Proceedings of the twenty-fifth Annual Conference on Ecosystems Restoration and Creation, May 1998. 1999

Plant City, Florida: Hillsborough Community College, 1999

https://digital.library.wisc.edu/1711.dl/DEFKS32CAJKZ68A

Proceedings of The 25th Annual Conference on Ecosystems Restoration and Creation

May, 1998

Sponsored by
Hillsborough Community College
Institute of Florida Studies

Hillsborough Community College

Proceedings of
The Twenty Fifth Annual Conference on Ecosystems Restoration and Creation

May, 1998

Sponsored by

Hillsborough Community College
Institute of Florida Studies

PATRICK J. CANNIZZARO
Editor

Hillsborough Community College
1206 North Park Road
Plant City, Florida 33566

This publication should be cited as:

Cannizzaro, P.J., Editor. 1999 Proceedings of the Twenty Fifth Annual Conference on Ecosystems Restoration and Creation. Hillsborough Community College, Tampa, Florida.

TABLE OF CONTENTS

INTRODUCTION

The Annual Conference on Ecosystems Restoration and Creation provides a forum for the exchange of results of scientific research in the restoration, creation, and management of freshwater and coastal systems. The conference is designed to be of particular benefit to governmental agencies, planning organizations, colleges and universities, corporations, and environmental groups. These proceedings are a compilation of papers and addresses presented at the Twenty Fifth 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 five years ago. We are grateful for his help and participation. Appreciation is also extended to Fred Webb and Charles Duesner for providing administrative support for the conference.

The following people also deserve acknowledgment for contributing to the conference and assisting in the preparation of the proceedings for publication: Elaine Baskin, Peter Rossi, Erica Moulton, Charles Mason and his staff. A very special thanks to Johnnie Hurst for her untiring assistance in handling the many details of conference planning.

Thanks are extended to the staff of **CF INDUSTRIES PLANT CITY PHOSPHATE COMPLEX** for arranging and conducting a very successful field trip to a wetland restoration site.

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

To all these people, thank you.

A COMPARISON OF TWO RESTORATION DESIGNS FOR DEGRADED NEW JERSEY SALT MARSHES

Lee L. Weishar, Aubrey Consulting, Inc. & Woods Hole Group
John Teal, Teal Limited & Woods Hole Oceanographic Institution
Ray Hinkle, Woodward Clyde Consultants
Kurt Philipp, Wetlands Research Associates, Inc.

ABSTRACT

Fifty years of diking in the Delaware River estuary have reduced marsh plain elevations, obliterated tidal channels, reduced *Spartina alterniflora* and increased *Phragmites australis* in marshes on the Delaware River Estuary. Two types of degraded wetlands were selected for restoration, active salt hay farms and *Phragmites* dominated areas that had been diked in the past but where the dikes had fallen into disrepair and tidal flows were restored to the marshes as the dikes were breached during storms. The restoration goal is to obtain a mixed salt marsh ecosystem dominated by *Spartina alterniflora* but with significant areas of *Spartina patens* high marsh. The initial physical data obtained on both types of marshes suggested that a single design philosophy could be adopted to restore the degraded salt marshes. This design called for excavating tidal channels to restore tidal inundation and increase marsh plain hydro-period. While this approach was successfully implemented for the engineering design, construction, and restoration of the three-diked salt hay farms, examination of tidal hydro-period and site topography showed that the design philosophy should be modified for the *Phragmites* dominated formerly diked areas. The restoration design philosophies for both types of sites and their application through adaptive management are discussed.

INTRODUCTION

This restoration project involves the restoration of 20,000 acres of salt marsh in Delaware and New Jersey (Fig. 1). The restoration of degraded salt marshes is not new or novel in the Delaware River Estuary (Weishar et al., 1996, 1997, 1998; Sebold, 1992). The mere return of tidal waters to the degraded salt marshes could be viewed as a successful restoration. However, establishment of natural hydro-periods and a natural marsh ecosystem consisting of upland buffer, high marsh, and low marsh plant communities are the general goals for this project. Diked salt marshes within the Delaware River estuary have been "naturally restored" both by nature or

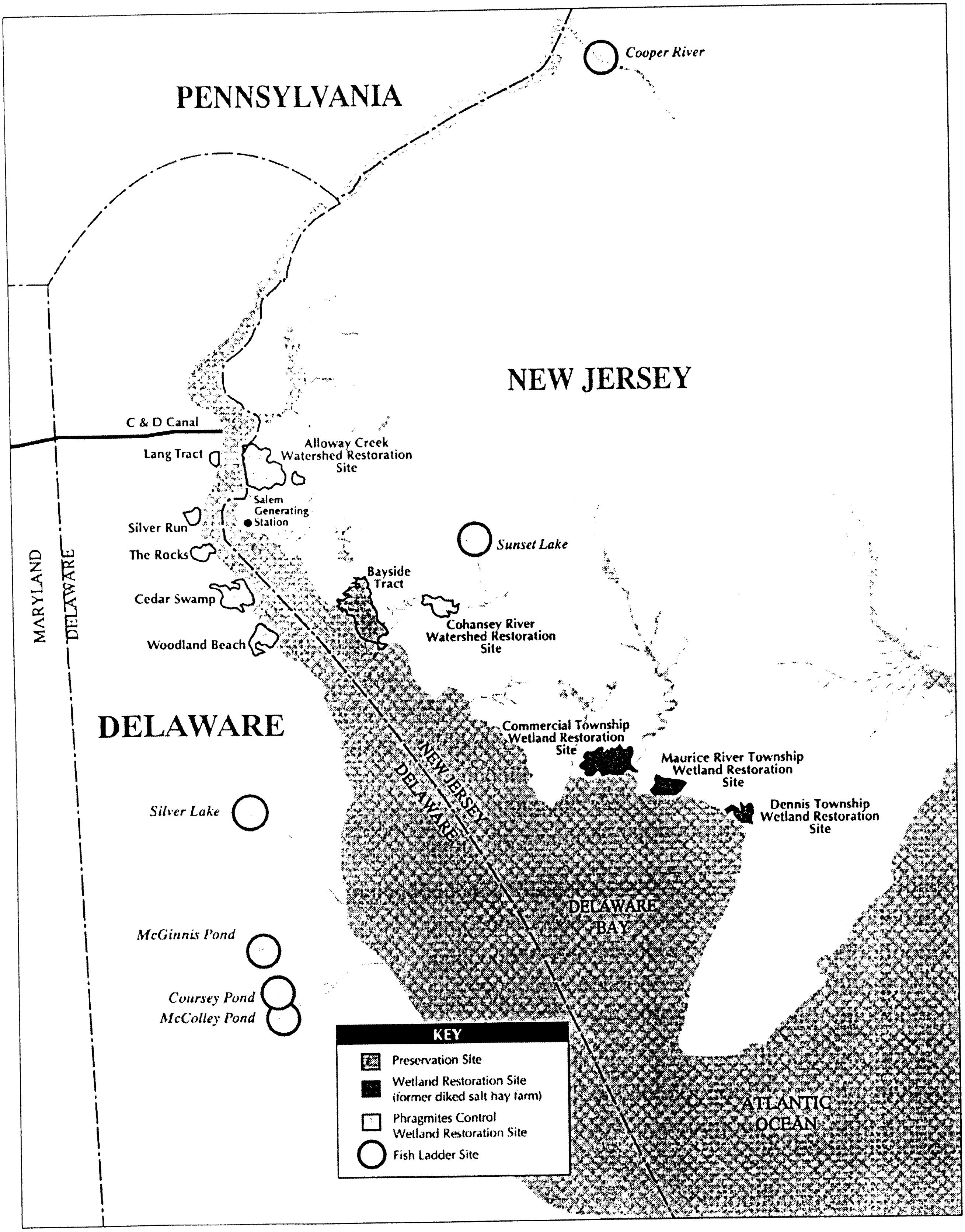


Figure 1: Location of salt hay and *Phragmites* dominated restoration sites in Delaware and New Jersey.

the New Jersey Department of Fish & Game. These naturally restored marshes exhibit widely different time courses on their way toward the desired goal of a fully functioning salt marsh. The reestablishment a natural marsh ecosystem with high biodiversity in a naturally restored marsh may take as long as 20 years. In some instances, marsh sediments were eroded to leave open water over large areas of the newly opened sites. While this does not indicate that these marshes are not on a successful restoration path, it does mean that the establishment of *Spartina alterniflora* will take decades. The restoration timeline is determined by the current rate of sediment deposition.

One might ask: "Why are salt marshes that have become a monoculture of *Phragmites australis* considered degraded?" In natural marshes within the Delaware River Estuary, *Phragmites* existed before about 1950 and had value as a plant that grew in a narrow band along the upland margins of salt marshes. It contributed to the diversity of habitats in the marsh by providing marsh "edge". The dense, tall stems provided nesting sites for birds, bedding areas for deer, and by virtue of its dense and long lasting stems also bound sediments at the upland edge of the marsh which slowed the delivery of sediments to the marsh from upland erosion.

Although *Phragmites* has a natural and useful place in salt marshes, when it becomes invasive and transforms the marsh from a diverse ecosystem into a *Phragmites* monoculture with low biodiversity the value of the marsh ecosystem is severely reduced. Another aspect of reduction of the marsh value by a *Phragmites* monoculture is blockage of the smallest marsh creeks. The rhizomes of *Phragmites* grow directly over and through the smallest, first order creeks at the edge of the marsh plain. This impedes water movement and over time, sediments accumulate to complete the creek blockage. As a result, small fish that feed on the marsh plain at high tide have their access to the marsh plain reduced. The result is a reduction in the ability of the marsh to support fisheries.

Phragmites has other effects on the physical structure of the marsh. The abundance of large rhizomes with their associated roots elevates the marsh plain. This reduces the time the marsh plain is flooded by the tides and the time that exchanges can take place between the marsh and the marsh creeks, and exchanges with the estuary, through the creeks. The rhizome mat also steepens the creek banks as a result of its resistance to erosion. The normal condition in a *Spartina* marsh is that the creek banks have a gentle slope except where they are undergoing severe erosion. These slopes offer a place for marsh animals such as fiddler crabs to feed at low tide. Additionally, a gentle slope provides an escape pathway for small fishes to escape predation from larger fish. These detritus-algal feeding fiddler crabs living on these gentle slopes serve as an important link between marsh and fish production since these animals are a favorite food for many fish. These gentle slopes are missing where the *Phragmites* mat produces an abrupt, steep bank, which occurs even in areas without much erosion.

Therefore, the goal of the restoration project is to restore the degraded salt marshes by reestablishing natural salt marsh communities with high biodiversity. This will be accomplished by restoring natural hydroperiods, reducing *Phragmites australis* coverage, and establish low marsh vegetation dominated by *Spartina alterniflora.* To accomplish these goals, an aggressive plan was adopted that used numerical modeling, engineering, and adaptive management to accelerate marsh evolution in such a manner that a natural mosaic of high and low marsh areas dominated by *Spartina* species would develop within 10 years.

PHRAGMITES CONTROL TECHNIQUES

Phragmites is found on every continent of the world except Antarctica and may have the widest distribution of any flowering plant (Tucker, 1990). Several researchers have noted the spread of *Phragmites* in New England marshes. Barret (1989) has documented replacement of healthy marshes with relatively high biodiversity with monotypic stands of *Phragmites* along the Connecticut River. Niering and Warren (1977) and Warren (1994) have observed that for the past 3,000 years *Phragmites* has been a relatively minor component of the marshes. However, it is now behaving as an aggressive invader outcompeting other native, brackish tidal marsh species.

Spread of *Phragmites* has been associated with human disturbance within the tidal marsh ecosystem. One such disturbance that has been reported to facilitate the spread of *Phragmites* is the reduction of hydroperiod, soil salinities, and/or lowering of the water table within the marsh (Roman et al., 1984). Recently, *Phragmites* has been found to be aggressively invading brackish marshes which do not show obvious signs of human disturbance (Barrett, 1989).

Phragmites has the capability of growing in most hydric soils excepting those characterized by high salinities (>15ppt), extreme nutrient deficiency, or rapidly moving water (Marks et al., 1994; Haslem, 1973). Therefore, controls of *Phragmites* have centered around manipulating the environment to either eradicate it or reduce its spread. A number of researchers have studied the effects of flooding and salinity on the growth of *Phragmites*. In tidally- restricted wetlands, Roman et al. (1984) found that rapid spread of *Phragmites* is associated with change in marsh plain elevation, debris deposited on the marsh surface, lowering of the water table, or decrease in soil salinity. Hellings and Gallagher (1992) performed experimental manipulations to determine the effects of salinity and flooding on *Phragmites* growth. They determined that high salinity water severely retarded *Phragmites* growth. Additionally, Squires and van der Valk (1992) found that *Phragmites* was not able to adjust its shoot length when covered with water exceeding 20 cm of depth.

