased sampling effort.

Because some similarity analyses are more effectively performed on transformed data, CAS provides various commonly-used transformation algorithms. CAS also allows choices between numerous qualitative (species presence-absence) and quantitative (species abundance) similarity indices (Figure 9). Output from the similarity analyses can be printed in a matrix (Figure 10) or saved for use in either classification or ordination. Several of the more commonly used similarity indices and the affect of index choice are reviewed by Bloom (1981).

CLASSIFICATION

The classification menu in CAS allows generation of dendrograms based upon similarities between samples (Figure 11). Samples which are most similar to one another in the dendrogram. Samples will be joined into clusters at some point along the similarity axis of the dendrogram. CAS allows the application of various clustering strategies, including nearest neighbor, furthest neighbor, group average, and flexibility. See Krzanowski (1990) or any comprehensive multivariate statistics text for further information regarding cluster analysis.

Proportional similarity	Coefficient of diVergence
Canberra metric	Gower's Angular separation
Gower's distánce measure	Rho
Morisita's index (original)	Czekanowski's Qualitative index
Morista's index (Horn's mod.)	Jaccard's index
Euclidean distance	Simple matching
Mean character Distance	

Press [F1] for help

Figure 9. Available similarity indices.

Page 1

Cheapskate Bay Stations 1..4

Transformation = None
Standardization = None
Index = Proportional Similarity

	Stat#1 Jul81	Stat#2 Jul81	Stat#3 Jul81
Stat#2 Jul81	0.16413		
Stat#3 Jul81	0.31891	0.66228	
Stat#4 Jul81	0.53061	0.16111	.0.56596

Figure 10. Matrix of similarity.

ORDINATION

Like classification, ordination is used to show relationships between samples. In ordination, each species in a set of samples is assigned an axis in multi-dimensional space, and each sample is plotted against the axes (Krzanowski, 1990). Although this is easily pictured for three species (a three-dimensional plot), visualization of plots containing greater than three species is generally impossible.

CAS uses principal coordinate analysis to reduce the number of axes from the maximum (which is equal to the total number of species in the sample) to three. In the three-dimensional representation, the first axis represents the line through multivariate space which contains the greatest variability; the second axis represents the line through multivariate space which contains the second greatest variability; and, the third axis represents the line through multivariate space which contains the third greatest variability. Variability explained by each line is reported by CAS, and a "lollipop diagram" representing the three dimensional figure is produced (Figure 12).