Common methods of *Phragmites* control in areas of relatively low salinities (<10 ppt) include mechanical removal methods, burning without herbicide application, and herbicide application with burning. Cutting and/or mowing of *Phragmites* has had limited success in small areas generally less than one acre (Marks et al., 1994). Tiner (1995) found that mowing alone was not an effective control for *Phragmites* unless augmented with another form of control. Repeated mowing of *Phragmites* prevents the plant from sprouting and forces it to use food reserves stored in the rhizomes (Osterbrock, 1984). The goal of burning *Phragmites* like mowing, is to force the plant to deplete food reserves stored in the rhizomes. However, to be effective, burning should occur prior to nutrient translocation. This would dictate that *Phragmites* stands should be burned when the stems were still green. As a result, burning alone would be difficult to achieve and of limited value. Herbicide application and prescribed burning has been shown to be an effect means of controlling *Phragmites*. The state of Maryland (Ailstock, undated) determined that herbicide application and prescribed burning was the preferred control method for large stands of *Phragmites*. These techniques produced an 82% increase sunlight penetration to the marsh plain and the fire removed standing stems and the dense detrital layer.

It is perhaps ironic that in the eastern United States there is widespread concern for the aggressive spread of *Phragmites* while in Europe there is concern that the monotypic stands of *Phragmites* are declining.

Kubfn and Meizer (1997) found that eutrophication associated with high phosphorus and nitrogen in three lakes in southern Germany was associated with *Phragmites* decline. Young et al. (1991 and 1997) found that high nitrates resulted in reduced thickness of the outer ring of sclerenchyma and an increased portion of parenchyma. Tobler (1943) first established that *Phragmites* culms grown in calcareous soils with increased fertilizers were less woody, with less sclerenchyma, than those plants grown calcareous soils without fertilizer. These less woody plants are more susceptible to insect damage. They also delay translocation to and storage in their rhizomes and so become susceptible to early frosts (Kühl and Kohl, 1993).

The key to controlling *Phragmites* is to understand the plant physiology and adapt control measures that facilitates those natural processes, which inhibit plant growth. The project design presented in the following paragraphs uses the concepts presented above.

SITE DESCRIPTION

Three active salt hay farms within the state of New Jersey were selected for restoration. The salt hay farms were salt marshes that had been diked between 50 and 100 years ago, to cut off daily tidal flows. They were ditched for drainage to facilitate harvesting of salt hay. These impounded areas were particularly appealing for restoration because they were former salt marshes that had been altered by farmers to exclude natural tidal flow and were located in areas with salinity high enough to inhibit *Phragmites*.

The Dennis Township site is approximately 561 acres (225 ha) located in Cape May County New Jersey. The Maurice River Township site is approximately 1,100 acres (445 ha) located in Cumberland County. The Commercial Township site is located in Cumberland County and at 4,100 acres (1,660 ha) is the largest restoration site within the project. When these active salt hay farms were purchased in 1994, perimeter dikes were still intact at the Dennis and Commercial Township restoration sites. However, the bay front dikes had been breached at the Maurice River Township restoration site. At this site severe winter storms produced two significant bay front breaches which resulted in flooding of the marsh plain prior to restoration activities.

Seven *Phragmites* dominated sites (*Phragmites* sites) were selected for restoration, two of these in New Jersey and five in Delaware. These sites had been historically diked; however, the dikes had been allowed to deteriorate over the past decades. The combined deterioration of the dikes and the disruption of the site allowed thick monoculture stands of *Phragmites* to become established on these sites. The New Jersey sites include a 3,000 acre (1,215 ha) site within Elsinboro and Lower Alloways Creek Townships in Salem County and a 1,000 acre (405 ha) site located in Fairfield and Hopewell Townships within Cumberland County. The five *Phragmites* dominated sites in Delaware comprise up to an additional 6000 acres (2,425 ha) in New Castle County.

SALT HAY FARM RESTORATION DESIGN

The value salt hay farms as an estuarine resource had been greatly reduced or eliminated by dike construction that eliminated tidal flows. Dikes prevented the export of detritus to the estuary and denied large areas of nursery habitat to juvenile fish. The natural, sinuous tidal channels were filled to facilitate farming and were replaced by small drainage ditches designed to dry-out the marsh plain and remove rain water from the salt hay fields. These ditches were usually connected to the bay and tidal creeks by a pipe and a tide gate (often a flapper valve) which excluded bay water from the site.

Phragmites progressively replaced the salt hay grasses, *Spartina patens* and *Distichlis spicata,* as the salt content of the sediments was reduced by the elimination of daily tidal flows. Although soil compaction and oxidation had reduced the elevation of the marsh plain, it was still high enough to support *Spartina alterniflora.* Once we were familiar with the sites, a restoration design was formulated which incorporated the following steps:

- Characterize existing vegetative communities
- Obtain marsh plain elevations
- Design tidal channels and inlets to increase hydroperiods
- Construct tidal channels and tidal inlets
- Allow natural marsh processes to complete restoration (application of Ecological Engineering)
- Apply adaptive management to ensure restoration progress

Our restoration model was based upon Ecological Engineering. Ecological Engineering principles dictate that baseline engineering and construction be the minimum which would start the natural restoration process. After the reintroduction of daily tidal flows, natural marsh processes would complete the restoration of the marsh ecosystem. The restoration of daily tidal flows would restore the competitive edge to *Spartina* and *Distichlis* species characteristic of a healthy salt marsh system. The result is a self-sustaining marsh ecosystem, which requires no long-term intervention.

Restoring tidal hydraulics at the salt hay farms required constructing tidal inlets and excavating tidal channels. *Phragmites* control will be obtained as the estuarine waters with a salinity of 15-25 ppt inundate the marsh plain and increase pore water salinities. The restoration design called for the construction of the primary and secondary (4^{th} and 3^{rd} order) tidal channels. The tertiary and smaller (2^{nd} and 1^{st} order) channels would evolve naturally (application of Ecological Engineering) as the marsh system matures and evolves toward a new equilibrium condition. The 4th order tidal channels would be connected to the source waters of a tidal creek or the estuary through an unstructured tidal inlet. The tidal inlets and tidal channels will restore daily tidal flows, allow sediment deposition, and provides a pathway for the waterborne seeds of *Spartina alterniflora* and other desirable marsh species to enter the site. As a result of incorporating Ecological Engineering into the restoration design, it was anticipated that the following activities would occur naturally after initial construction was completed:

- 2nd and 1st order channels will form naturally.
- Channel velocities will average less than 2.0 ft/sec over a tidal cycle.
- 3rd and 4th order channels will evolve and migrate naturally.
- Sediment will accumulate on the marsh plain.
- Soil Salinities will increase (return to natural values).
- Seeds from *Spartina* species will be deposited with the sediment.
- Other non-*Phragmites* high and low marsh species will germinate and grow naturally at the proper elevations.
- *Phragmites* will be controlled through reintroduction of saline water.

The construction of the three salt hay farms has been completed and the marsh plain is beginning to accumulate sediment. The first restoration site was opened in the fall of 1997. The restoration site has approximately 64% coverage with *Spartina* species. The two other sites were completed late in 1997 and early in 1998. Preliminary revegetation data are not yet available although *Spartina alterniflora* is invading at all sites.

PHRAGMITES SITE DESIGN

As a result of the initial successes at the salt hay farms, the same model was used as a beginning hypothesis for the *Phragmites* sites restoration design. The *Phragmites* sites are located higher in the Delaware River Estuary than the active salt hay farms. As a result, salinities at the *Phragmites* sites range from 0 to 8 ppt in contrast to 15 to 25 ppt at the salt hay farms. The lower salinities in the *Phragmites* sites occur during the winter and early spring rains and higher salinities occur during the summer and times of low rainfall. The tidal range at the *Phragmites* sites range on average between 4 to 6 ft (1.2-1.8 m).

The restoration design for the *Phragmites* sites contained the following major components:

Characterize existing vegetative communities
Obtain marsh plain elevations after herbicide application and prescribed burns

Design tidal channels and inlets to increase hydroperiods
Construct tidal channels and tidal inlets
Allow natural marsh processes to complete restoration (application of Ecological Engineering)
Apply adaptive management to ensure restoration progress

Examination of aerial photographs and site visits showed the presence of only the larger channels corresponding to 4th and 3rd order channels at reference sites. Since the dense standing stalks of *Phragmites* monoculture does not fall down during the winter, details of the marsh plain could not be seen and site topography could not be obtained until the herbicide application and prescribed burning were completed. Examination of historical aerial photographs prior to completing the prescribed burns revealed remnant perimeter dikes constructed for farming and no small channels. We speculated that there were additional interior dikes remaining from farming activities that could not be seen on the aerial photos.

Combining our inability to see 1st and 2nd order channels, the relatively high marsh plain elevation within the *Phragmites* sites, and the suspected presence of remnant dikes, we anticipated there would be significant tidal lags and reduced hydroperiods within the interior of the *Phragmites* sites. Based on the initial reconnaissance and examination of the aerial photographs, the preliminary restoration design included dredging of additional tidal channels (Fig. 2).

After the prescribed burns were completed, site topography and new aerial photography was obtained which showed the presence of 2nd and 1st order channels throughout the *Phragmites* sites. The small rivulets along the 3rd and 2^{d} order channels in the *Spartina alterniflora* areas were absent in the *Phragmites* areas. More remnant dikes were visible in the new aerial imagery. An overlay of the aerial photography and the preliminary channel design showed that the designed and existing channel density was similar. The dense Phragmites cover had obscured the majority of the smaller channels (Fig. 3).

Additional field visits to the *Phragmites* sites confirmed the conclusions drawn from the aerial photographs. We found that flooding during high tide occurred uniformly across the entire observable marsh plain. Subsequent tidal elevation measurements confirmed that remnant levies or dikes did not cause tidal restrictions. The flat, featureless *Phragmites* marsh plain, initially observed near the creeks, extended far beyond the channel levies and was characteristic of large expanses of the marsh.

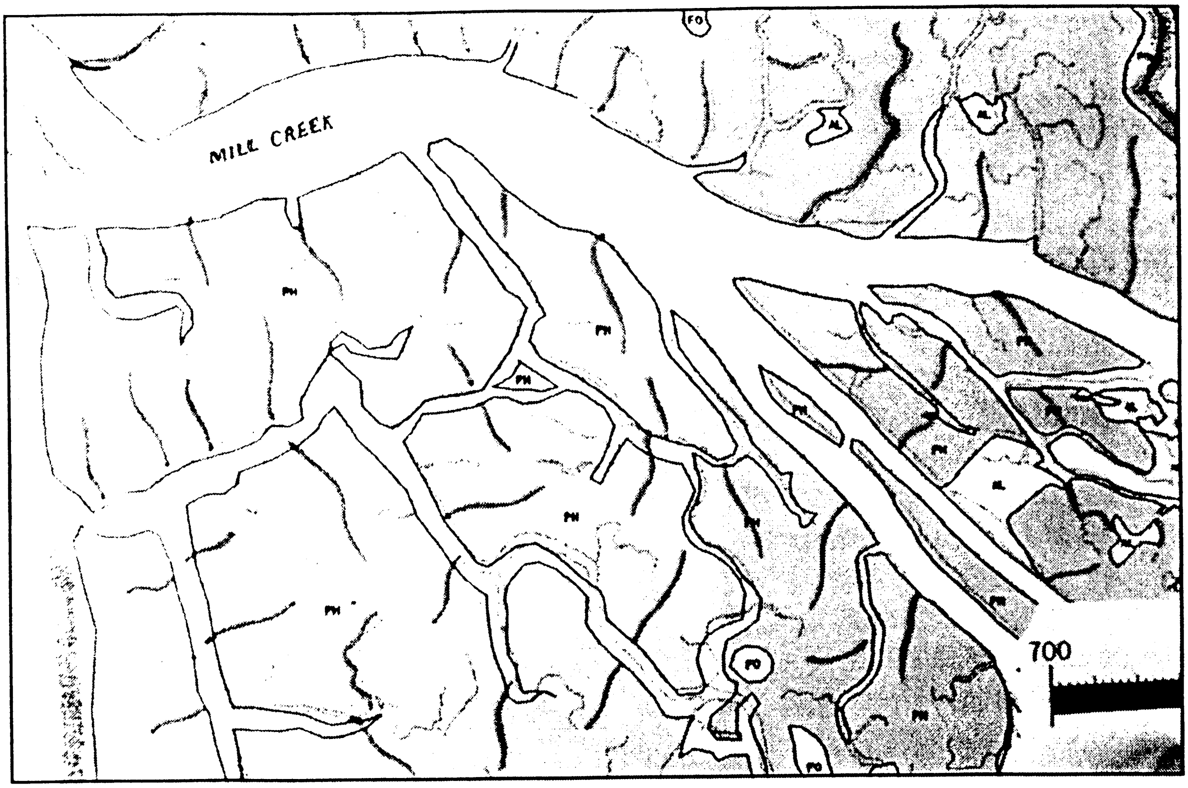


Figure 2: Preliminary channel design for *Phragmites* dominated restoration site prior to prescribed burn.

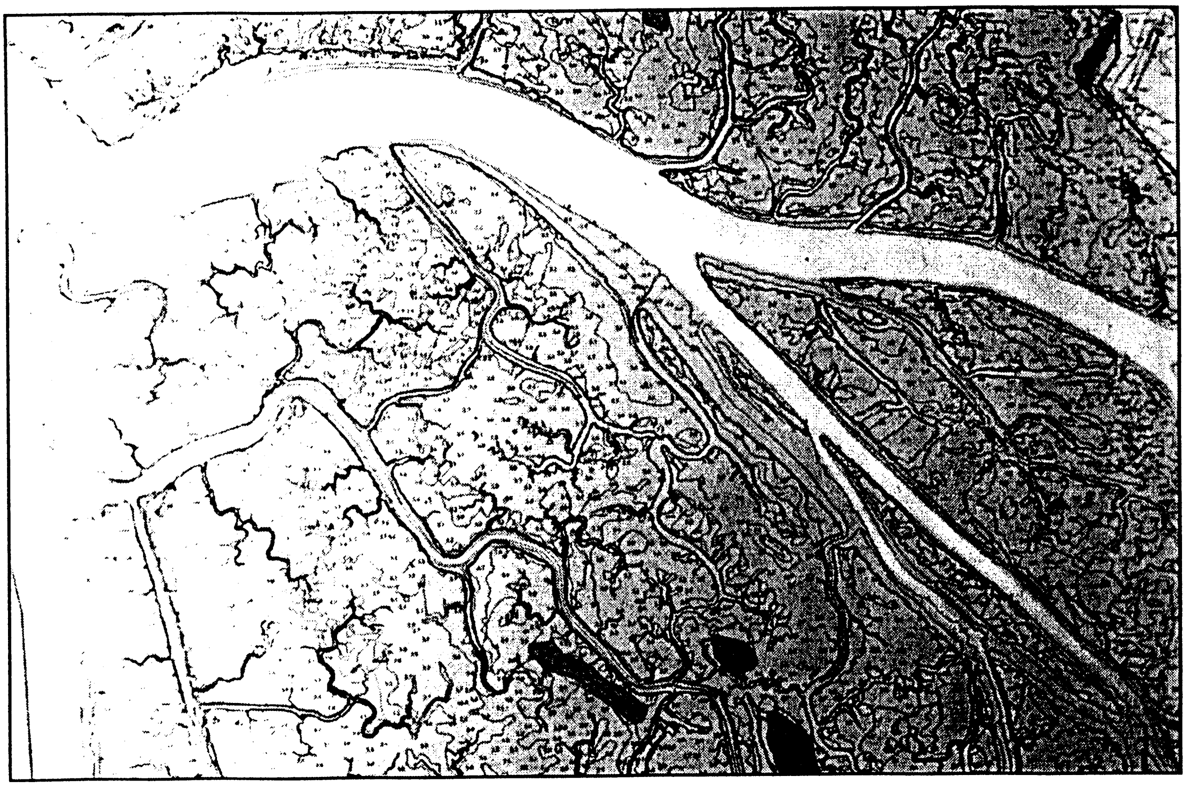


Figure 3: Channel and marsh drainage features identified after prescribed burn at *Phragmites* dominated restoration site.

REEVALUATION OF THE *PHRAGMITES* RESTORATION DESIGN

It was clear from the data collected that increasing the channel density would not improve marsh plain hydroperiods and reduce the competitiveness of *Phragmites*. The channels could easily be excavated, but since the marsh system hydroperiod is in equilibrium at present, these channels would only refill with sediment.

After the burn, on numerous traverses through the *Phragmites* areas and into adjacent areas of *Spartina alterniflora*, we observed a distinct difference in soil properties between the two adjacent sites. The sediments within the *Phragmites* areas were finer, better-drained, more compact, flatter, and lacked the microtopographic relief characteristic of adjacent *Spartina alterniflora* areas. The *Spartina alterniflora* areas adjacent to the *Phragmites* sites were characterized by poorly-drained soils, which were softer, had numerous small hummocks and small rivulet channels and puddles which retained water for most of the tidal cycle. While footing was firm throughout the *Phragmites* areas, it was only possible to traverse the *Spartina alterniflora* sites by stepping from hummock to hummock. If you missed a hummock, you sank to your knees into the soft sediments. Where *Spartina alterniflora* grew all the way to the creek banks, the banks sloped smoothly and gently to the water. Nearly vertical slopes characterized banks with *Phragmites* at the edges with *Phragmites* rhizomes exposed for 3 to 4 feet (1.0 to 1.2 m) beneath the marsh plain surface.

Our field observations suggested that one major difference between the *Phragmites* and the *Spartina alterniflora* dominated areas was the presence of microtopography and related soil characteristics. An examination of the literature suggested that this was not uncommon in other marsh and swamp areas. Reed and Cahoon (1992) reported small 12 cm variances in marsh plain elevation had a dramatic effect on the hydroperiod of the marsh in Louisiana. They also noted that the microtopography extended throughout the marsh and were not associated with stream side (channel) effects noted by Baumann et al. (1984); Hatton et al., (1983), and Sharma et al., (1987). While they were not able to identify the source or origin of the microtopography, they speculated that the small hummocks may be the result of animal tracks, or biogenic processes (Bertness, 1984). They noted that the microtopography resulted in longer hydroperiods and increase flooding frequency in these sections of the marsh. Additionally, they noted increased water saturation in the areas of microtopography similar to Mendelssohn and McKee (1987).

Based on our observations and the available literature, we modified the restoration design to increase the hydroperiod using microtopography and placed increased emphasis on controlling sources of *Phragmites* recolonization along remnant dikes and other manmade high areas. Introducing microtopography will breakup the flat, consolidated, well-drained tabletop topography produced by *Phragmites* by creating the microtopography characteristic of adjacent *Spartina alterniflora* areas. This breakup can be achieved by disturbing the marsh sediments with narrow, 12 to 18 in. long, "V" shaped scribes making grooves about two to three feet apart and one foot deep in a more or less random pattern. These grooves will have several effects. They will damage surviving *Phragmites* rhizomes, and, additionally, the scribing will disrupt rhizome air passages and allow water into the rhizome which will effectively drown the plant. The microtopography will create little holding basins for tidal waters, which will increase the wetness of the sediment. The grooves will not be connected in any systematic way to the tidal channels so they would not drain the sediments. The microtopography will also collect some sediment and water-borne seeds of *Spartina*, speeding up the desired revegetation. The breakup of the *Phragmites* rhizome mat will enhance the development of the first order tidal creeks and rivulets by which small fish obtain access to the marsh plain at high tide.

The second component of our revised plan incorporated aggressive source control. Previous investigations of *Phragmites* invasion pathways on these sites showed that *Phragmites* invaded the marsh plain by first becoming established on the dikes and drainage ditch levies that were above MHHW and dryer than the adjacent marsh plain. These high areas provided a location for *Phragmites* seeds to germinate. Once established, *Phragmites* rapidly moved on to the marsh by rhizome propagation. The source control techniques incorporated into the original design were limited removal of high areas, herbicide application, rhizome ripping, and mowing. These measures were designed to remove areas favorable for *Phragmites* seed germination and to inhibit rhizome propagation by damaging the *Phragmites* rhizomes.

DESIGN COMPARISON

On initial examination, the two restoration designs appear to be radically different but in reality they incorporate the same basic components. Each design was begun by gathering data on existing conditions and formulating a preliminary restoration design, which increased hydroperiod and provided for source control. At the salt hay farms, increasing hydroperiod was accomplished by breaching perimeter dikes and excavating channels. At the *Phragmites* sites, increasing hydroperiod was accomplished by constructing microtopography, which increased the hydroperiod on a micro-scale. Source control at the salt hay farm restoration sites was implicit in the design. Reintroducing tidal waters with relatively high salinity will accomplish source control at these sites. Source control at the *Phragmites* restoration sites was accomplished with more direct and aggressive measures. The final and most important component of both restoration designs is the application of adaptive management to monitor the restoration project to ensure that our restoration goals are accomplished. Both restoration projects will be monitored throughout future years and modifications to the design will be implemented if required, to ensure the restoration goals are met.

LITERATURE CITED

Ailstock, S. M. Undated. Environmental impacts, treatment methodologies and management criteria for establishment of a statewide policy for the control of the marsh plant *Phragmites*. Report to the Maryland Dep. Nat. Res., Tidewater Admin. Annapolis MD. 61p.

Barret, N. E. 1989. Vegetation of tidal wetlands of the lower Connecticut river: Ecological relationships of plant community-types with respect to flooding and habitat. Master's Thesis, Univ. Conn., Storrs CT.

Baumann, R. H., J. W. Day, and C. A. Miller. 1984. Mississippi deltaic wetland survival: Sedimentation versus coastal submergence. Science, 224, 1093-1095.

Bertness, M. D., 1984. Ribbed mussels and *Spartina alterniflora* production in a New England salt marsh. Ecology, 65: 1794-1807.

Haslem, S. M. 1973. Some aspects of the life history and autoecology of *Phragmites communis* Trin. Pol. Arch. Hydrobiol. 20: 79-100.

Hatton, R. S., R. D. Delaune, and W. H. Patrick. 1983. Sedimentation, accretion and subsidence in marshes of Barataria Basin, Louisiana. Lim. And Oceano., 28: 494-502.

Hellings, S. E., and J. L. Gallagher. 1992. The effects of salinity and flooding on *Phragmites australis*. J. App. Ecol. 29: 41-49.

Kubfn, P. and A. Meizer. 1997. Chronological relationship between eutrophication and reed decline in three lakes of southern Germany. Folia Geobot. Phytotax. 32: 15-23.

Kühl,H. and J.-G Kohl. 1993. Seasonal nitrogen dynamics in reed beds (*Phragmites australis*(Cav.)Trin. Ex. Steudel) in relation to productivity. Hydrobiologica 251: 1-12.

Marks, M., B. Lapin, and J. Randal. 1994. *Phragmites australis* (*P. cummunis*): Threats, management and monitoring. Natural Areas J.14: 285-294.

Mendelssohn, L. A. and K. L. McKee, 1987. Experimental field and greenhouse verification of the influence of saltwater intrusion and submergence on marsh deterioration: Mechanisms of action. In: Turner, R. E. and D. R. (eds.), *Causes of Wetland loss on the Coastal Central Gulf of Mexico.* Vol. 2 Technical Narrative. Final report submitted to MMS, New Orleans, LA. Contract #14-12-0001-32252. OCS study/MMS87-0120, 145-179.

Niering, W. A. and R. S. Warren. 1977. Our dynamic tidal marshes: vegetation changes as revealed by peat analysis. Conn. Arboretum Bulletin. 22. 12p.

Osterbrock, A. J. 1984. *Phragmites australis.* The problem and potential solutions. Ohio Field Office, Stewardship, 8p.

Reed, D. J. and D. R. Cahoon. 1992. The relationship between marsh surface topography, hydroperiod, and growth of *Spartina alterniflora* in a deteriorating Louisiana salt marsh. J. Coast. Res., 8: 77-87.

Roman, C. T., W. A. Niering, R. S. Warren. 1984. Salt marsh vegetation change in response to tidal restriction. Environ. Manag. 8: 141-150.

Sebold, K. R., 1992. From marsh to Farm: The landscape transformation of coastal New Jersey. U. S. Dept. of Interior, NPS Culteral Res. 95p.

Sharma, P., L. R. Gardner, W. S. Moore, and M. S. Bollinger, 1987. Sedimentation and bioturbation in a salt marsh as revealed ^{210}Pb, ^{137}Cs, ^{7}Be studies. Lim. and Ocean., 32: 313-326.

Squires, L. A. and A. G. van der Valk. 1992. Water depth tolerances of dominant emergant macrophytes. Can. J. Bot. 70: 1860-1867.

Tiner, R. 1995. *Phragmites* – Controlling the all-too—common reed. Mass. Wetlands Restor. Notes. 1: 1-3.

Tobler, F. 1943. Stengelbau, Festigkeits-Verwertungsuntorschiede beim Schilfrohr *Phragmites cummunis* Trin. Angew. Bot. 25: 165-177.

Tucker, G. C. 1990. The genera of *Arundinoideae (Gramineae)* in the southern United States. J. of the Arnold Arboretum. 71: 145-177.