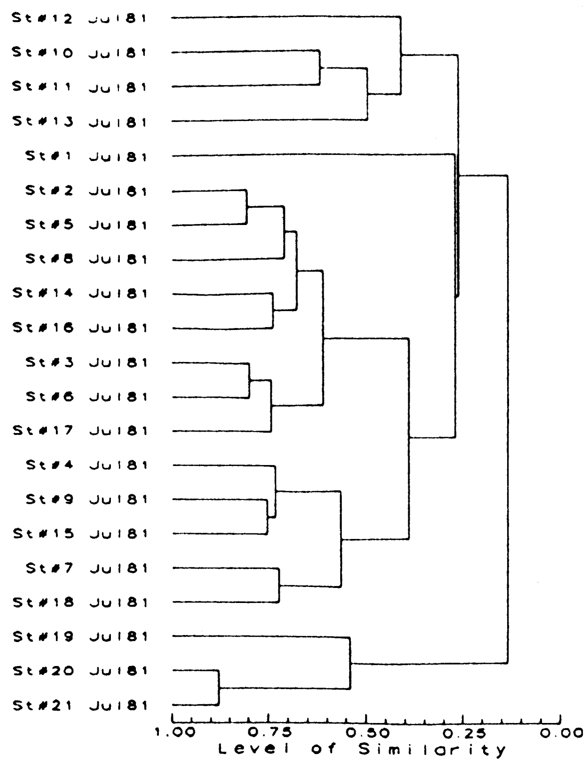


Figure 11. Cluster dendrogram

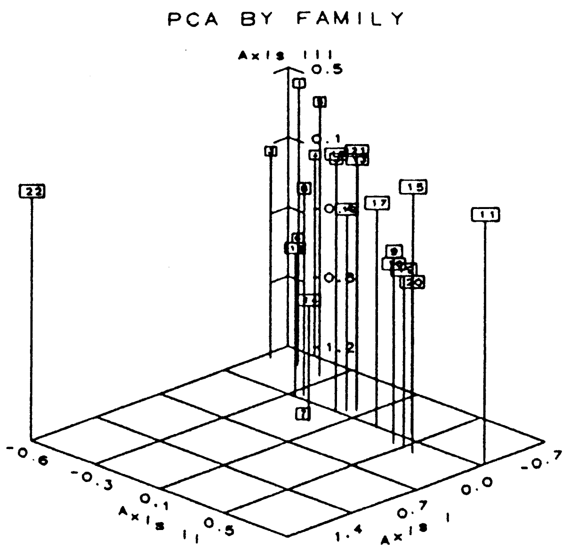


Figure 12. Principle coordinate Analysis plot.

STABILITY ANALYSIS

The stability analysis menu provides a method of tracking the development of a community through multivariate space. If the plant community in a restored marsh is sampled over a period of several years, one might expect it to return to a "natural" condition. If data is available to indicate what constitutes a "natural" condition, the approach of the restored marsh to the end point "natural" condition can be plotted. Bloom (1980) explains stability analysis.

MISCELLANEOUS

The Miscellaneous Menu provides various data manipulation operations that may be useful for some analyses. For example, if a researcher is interested in identifying species assemblages, samples and species could be inverted. In an inverted data matrix, the samples become attributes of the species, thus allowing operations such as generation of classification dendrograms which will cluster species assemblages. If a researcher is interested in the effect of removing some taxon, such as Typha from a data set, this is easily accomplished with the "Drop species from one or more samples." option. Likewise, a researcher might be interested in the effect of merging two taxa, such as Taxodium distichum and Taxodium ascendens.

One further option available on the Miscellaneous Menu, "Convert spreadsheet tables to CAS data-sets." will be of importance to users who store data in Lotus or Quattro' spreadsheets. This option creates crude taxonomic catalogs and enters all data from the spreadsheet into CAS.

CAS PLOTTER

The CAS Plotter Menu generates figures from programs run in CAS. Figures 11 and 12 were generated by the CAS plotter. In general, figures made on CAS are suitable for direct insertion into reports or publications.

SUMMARY

CAS is not a substitute for major statistical software packages such as SAS and SPSS. Instead, CAS provides a user-friendly environment in which to explore ecological data with techniques commonly employed by ecologists. CAS also provides an efficient means of generating tables and figures for use in reports and publications. Although this paper is not intended as an advertisement for CAS software, readers should be aware that it is available from Ecological Data Consultants, Inc., P.O. Box 760, Archer, Florida 32618. A low-cost "demo disk" is available for users who wish to explore the program prior to purchase.

ACKNOWLEDGEMENTS

We would like to thank S. A. Bloom and Brian Rood for their comments regarding this manuscript. This work was supported in part by a National Science Foundation Fellowship to W. J. Streever.

LITERATURE CITED

Bloom, S. A. 1980. Multivariate quantification of community recovery. In: The Recovery Process in Damaged Ecosystems. J. Cairns, Jr. (Editor). Ann Arbor Science. Ann Arbor, Michigan.

Bloom, S. A. 1981. Similarity indices in community studies: potential pitfalls. Marine Ecology Progress Series. 5:125-128.

Krzanowski, W. J. 1990. Principles of Multivariate Analysis. Oxford University Press. New York.

May, R. M. 1975. Patterns of species abundance and diversity. In: Ecology and Evolution of Communities. M. L. Cody & J. M. Diamond (Editors). Harvard University Press. Cambridge, Massachusetts.

Pielou, E. C. 1977. Mathematical Ecology. John Wiley and Sons. New York.

Southwood, T. R. E. 1991. Ecological Methods. Chapman and Hall. New York.

NOTES