Young, S. W., D. H. Davies, and P. J. M. Milligan. 1991. The potential of antomical features of the common reed *Phragmites australis* (Cav.) Trin. Ex Steudal as a biotic indicator of adjoining land use. Arch. Hydrobiol. 122: 297-304.

Young, S. W., P. J. M. Milligan, and D. H. Davies. 1997. Correlation of concentrations of water nutrients with anatomical features of three species of aquatic grass. Arch. Hydrobiol. 140: 419-430.

Warren, R. S. 1994. *Phragmites australis* on the tidelands of the Lower Connecticut River: Patterns of invasion and spread. Report to the Nature Conservancy. Connecticut Chapter.

Weishar, L. W., J. Teal and J. Balletto. 1996. The design process utilized to restore diked salt hay farms to natural marshes. Proc. of the 23rd Ann. Conf. On Ecosys. Rest. Tampa, FL.

Weishar, L. W., J. Teal and J. Balletto. 1997. Implementation of adaptive management to restoration of diked salt hay farms to natural marshes. Proc. of the 24th Ann. Conf. On Ecosys. Rest. Tampa, FL. (in press).

Weishar, L. W., J. Teal, and J. Balletto. 1998. The role of adaptive management in the restoration of degraded diked salt hay farm wetlands. Proc. Wetlands Eng. & River Rest. Conf. Denver, CO. (in press).

DESIGN AND CONSTRUCTION OF SALTMARSH NURSERIES IN TAMPA BAY AREA HIGH SCHOOLS

Peter A. Clark and Sari E. Schlossberg
Tampa BayWatch, Inc.
8401 Ninth Street North
Suite 230-B
St. Petersburg, FL 33702

ABSTRACT

Tampa BayWatch supports the efforts to protect and restore coastal wetland communities in the Tampa Bay estuary by coordinating the establishment of saltmarsh nurseries within the bay region's high school ecological or science clubs. To date, Tampa BayWatch has built eight school saltmarsh nurseries to grow *Spartina alterniflora* that is then available for large scale habitat restoration projects currently being accomplished in Tampa Bay by local, state and federal agencies. Each high school nursery can cultivate over 5,000 planting units every six months - providing enough plants for about two acres of new marsh restored back to Tampa Bay every year. Our high school program teaches students the value of maintaining a healthy environment while promoting public education and involvement in restoration activities of the students whose responsibility it will ultimately be to act as guardians of the environment.

INTRODUCTION

Estuaries are extremely productive ecosystems where saltwater from the ocean meets freshwater from inflowing rivers and streams in a semi-enclosed coastal body of water. The blend of bay environments - ranging from underwater seagrass meadows, to surrounding intertidal salt marshes and mangrove forests, to uplands - provides food and shelter for a multitude of wildlife and marine species.

Saltmarsh plants act as a filtering agent for stormwater runoff, and serve as a vital link in the marine food web. These marshes, which periodically become submerged with the rise and fall of tides, support crabs, shrimp, snails, mussels, juvenile fish and a variety of birds. Salt marshes also stabilize shorelines and buffer uplands from storms. However, coastal wetlands, including salt marshes, throughout the nation have suffered extensive losses which have resulted in major declines in fisheries and wildlife who depend upon these habitats for a portion of their life cycle.

Over the past 100 years, the Tampa Bay estuary has suffered from the loss of coastal salt marsh and mangrove habitats. Due to the dredging and filling for port and residential waterfront development, the Tampa Bay estuary has lost 44 percent of its original intertidal wetlands. Over the past 20 years the Tampa Bay community has acknowledged the tremendous habitat losses and have made significant strides to address the problem through permitting activities and habitat restoration efforts.

Restoration activities have become an increasingly popular tool to revitalize saltmarsh communities by re-grading shoreline elevations and planting native vegetation that will mimic natural communities. These restoration projects also give the public the opportunity to take an active role in restoring the environment while promoting education of volunteers on the bay's problems and solutions.

Tampa BayWatch supports large scale habitat restoration efforts by facilitating the construction of saltmarsh nurseries within the bay region's schools. The nurseries provide a source of native wetland plants to be used in habitat restoration projects around Tampa Bay. Our secondary school program teaches students the value of maintaining a healthy environment while promoting public education and hands-on habitat restoration activities.

STUDY SITE

Tampa BayWatch coordinates the establishment of saltmarsh nurseries within secondary school ecology and science clubs around Tampa Bay area, Florida.

MATERIALS AND METHODS

The schools must have a site to accommodate a 16' X 16' nursery that is secure and located near a freshwater source. The location of the nursery and preparation of the site is crucial to the success of the project. Like all plants, saltmarsh grasses need plenty of sunshine to grow. An open area, without overhanging trees, rooftops, etc., is an excellent spot to build the nursery.

Once a site is chosen, the ground is leveled and any debris is removed. Next, eight 4" X 4" X 8' wood boards are placed on the ground to form a 16' X 16' area. Landfill liner material (made of HDPE) is laid over the nursery to hold the water thus creating a pond. To prevent the liner from shifting, 1" X 2" X 8' wood strips are nailed on top of the wooden posts securing the liner in between the wood.

After the pond is constructed, an irrigation system will need to be built. Using schedule 40 PVC pipes and PVC connectors, water can be brought from a nearby watering source to the nursery with a timer device. All PVC pipeline should be buried to prevent unintentional breaks from foot traffic or lawn mowing. Once the water is brought to the nursery, a 14' PVC pipe will be placed on the bottom of the pond along one side wall with holes evenly spaced and pointing downward. The water should flow into the bottom of the pond so the plants can take up the water through their roots and less water will be lost from the nursery through evaporation, thus conserving water.

Native *Spartina alterniflora* plugs, or more commonly called smooth cordgrass, can be planted in a beach sand and peat mixture and placed in trays within the nursery once construction is completed. Generally, 2¼" (square tops) X 4" (deep) X 12 (in a cluster) peat pots are used to grow the *S. alterniflora*. Approximately 5,000 - 6,000 *S. alterniflora* are a reasonable number of plants to cultivate in a nursery. The process of obtaining smooth cordgrass to start the nursery should be a one time effort. Ideally, after initially stocking the nursery, the nursery can be recycled, always keeping enough salt marsh plants to begin a new growth cycle while still donating a sufficient amount of salt marsh plants to restoration projects.

When planting a school's nursery, the students are divided into three groups: Soil Mixers, Plant Separators, and Potters. The soil mixers are responsible for blending the soil ingredients. The soil mixture is made up of equal parts of soil (school yard soil or purchased peat) and sand (preferably beach sand) and a small amount of vermiculite and water until the mixture has a consistency of oatmeal. The plant separators are responsible for splitting the donor salt marsh into individual plants with roots intact. The potters are accountable for planting the individual plants into a planting tray using the soil mixture and then placing the planted trays into the nursery. Potters may also trim 6" - 8" off the top of the blades (depending on the height) of the *S. alterniflora* plant. Cutting the top blades of the plant stimulates roots to grow quicker.

Once all the trays have been filled and set into the pond, the nursery is partially filled with water, and salt (up to 10 - 15 parts per thousand) and fertilizer are added. The nursery is now complete and must be monitored on a weekly basis by the students and teachers. Salt should be added when needed to maintain a salinity reading between 10 - 12 ppt. The nursery should be watered once a week, but it is acceptable to allow the nursery to dry between watering cycles, but for no more than one week in duration.

After six to eight months of cultivating the salt marsh, the nursery is ready for a transplant. The nursery can be allowed to dry out for two or three weeks prior to transplanting, to aid in harvesting. A third to a half of the nursery should be saved and the plant plugs divided into individual plants and replanted into the nursery for another growing cycle. The other two-thirds to a half of the nursery's plugs should be used in a restoration project.

Transplanting procedures must incorporate three aspects in order to be successful and insure optimum benefits of re-vegetation - selection of a suitable site, determination of an appropriate tide, and proper transplanting techniques. Ideally, a calm backwater area should be chosen as the restoration site. Boat and human traffic, as well as wave action, should be at a minimum to prevent erosion of the newly transplanted cordgrass. The optimum elevation that *S. alterniflora* salt marsh grows in is +0.5' to +1.8' MLW (Mean Low Water). Planting events should take advantage of the lower tides (below +0.5' MLW) in order to help volunteers plant along semi-dry shorelines.

When planting, the planting rows should be about two to three feet apart from each other and along the contours of the shoreline. The spacing will allow for optimal growth from each plant. Each hole dug along the planting line should be spaced approximately two to three feet apart from the other holes. Again, the spacing will encourage optimal growth. Some fertilizer may be added in the hole before the plant is placed in the hole. Firmly place the soil on top of the planting units to fill the hole and secure the plug. The top of the soil from the planting unit should also be kept even with the soil at the restoration site. The salt marsh will grow together in about 18 months and mangroves will begin to sprout throughout the new marsh and eventually replace the salt marsh.

RESULTS

The High School Wetland Nursery Program alleviates a portion of the expenses incurred by publicly-funded agencies undertaking the costly process of restoring native habitat. With many restoration projects being planned, our high school salt marsh plants are in constant demand. The eight school wetland nurseries currently have the capacity to propagate a total of 80,000 planting units every year (10,000 planting units per nursery) with the potential to restore 14 acres of new salt marsh habitat per year. This is a significant contribution to the long-term health and recovery of the Tampa Bay estuary. Tampa BayWatch estimates an overall direct savings of $72,000 (or $0.90 per planting unit) each year in the cost of marsh grasses and planting charges for the public agencies involved in bay restoration activities.

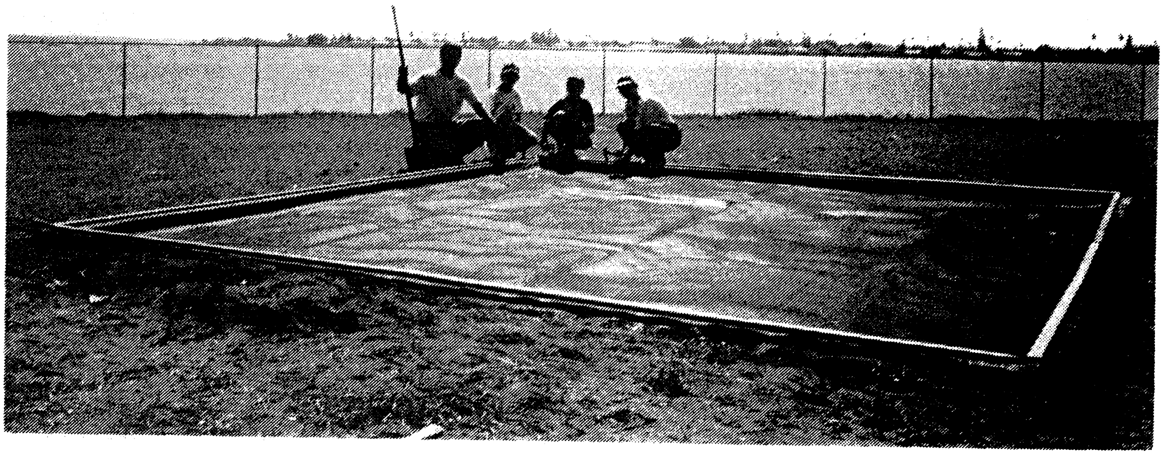
DISCUSSION AND/OR CONCLUSION

The program promotes student involvement with local scientists who are accomplishing community-based restoration activities. Through construction and maintenance, the young "biologists" monitor their nurseries by testing salinity, recording growth rates, performing routine maintenance and documenting other pertinent information. This not only insures proper growth within the nursery, but allows the students to gain knowledge about the intricacies of the nursery and restoration processes.

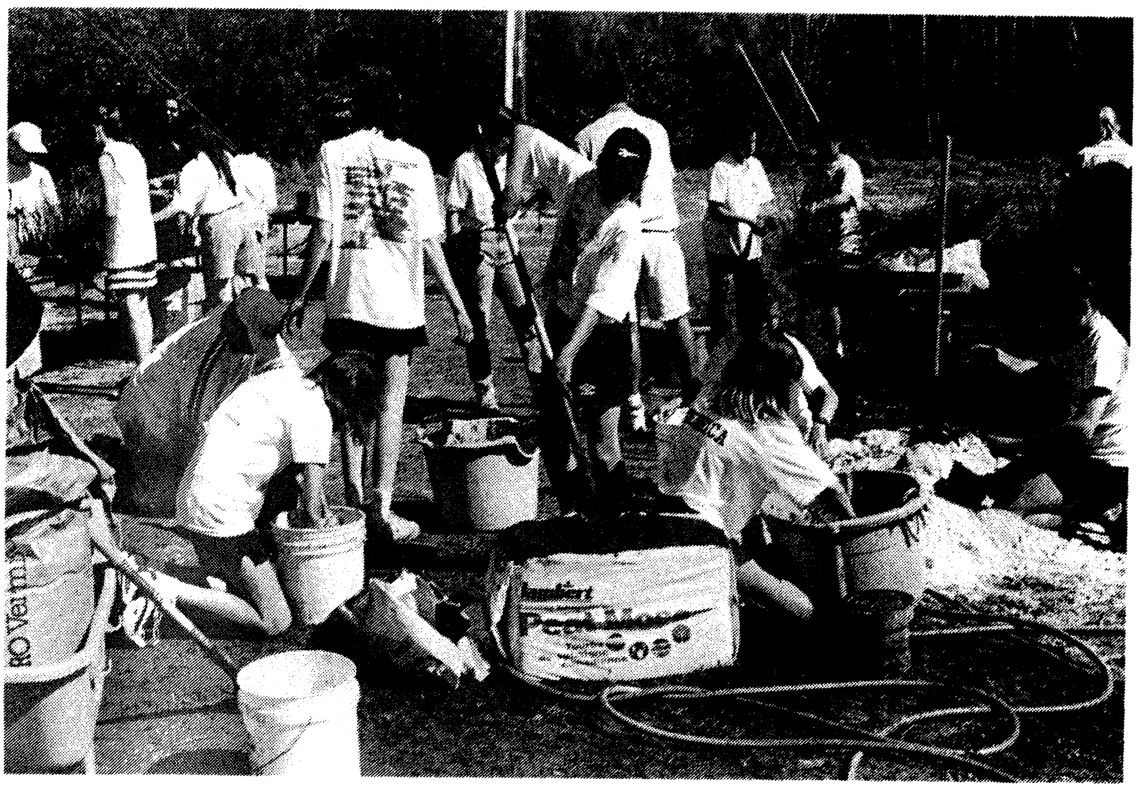
Due to Mother Nature and other ailments, the nurseries will not always produce the maximum number of plants, however, the knowledge gained by the students is never compromised. Each of the Tampa BayWatch school nurseries involves at least 100 - 200 students every year. These students would not otherwise be involved with such intensive bay restoration projects, or have the opportunity to work directly with scientists who are trained in bay restoration projects. Reaching these students now with a hands-on environmental message is important to their interest and involvement as adult citizens and bay managers of the future. The program instills in students an understanding and appreciation of the Tampa Bay estuary, the watershed that feeds it, and the wildlife that depend on it creating a heightened awareness of issues and providing intellectual incentive to students to change certain behaviors that impact the bay. A student who has worked to restore bay habitat systems is likely to become a more enlightened bay user, as well as an outspoken advocate for the bay to friends and family.

ACKNOWLEDGMENT

The High School Wetland Nursery Program would not be possible without the faculty, staff, and students from Bloomingdale, Chamberlain, Countryside, King, Tampa Bay Technical, and Lakewood high schools and Madeira Beach Middle School and Shorecrest Preparatory School. Funding was made possible through: the Tampa Bay National Estuary Program, Publix Supermarkets, Inc., Gardinier Settlement Trust Fund, National Marine Fisheries Service, and the Environmental Protection Agency. Furthermore, Southwest Florida Water Management District - Surface Water Improvement and Management Program, Florida Department of Environmental Protection, and the Environmental Protection Commission of Hillsborough County extended their local support and technical services.



1. Nurseries are constructed on school grounds and monitored for growth and success. This site is Madiera Beach Middle School's nursery in the early process of planting.



2. Students perform a variety of tasks during the construction and planting of the on-campus nurseries. Here, students at Shorecrest Preparatory School prepare the soil mixture.

3. Shorecrest Preparatory School students place the plants in rooting trays and add the soil mixture.

4. Lakewood High School's nursery after one growing season (6 - 8 months). Note the different growth rates of the school marsh grass. Growth rates and monitoring efforts provide a scientific exercise for students needing research projects.

5. Here is a salt marsh planting project using plants grown from several of the high schools. The High School Wetland Nursery Program provides a critical source of wetland plants and dedicated volunteers to support government habitat restoration programs while saving public funds.

6. Environmental education is a program priority as students are able to work side by side with area scientists to accomplish restoration of critical bay habitats. King High School students, during one of their 1997 transplants, check out other schools's restoration efforts from seven months earlier with staff scientists from Tampa BayWatch, SWFWMD-SWIM and Hillsborough County.

PRERESTORATION MONITORING OF FISHES IN A BORROW PIT CONNECTED TO TAMPA BAY, FLORIDA

[1]Kevin M. Peters and [2]Robert W. McWilliams
[1]Fla. Dept. of Environ. Protection, Fla. Mar. Res. Inst.,
100 Eighth Ave. SE, St. Petersburg, FL 33701
[2]Pinellas Co. Dept. of Environ. Mgmt.,
512 S. Ft. Harrison Ave., Clearwater, FL 33756.

ABSTRACT

We monitored fish species composition and relative abundance monthly from March 1995 to February 1996 at three sites along the shores of a eutrophic borrow pit that was to be restored. We found the pit to be highly productive during the year of sampling: a total of 119,857 fishes in 37 species were collected. Most were forage fishes (poeciliids, atherinids, engraulids, and cyprinodontids), although juvenile fishes of recreational and commercial importance (mullet, snook, and red drum) were also present. Subadult tarpon, adult mullet, and subadult to adult snook were found in ancillary collections of larger fishes. Habitat variables such as water temperature; shoreline slopes; and levels of salinity, dissolved oxygen, and sulfide were profiled.

These data were used to evaluate conditions in the borrow pit and to assess the efficacy of this restoration project. The pit contained habitat for both juvenile and adult fishes. The main factors affecting fish populations were the sulfide buildup (caused by eutrophic conditions in the deep basin) and the mixing of sulfide into the water column. However, conditions may be improved with relatively minor changes to the borrow pit itself. This study shows the need for both prerestoration monitoring and for tailoring restoration plans to a particular site.

INTRODUCTION

Many factors, such as property ownership, prior land use, and the condition of flora and fauna, are considered when deciding to restore particular sites. The City of St. Petersburg and the local Surface Water Improvement and Management office (SWIM)

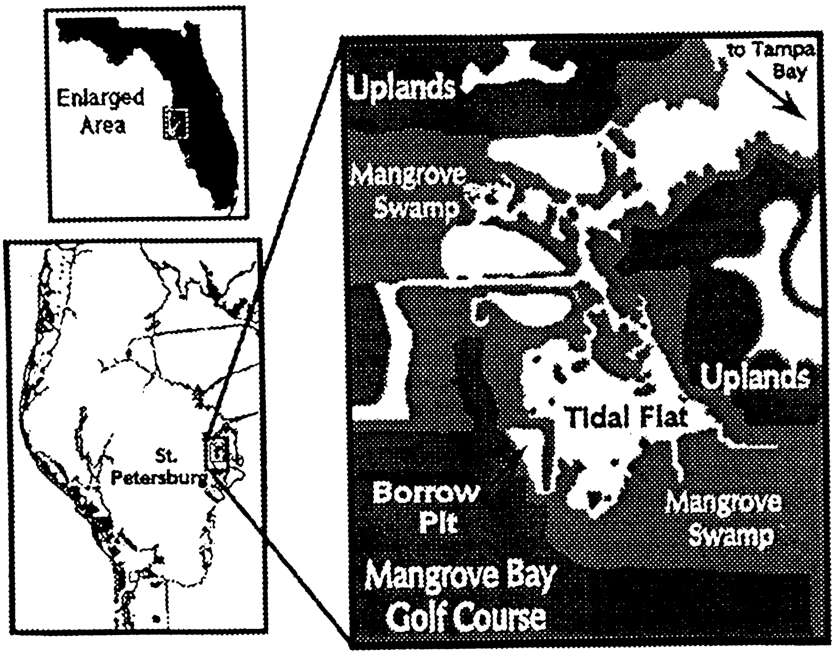


Figure 1. Location of borrow pit within east-central Pinellas County, Florida.

selected a borrow pit for restoration that was publicly owned, had previously been dredged, had poor tidal circulation, and was eutrophic (indicated by an apparent phytoplankton bloom on an aerial photograph and by reports of fish kills). This site was also adjacent to another restoration site.

Restoration plans included building sand spits in the borrow pit to create *Spartina* marshes and cutting new channels to increase flushing. There were, however, no long-term monitoring plans for faunal assessment, either before or after the restoration work. The Florida Department of Environmental Protection's Florida Marine Research Institute (FDEP) established a program in which small fishes were sampled monthly at the borrow pit; the goal of this program was to document the current state of shoreline ichthyofauna and associated habitat in the borrow pit so that researchers could provide advice concerning restoration plans.

STUDY SITE

This study was conducted in a borrow pit in the Mangrove Bay area of northeastern St. Petersburg, Pinellas County, Florida (Fig. 1). Soil from the borrow pit was removed in the 1950s and used to cover an adjacent land fill and to construct the city's Mangrove Bay Golf Course. The borrow pit is located 5 km from Tampa Bay to which it is connected by way of contiguous shallow tidal creeks, a large tidal flat, braided channels, and several small bays. The pit is in the shape of a right triangle, covers about 11,000 m², and has a maximum depth of 7.5 m (Fig. 2). The northern and eastern shores have steep banks; the western and southern shores (those adjacent to the golf course) have banks with more moderate slopes. Black mangroves (*Avicennia germinans*) are the predominant vegetation lining the shore, except at two open areas on the western shore.

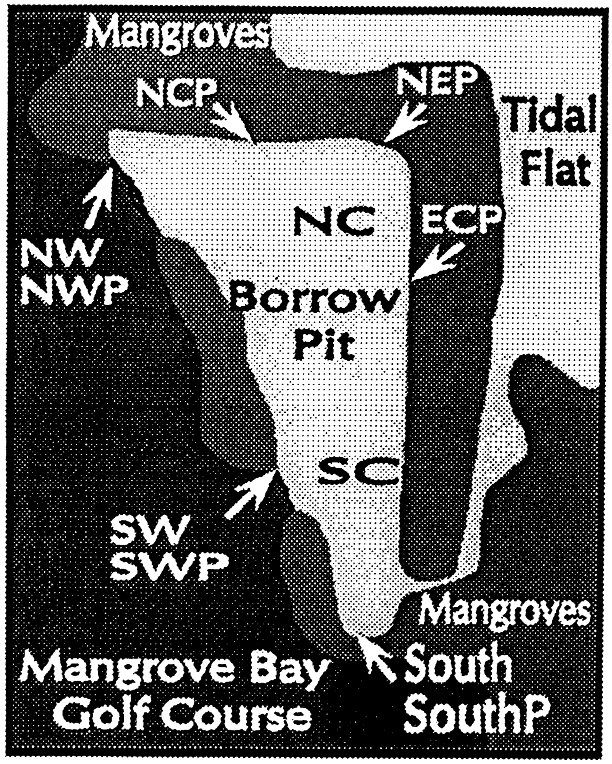


Figure 2. Detail of borrow pit. NC and SC: north-central and south-central water sample sites. ECP, NCP, NEP, NWP, SouthP, and SWP: east-central, north-central, north-east, south, and southwest bottom profile sites. NW, South, and SW: northwest, south, and southwest seine sites.

MATERIALS AND METHODS

The abundance and sizes of small fishes in the borrow pit, along with basic hydrological variables, were monitored monthly from Mar 1995 to Feb 1996. Sulfide levels were measured during Jun and Sep 1996.

HYDROLOGY: Water temperature and levels of salinity, pH, and dissolved oxygen (DO) were measured with a Hydrolab Surveyor III. Oxygen levels above the limits of our instrument are reported as >14.6 mg/l. Water clarity was measured with a Secchi disk, and extinction depths were recorded to the nearest 0.1 m. These variables were measured between 1100 h and 1300 h on each month's regular sampling date.

Hydrolab samples were taken in the north-central and south-central basins (NC and

SC; Fig. 2). Data reported herein are from the deeper, north-central basin site and includes hydrological data for the entire water column. Monthly measurements were taken at the surface and bottom and at 1-m depth intervals except during Mar and Apr 1995, when only surface, midwater, and bottom samples were taken. Hydrological measurements were taken on nine occasions in the tidal creek and on six occasions on the tidal flat. Tidal-creek samples were taken at the surface and at the bottom; tidal-flat samples were taken only at the bottom.

Water samples for sulfide analysis were taken during the summer at the surface and bottom and at 1-m depth intervals. One sample was taken during Jun 1996 after a period of windy weather so that we could document sulfide concentrations while the water column was mixed. Two samples were taken during Sep 1996 after a period of calm weather so that we could document sulfide concentrations while the water column was stratified. Sulfide samples were collected with glass syringes, preserved with sulfide antioxidant buffer, and returned to the laboratory. The sulfide concentration was measured to the nearest 0.1 micro mole (μM) by using sulfide-ion-specific electrodes. We took concurrent temperature and salinity profiles using the Hydrolab in order to document the degree of mixing or stratification of the water column.

BOTTOM PROFILES: The bottoms at six sites around the borrow pit were profiled for comparison with bottom profiles of a natural mangrove lagoon in lower Tampa Bay that has been documented to be a snook nursery (Peters et al., 1998). Borrow-pit sites included two sites each along the western (NWP and SWP) and northern (NCP and NEP) shores and one site each along the eastern (ECP) and southern (SouthP) shores (Fig. 2). Depths used for profiles were measured perpendicular to the shore at each meter of horizontal distance from the high-tide line to ~2-m depths. Water depths were recorded relative to mean lower low water (mllw). Slope measurements are rise divided by run (no units). Borrow-pit slopes were expressed as an average slope for an entire transect (average slope) and the maximum slope at the water's edge (intertidal slope).

ICHTHYOFAUNA: Small fishes were sampled monthly from Mar 1995 to Feb 1996 with a 1.6-mm-mesh, 1.2-m x 12-m beach seine. Each month, a single seine haul was taken at each of three fixed sites between 0900 h and 1200 h and during intermediate tides. Seine sites were located at the north end of the northwest clearing (NW), the north end of the southwest clearing (SW), and on the southern shore (South; Fig. 2).

The seine was deployed from the beach at the NW and SW sites and from a boat at the South site. The offshore end of the net was kept in front of the shoreward end as the net was pulled parallel to shore. The net was retrieved after covering a distance of about 12 m. The NW and SW sites were sampled to include equal amounts of mangrove and open shoreline during individual hauls. The South site had only mangrove shoreline.

Areal coverage of the individual sites was consistent between months, allowing us to compare fish abundance between months. In addition, estimates of fish densities were made. The area sampled during individual hauls (72 m^2) was estimated by multiplying the maximum distance offshore (6 m) by the length of the shoreline sampled (12 m).

During all months, fish >100 mm standard length (SL) were field identified, measured to the nearest 1.0 mm, and released. During Mar and Apr 1995, small fishes (<100 mm SL) were sorted by species, counted, and measured for minimum and maximum lengths to the nearest 1.0 mm SL before release. From May 1995 to Feb 1996, small fishes were returned to the laboratory, sorted by species, counted, and up to 20 randomly selected specimens per species were measured for length.

The occurrence of subadult and adult fishes in the borrow pit was documented mainly by observation. However, the presence of adults of fisheries species was confirmed through the use of trammel nets set around the perimeter of the borrow pit (B. Mahmoudi, FDEP).

RESULTS

HYDROLOGY: Water Temperature: Borrow-pit water temperatures varied from 15.4° (surface, Feb) to 30.4°C (surface, Oct; Table 1; Fig 3). Bottom-water temperatures varied from 16.2° to 27.0°C and were more stable than surface temperatures. Bottom temperatures were cooler than surface temperatures during the spring and summer (Mar to Oct) and warmer than surface temperatures during the fall and winter (Nov to Feb). Temperature changes between the surface and bottom were characterized by a gradual change during the months of Mar, Apr, Jul to Aug, and Oct. However, thermoclines were well developed in May, Sep, and from Nov to Feb, indicating density stratification in the water column.

Tidal-creek water temperatures varied from 14.4° to 29.6° C and tidal creek temperatures varied from 18.4° to 29.4°C (Table 1).Temperatures were lowest in Jan-Feb and highest in Jul. Tidal-flat temperatures were generally lower than borrow-pit surface-water temperatures during the summer and warmer than borrow-pit surface-water temperatures during winter.

Table 1. Water temperature and salinity in the borrow pit, in the tidal creek, and on the tidal flat, Mar 1995 - Feb 1996. Cs = tidal-creek surface water; F = tidal-flat bottom water; Pb = pit bottom water; Ps = pit surface water.

Water Temperature (° C)

	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb
Ps	23.0	24.6	24.1	28.9	29.6	29.4	28.4	30.4	20.8	19.7	16.3	15.4
Pb	17.6	20.7	21.8	27.0	25.7	25.5	25.7	26.0	24.4	20.4	16.7	16.2
Cs		26.5	23.1	28.8	29.6		27.9		20.6	19.0	18.1	14.4
F			23.0		29.4		27.8			22.0	18.4	19.9

Salinity (ppt)

	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb
Ps	23.4	23.3	23.9	25.8	17.2	20.8	16.6	12.3	17.6	20.0	18.0	20.2
Pb	29.2	24.2	27.3	26.3	27.5	27.8	26.7	26.8	26.2	24.4	24.3	23.9
Cs		23.7	23.8	26.4	17.1		16.7		17.8	19.8	18.5	20.1
F			23.8		16.7		16.7			21.8	18.5	22.4

Salinity: Borrow-pit salinity varied from 12.3 to 29.2 ppt. The minimum salinity occurred at the surface during Oct; the maximum salinity occurred at the bottom during Mar (Table 1; Fig. 3). Bottom salinities were high and stable, ranging from 23.9 to 29.2 ppt; surface salinities were lower and more variable, ranging from 12.3 to 25.8 ppt.

The water column was stratified in terms of salinity in every month except Apr and Jun (Fig. 3). Salinity stratification was most pronounced and closest to the surface in Jul, Sep, and Oct. During other months, salinity stratification occurred at depths ≥3 m.

Tidal-creek salinity varied from 16.7 to 26.4 ppt and tidal-flat salinity varied from 16.7 to 23.8 ppt (Table 1). At these sites, salinity was lowest during Jul and Sep and highest during May and Jun. Tidal-flat salinity was slightly lower than borrow-pit surface-water salinity in Jul and slightly higher than borrow-pit surface-water salinity in Dec and Feb.

Table 2. Levels of dissolved oxygen and of pH in the borrow pit, in the tidal creek, and on the tidal flat, Mar 1995 - Feb 1996. Cs = tidal-creek surface water; F = tidal-flat bottom water; Pb = pit bottom water; Ps = pit surface water.

Dissolved oxygen (mg/l)

	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb
Ps	>14.6	11.4	4.2	9.0	12.1	4.2	2.4	7.0	1.8	2.1	11.8	4.9
Pb	0.3	0.3	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.1	0.2
Cs		>14.6	5.3	9.2	12.6		2.8		2.0	2.0	9.3	5.3
F			7.3		7.1		4.0			>14.6	9.5	>14.6

pH

	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb
Ps	9.17	8.23	7.63	7.86	8.63	7.92	6.96	7.62	7.36	7.57	8.71	7.12
Pb	7.22	6.85	6.99	7.29	7.23	7.27	6.54	6.66	6.93	7.10	7.26	6.45
Cs		8.44	7.65	7.86	8.72		7.04		7.42	7.46	8.72	6.96
F			7.83		8.25		7.21			9.22	8.80	8.63

Dissolved Oxygen: Borrow-pit DO varied from <0.1 mg/l to >14.6 mg/l, with minimum DO levels at the bottom and maximum DO levels at the surface (Table 2; Fig. 3). Surface DO varied from 1.8 mg/l to >14.6 mg/l and followed no seasonal pattern; bottom DO was always <0.5 mg/l. Dissolved oxygen levels in the upper half of the water column were generally higher from Jan to Apr than from May to Dec.

Tidal-creek and tidal-flat DO levels varied from 2.0 to >14.6 mg/l. Dissolved oxygen levels in the tidal creek were lowest during Nov and Dec and highest during Apr. Dissolved oxygen levels on the flats were lowest during Sep and highest during Dec and Feb. Tidal-flat DO was somewhat higher than borrow-pit surface-water DO in May and Sep and much higher than borrow-pit surface-water DO in Dec and Feb.

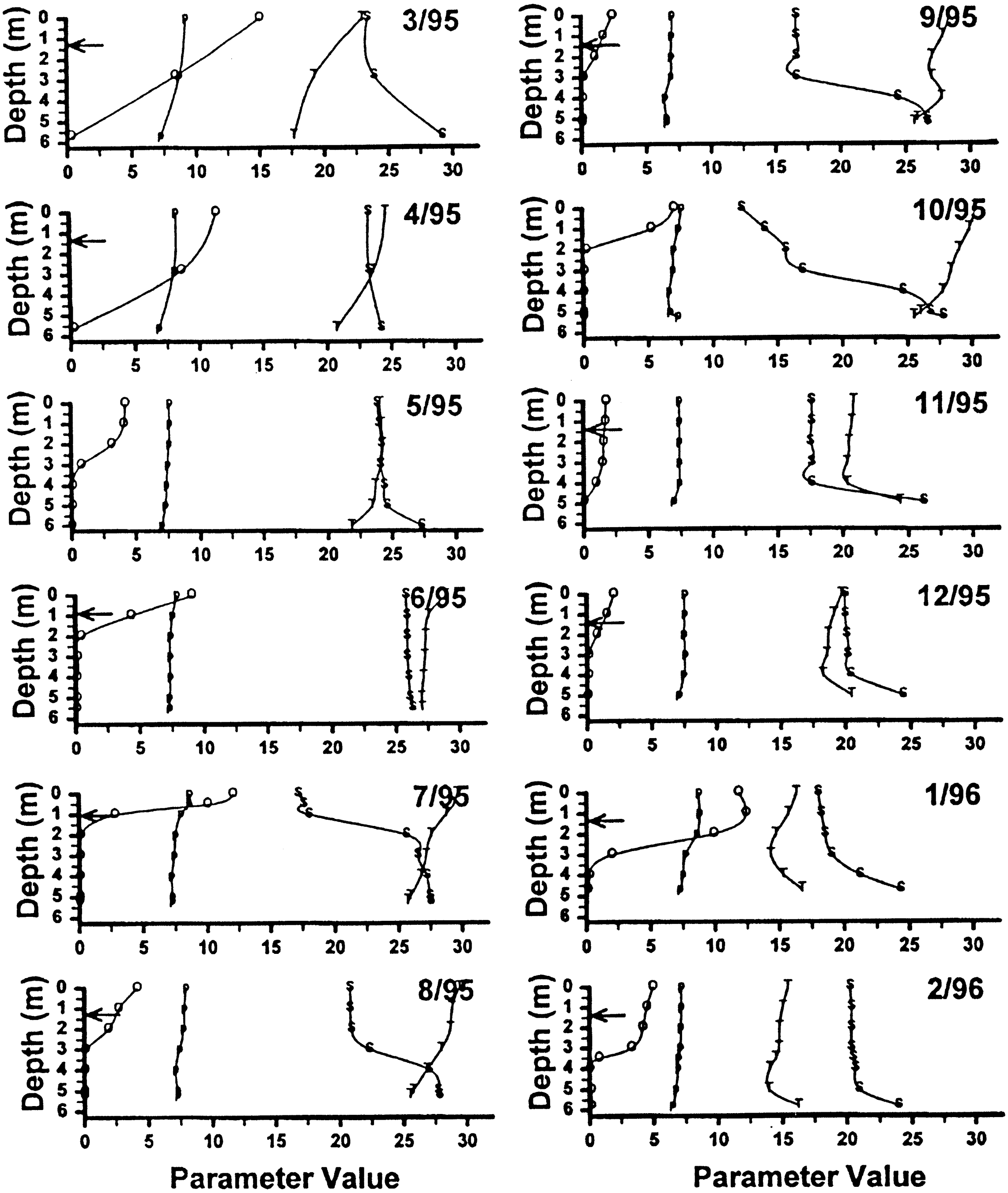


Figure 3. Regular monthly depth profiles of water temperature (T), salinity (S), dissolved oxygen (O), and pH (p) at the NC site of the borrow pit. Arrows denote Secchi disk measurements.

pH: Borrow-pit pH generally followed the pattern of DO (Table 2; Fig. 3). Surface-water pH varied from 6.96 to 9.17; bottom-water pH varied from 6.45 to 7.29. Surface and bottom pH followed similar patterns, with surface pH 0.42-1.95 units higher than bottom pH. Tidal-creek and tidal-flat pH varied from 7.04 to 9.22 and followed patterns similar to DO.

SULFIDE: Sulfide profiles of Jun and Sep samples were distinctly different (Fig. 4). During Jun, sulfide was present but was <10 μM throughout the water column. During both Sep 1996 sampling events, sulfide levels were <10 μM in the upper 1 m of the water column but began increasing at depths of 2-3 m and reached levels near 5,000 μM at the bottom of the water column. Temperature and salinity profiles taken at the time that sulfide samples were taken indicated that the water column was unstratified during Jun but was stratified in Sep (Fig. 5). Dissolved oxygen levels associated with these sulfide profiles were >1.0 mg/l at 4.0 m during Jun, but declined to 1.0 mg/l at 1.0-2.5 m during Sep.

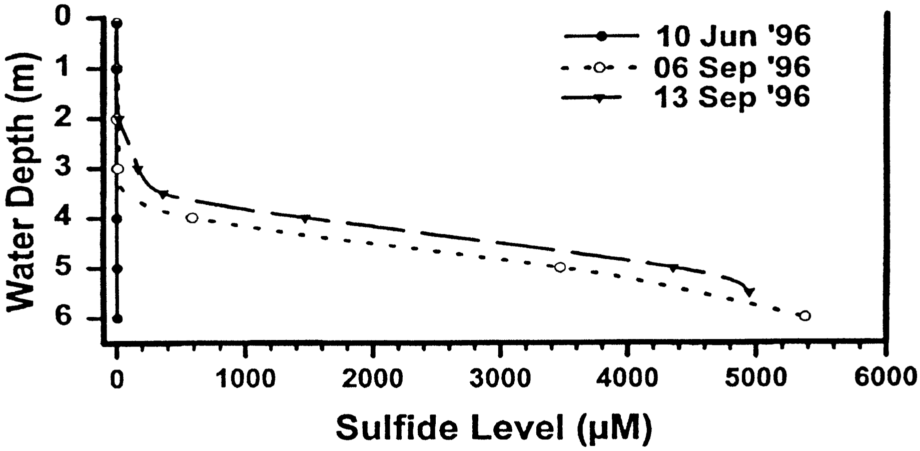


Figure 4. Depth profiles of sulfide levels measured at the NC borrow-pit site on three dates during 1996.

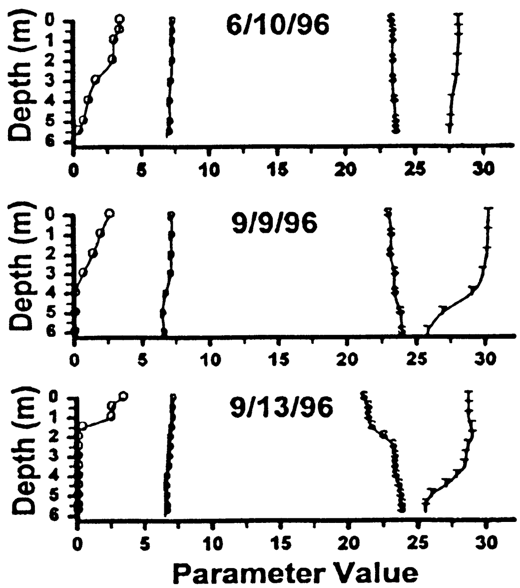


Figure 5. Water temperature (T), salinity (S), dissolved oxygen (O), and pH (p) profiles recorded at the time of sulfide sampling.

Table 3 . Bottom slopes of shores surrounding Mangrove Bay borrow pit. ECP = east-central profile; NCP = north-central profile; NEP =northeast profile;NWP = northwest profile; SouthP = southern profile; SWP = southwest profile (see Fig. 2).

Site		Transect Length (m)	Average slope over transect	Intertidal slope
Moderate slopes	NWP	15	0.17	0.15
(seined)	SWP	15	0.16	0.12
	SouthP	16	0.14	0.12
Steep slopes	NCP	11	0.27	2.68
(unseined)	NEP	13	0.20	4.00
	ECP	9	0.33	1.97

SECCHI EXTINCTION DEPTHS: Secchi extinction depths varied from 0.9 m (Jul) to 1.5 m (Sep; Fig. 3) and had a mean value of 1.3 m. There was no seasonal trend indicated for water clarity in the borrow pit.

BOTTOM PROFILES: The western and southern shores of the borrow pit were less steep than the northern and eastern shores (Table 3; Fig. 6). The average slopes of the western and southern shores (seined shores) were moderate (0.14 to 0.17), and varied little over the length of the transect. The average slopes of the northern and eastern shores (unseined shores) were steeper (0.20 to 0.33) than those of the western and southern shores. In addition, the intertidal zone of the northern and eastern shores was extremely steep (1.97 to 4.00).

ICHTHYOFAUNA: DIVERSITY: A total of 119,857 fish representing 37 species and 35 genera were collected in the 36 regular seine hauls (Table 4). Three additional species in two additional genera were collected in ancillary seine hauls (this study) and trammel nets (B. Mahmoudi, FDEP, pers. comm.). However, abundant fish species (those with >100 individuals) comprised only 17 species and accounted for

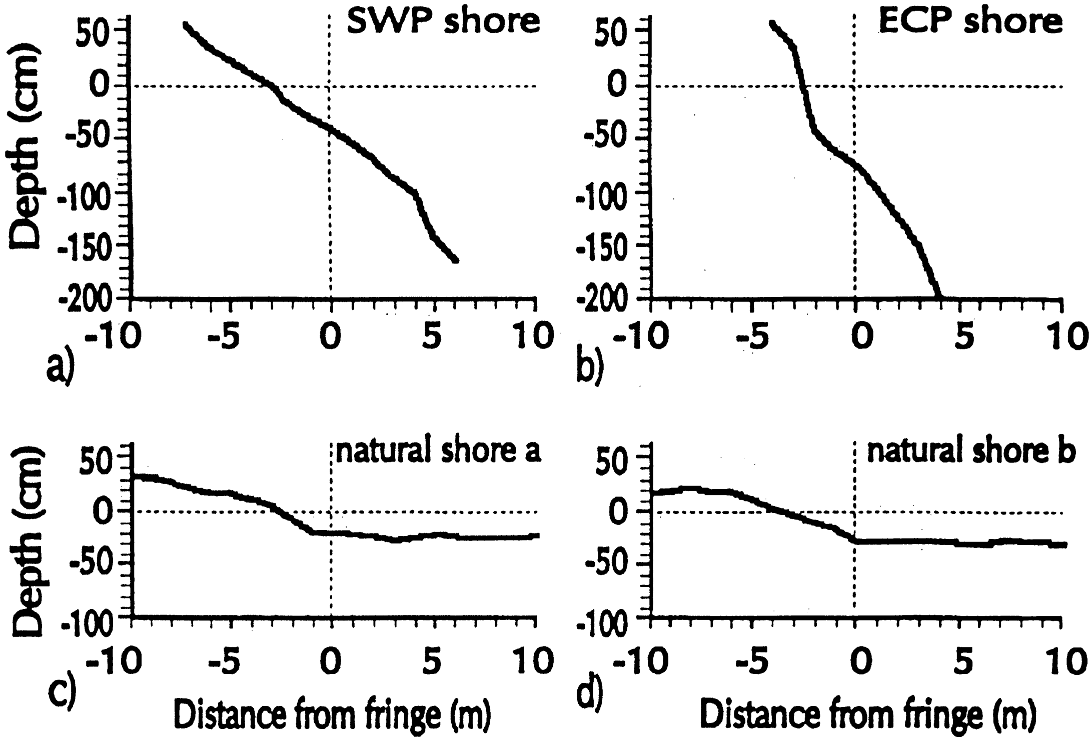


Figure 6. Bottom profiles from two borrow-pit sites (a and b) and two natural mangrove shorelines used as snook nurseries (c and d). Depths are relative to mean lower low water; distances are relative to the outer edge of the mangrove canopy.

99% of the total number of fishes. On a seasonal basis, species diversity was highest during May, Jun, and Dec, and lowest during Aug, Sep, and Mar (Fig. 7).

The 17 most abundant fish species were all considered forage species at small sizes. Ten species were small forage fishes, including the seven most abundant species (eastern mosquitofish, inland silversides, sailfin mollies, bay anchovies, rainwater killifish, sheepshead minnows, and clown gobies) and three additional species (gulf killifish, striped killifish, and goldspotted killifish). The remaining seven species (tidewater mojarra, spot, blue tilapia, pinfish, striped mullet, striped mojarra, and menhaden) were early stages of somewhat larger fishes whose young are typically abundant in the estuary and may be used as forage by piscivores.

Eleven commercially or recreationally harvested species were collected in the borrow

Table 4. The number (No.), percentage of total number (%), frequency of occurrence (Freq.), and minimum and maximum standard lengths (SL) of fishes collected in the borrow pit between Mar 1995 and Feb 1996. Species are listed in order of decreasing abundance. FS = fisheries species.

Species and Common Name	FS	No.	%	Freq.	SL (mm)
G *Sambusia holbrooki* (eastern mosquitofish)		44,731	37.3	1.00	6-38
Menidia beryllina (inland silverside)		19,402	16.2	1.00	7-68
Poecilia latipinna (sailfin molly)		15,778	13.2	1.00	8-62
Anchoa mitchilli (bay anchovy)		15,460	12.9	0.75	14-57
Lucania parva (rainwater killifish)		8,995	7.5	0.92	7-40
Cyprinodon variegatus (sheepshead minnow)		6,628	5.5	0.92	6-43
Microgobius gulosus (clown goby)		2,724	2.3	1.00	7-52
Eucinostomus harengulus (tidewater mojarra)		1,652	1.4	0.83	10-88
***Leiostomus xanthurus* (spot)**	X	1,331	1.1	0.50	10-82
Tilapia aurea (blue tilapia)		503	0.4	0.83	7-315
Lagodon rhomboides (pinfish)		396	0.3	0.50	10-84
Fundulus grandis (gulf killifish)		388	0.3	1.00	8-100
Fundulus majalis (striped killifish)		378	0.3	0.75	9-67
***Mugil cephalus* (striped mullet)**	X	276	0.2	0.67	19-150*
***Diapterus plumieri* (striped mojarra)**	X	238	0.2	0.83	5-155
Floridichthys carpio (goldspotted killifish)		205	0.2	0.42	10-32
***Brevoortia* sp. (menhaden)**	X	108	0.1	0.50	16-34
***Centropomus undecimalis* (common snook)**	X	96	0.1	0.50	15-193*
***Sciaenops ocellatus* (red drum)**	X	86	0.1	0.42	19-138
Syngnathus scovelli (gulf pipefish)		67	0.1	0.58	26-97
Gobiosoma bosc (naked goby)		66	0.1	0.58	6-39
Achirus lineatus (lined sole)		60	<0.05	0.67	13-32
Fundulus confluentus (marsh killifish)		60	<0.05	0.58	9-36
Arius felis (hardhead catfish)		58	<0.05	0.25	270-330

Table 4. Continued.

Species and Common Name	FS	No	%	Freq.	SL
***Pogonias cromis* (black drum)**	X	44	<0.05	0.17	8-53
***Elops saurus* (ladyfish)**	X	43	<0.05	0.25	21-61
Adinia xenica (diamond killifish)		17	<0.05	0.25	19-28
Gobiosoma robustum (code goby)		15	<0.05	0.17	11-27
***Cynoscion nebulosus* (spotted seatrout)**	X	14	<0.05	0.42	33-185
Strongylura marina (Atlantic needlefish)		13	<0.05	0.25	50-231
Trinectes maculatus (hogchoker)		11	<0.05	0.25	10-37
Strongylura notata (redfin needlefish)		5	<0.05	0.17	270-297
Opsanus beta (gulf toadfish)		3	<0.05	0.17	10-16
Oligoplites saurus (leatherjack)		2	<0.05	0.17	21-29
***Cynoscion arenarius* (sand seatrout)**	X	2	<0.05	0.17	32-36
Syngnathus louisianae (chain pipefish)		1	<0.05	0.08	156
Gobiesox strumosus (skilletfish)		1	<0.05	0.08	13
TOTAL OF ALL FISH		119,857	100		
Ancillary Collections					
Strongylura timucu (timucu)		(1)			(236)
B. Mahmoudi Collections					
***Megalops atlantica* (tarpon)**	X	few			subadult
Chaetodipterus faber (spadefish)		(1)			(336)

* larger specimens reported from B. Mahmoudi collections.

pit (Table 4). These species made up 1.5% of the total number of fishes. The four most abundant fisheries species (spot, striped mullet, striped mojarra, and menhaden), each represented by more than 100 individuals, were species whose young are considered forage. The remaining fisheries species were juveniles or subadults of large predatory species (snook, red drum, black drum, ladyfish, spotted seatrout, sand seatrout, and tarpon); of these, snook juveniles were the most abundant (n = 96).

FREQUENCY OF OCCURRENCE: Six of the eight most abundant forage fishes were full-time residents of the borrow pit (Table 4). Four of these (eastern mosquitofish, inland silversides, sailfin mollies, and clown gobies) were present every month (frequency of occurrence = 1.00), and the two others (rainwater killifish and sheepshead minnows) were present during every month except Mar (frequency of occurrence = 0.92). The remaining two of the eight most abundant forage fishes (bay anchovy and tidewater mojarra) were seasonally abundant in the borrow pit but were collected less frequently (frequencies of occurrence = 0.75 and 0.83, respectively) than the other six species. Bay anchovies were abundant only during Dec; tidewater mojarras were abundant only during Mar.

The young of fisheries species were usually more transient than forage species. Striped mojarras were the most common (frequency of occurrence = 0.83) followed by striped mullet (frequency of occurrence = 0.67). Spot, menhaden, and snook were collected in half the samples (frequencies of occurrence = 0.50), but other fisheries species were not commonly caught (frequencies of occurrence <0.50).

ABUNDANCE AND SEASONALITY: The 119,857 fish collected in our seines

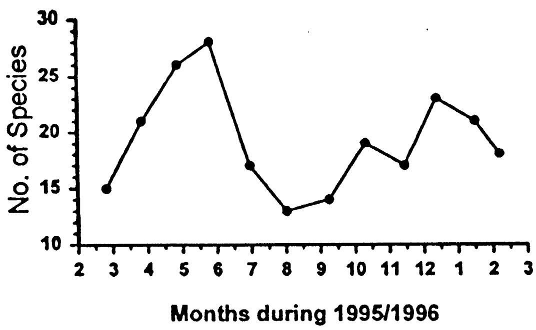


Figure 7. Changes in fish species diversity in the borrow pit during regular sampling.

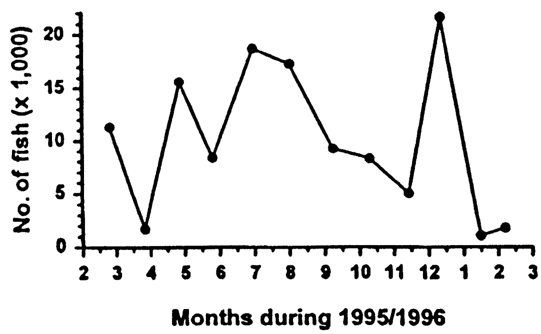


Figure 8. Number of fish collected monthly in borrow pit. Values are totals from three 12-m-seine hauls per month.

resulted in an overall mean density estimate of 46 fish/m². Average monthly densities varied from 5 fish/m² in Jan (based on 1,088 fish) to 100 fish/m² in Dec (based on 21,590 fish). Average site-specific densities varied from ~1 fish/m² at the NW site in Jan (n = 87 fish) to ~133 fish/m² at the SW site in Dec (n = 9,595 fish). Fish abundance generally increased during the late spring and early summer and, except for Dec, declined during the fall and winter (Fig. 8).

Forage species such as the livebearing poeciliids (mosquitofish and mollies), silversides, and anchovies were the dominant taxa in the borrow pit. The poeciliids were the most abundant taxa in Mar and from Jul to Nov (with highest abundance in Jul and Aug). Silversides were the most abundant taxon in May and Jun, whereas anchovies were abundant only during Dec. Other species that were abundant for one or two months included tidewater mojarras and spot (Mar), sheepshead minnows (May), rainwater killifish (Jul and Dec), and clown gobies (Oct). Few forage species were present during Jan, Feb, and Apr.

Among the young of large predatory fishes, snook was the only species abundant for multiple months (Aug-Nov). Other young predatory species were abundant only for a single month, including black drum and ladyfish (May), spotted seatrout (Oct), and red drum (Dec). Young of large predatory species were rare during Jan, Feb, Apr, Jun, and Jul.

ANCILLARY FISH DATA: Subadult and adult snook, striped mojarras, mullet, and tarpon were the fisheries species commonly captured in trammel nets in the borrow pit (B. Mahmoudi, FDEP; pers. comm.). These collections documented the occurrence of large snook and striped mullet throughout the year and of tarpon during the winter. Adult snook were observed in the borrow pit and tidal creek, but we never captured these large fish in our seine. One notable seasonal change observed in adult fish abundance was an increase in the number of mullet during Sep and Oct.

DISCUSSION

HYDROLOGY: The hydrology of borrow-pit water seemed to be influenced by its greater depth relative to surrounding water; stratification of the water column; its small size and vegetated shores (which limited wind-generated circulation); and its tidal exchange, principally with the flats (which integrated the hydrology of the two basins). The relatively calm, deep water allowed for density stratification and a buildup of sulfide. Periodic mixing of the water column would bring sulfide and hypoxic bottom water to the surface, where sulfide is rapidly oxidized. This process not only increases toxic sulfide in surface water but also causes DO declines. The amount of sulfide buildup and the severity of the mixing event would determine the degree of stress on fish. However, any decline in DO associated with mixing was apparently short lived; the primary indicators of mixing events were changes in temperature and salinity profiles between months.

Temperature, salinity, and sulfide profiles from 6 and 13 Sep 1996 indicated a high buildup of sulfide during calm weather and stratified water conditions. In contrast, temperature, salinity, and sulfide profiles from 10 Jun 1996 indicated reduced sulfide levels after the water column had been mixed by strong winds. The alternating buildup of sulfide in bottom water and mixing of sulfide into surface water appears to be a common event affecting the fauna of the borrow pit.

In addition to the direct effects of sulfide, reduced DO was occasionally detected in near-surface water and may have affected shoreline inhabitants. The decline of DO near the surface during Jun and Jul probably restricted the depth distribution of fishes and other fauna to the upper 1 m of the water column and, therefore, would have restricted the distribution of shore fauna to the upper 1 m of shoreline slope.

Dissolved oxygen levels >10 mg/l in surface waters during Jan, Mar, Apr, and Jul may have been related to extremely high plant productivity in the area. However, there was no evidence of phyto-plankton blooms; Secchi depths were ~1 m, regardless of DO levels. The mean Secchi disk reading of 1.3 m for the year may indicate that the borrow pit is an effective settling basin for particulates or that particulates settle out on the tidal flat before water reenters the borrow pit.

Borrow-pit hydrology was influenced by conditions on the tidal flat in other ways as well. Dissolved oxygen levels were frequently higher in the creek and flats than in the borrow pit, and comparisons among sites during flood tides indicated that tidal currents were moving high-DO water from the flats into the pit. This was evident during months when we sampled during flood tides but likely occurred during other months as well. Thus, oxygen-rich water from the tidal flats increased the DO in the pit and may have helped sustain a higher productivity than would have occurred otherwise. Eutrophic conditions in the area also produced mats of drift algae (*Ulva* spp. and *Gracilaria* spp.) on the flats that became partially desiccated during winter low tides. The algae was then pushed into the borrow pit by wind and flood-tide currents, where it sank to the bottom and decomposed. Nutrients released by this decomposition possibly helped to sustain the area's eutrophic conditions.

Tidal-flat and/or tidal-creek temperature and salinity sometimes differed from those in the borrow pit even though these waters were constantly mixing with each other. These differences in temperature and salinity could potentially have affected borrow-pit circulation or stratification as flood-tide water moved from the tidal flat into the borrow pit.

BOTTOM PROFILES: The western and southern shores of the borrow pit were less steep than the northern and eastern shores, but all borrow-pit shores were steeper than the natural lagoon in lower Tampa Bay was (Fig. 6). The slope of the natural lagoon averaged 0.07 for the 10 m shoreward of the mangrove fringe and leveled off at about

0.3 m below mean lower low water for 10 m outside the mangrove fringe. The natural slopes were less than half the slope of the shores in the borrow pit that we seined.

However, the western shore of the borrow pit was shoal enough that large overhanging black mangrove branches resting on the sediment continued to grow and produced a dense canopy over the water that provided numerous branches and snags beneath the surface. This also created habitat for juvenile snook, albeit in a narrower zone than was produced by red mangroves in the natural lagoon. The steep slope and deeper basin of the borrow pit limited the extent of intertidal and subtidal area beneath the black mangroves to 6-7 m, whereas the corresponding area of red mangroves in the natural lagoon was 20-30 m wide.

The northern and eastern borrow-pit shores were extremely steep within the intertidal region, severely limiting the region of overhanging mangroves. This resulted in less canopy cover, underwater structure, and shallow-water habitat for small fishes than was found along the western shore. Almost no fish were collected during our attempts to seine here.

ICHTHYOFAUNA: Fish production and diversity in the borrow pit were relatively high despite the altered habitat and harsh conditions. However, the three species with the highest abundances and frequencies of occurrence (eastern mosquitofish, inland silversides, and sailfin mollies) are year-round residents and are morphologically adapted to respire near the surface during periods of low DO (Lewis, 1970; Kramer, 1987). These same adaptations probably help mosquitofish, silversides, and mollies survive sulfide in the water column. The only abundant fisheries species that was caught over multiple months in the borrow pit was juvenile common snook. This species not only moves towards the surface during hypoxic events but is also physiologically able to tolerate low levels of DO (Peterson et al., 1991; Peterson and Gilmore, 1991).

Subadults and adults of several fisheries species were also abundant in the borrow pit and may be more tolerant of sulfide than smaller fish are, or they may simply leave if conditions decline. Adult mullet were abundant during the summer but reached highest abundance in the fall prior to their offshore, spawning migration. Snook fed in the borrow pit during the summer, and both snook and tarpon sought refuge and/or fed in the borrow pit during winter. The authors' discussions with fishermen and biologists indicate that snook and tarpon seek deep, protected basins such as the borrow pit during the winter for thermal refuge.

Many species (bay anchovies, sheepshead minnows, tidewater mojarras, ladyfish, spot, black drum, and red drum) were abundant for mainly just one month of the year.

Their declines in number coincided with increased mixing of the water column, particularly in Jan, Feb, Apr, and Jun. Even poeciliid numbers declined after Jan and Jun mixing events.

We observed the consequences of wind-induced mixing after a Dec cold front. During this event, numerous herons, egrets, and gulls were seen feeding on stressed fish. Our seines collected a very high number of anchovies despite, or possibly because of, the mixing event because the stressed anchovies may have been unable to avoid our seine. A decline in fish abundance due to this and possibly other mixing events was not apparent until Jan and Feb samples. Such low Jan and Feb fish numbers were probably not the seasonal norm because the previous winter's sample (Mar) contained relatively greater numbers of fish in both resident (mosquitofish, mollies) and transient (gerreids, spot, pinfish) categories.

The relatively infrequent occurrences of some fisheries species in the borrow pit may have been partially due to the steep bottom slopes and narrow shelf around the pit or to the effects of sulfide on young fish. For example, the number of young striped mullet, black drum, and red drum may have been low due to a lack of appropriate shallow-water habitat; however, the low recruitment of mullet during Apr and a decline in the abundance of black drum in Jun and red drum in Jan, all concurrent with water-column mixing, suggest that these species were affected by sulfide.

The livebearers (mosquitofish and mollies) that dominated monthly collections are important forage for juvenile snook and other piscivores (Harrington and Harrington, 1961; Gilmore et al., 1983). Juvenile snook in particular are dependent on the neonatal young of livebearers as they progress from a planktonic to a piscivorous diet and the peak in snook abundance followed that of mosquitofish by just one month. Thus, the production of snook in the borrow pit may depend not only on water quality and physical habitat, but also on a healthy poeciliid population.

RECOMMENDATIONS: An important step in rehabilitating the borrow pit would be to identify and curb the source of nutrients that are thought to stimulate the eutrophic conditions, regardless of whether these nutrients enter the borrow pit via runoff, algae decomposition, or tidal exchange with the flat. Reduction of organics in the system may help reduce the anaerobic decomposition and sulfide buildup. Alternately, generated sulfide might be dissipated by reducing the pit's depth or by artificially mixing the water column with aerators. Caution should be taken to implement these options without destroying the thermal and calm-water aspects important to young snook and overwintering adult fishes.

Snook should be considered a priority species in restoration planning because it is a very important part of Florida's fisheries, and young snook were the most abundant recreational-fisheries species found in the borrow pit. Efforts should be made to retain and enhance the densely vegetated and moderately sloping shorelines that are important habitats for juvenile snook. Although not identical to the natural lagoon, the western shore did support juvenile snook and might be improved for this purpose by reducing the nearshore slopes to about half their present values. Slightly decreased shoreline slopes and the establishment of red mangroves would create a canopy and underwater structure (via prop root formation) that is better suited to juvenile snook (Peters et al., 1998).

Shallow-water spits planned for the borrow pit should have minimal connections to the shoreline to minimize the amount of impacted mangrove fringe and should be constructed primarily along the northern and eastern (steep) shores to minimize the impact on the shores currently used by young snook. Spit slopes should be designed to add shorelines appropriate for juvenile snook habitat but might also include shallower sand and mud shores to increase nursery habitats of other fisheries species, such as red drum, black drum, and striped mullet.

A portion of the deep water and the steep, vegetated shoreline habitat should be retained for the large mullet, snook, and tarpon currently using the pit. These species cannot flourish in the area without places such as the borrow pit for feeding and winter refuge. It would be senseless to enhance the survival of young fishes at the expense of adult fish habitat.

Additional tidal cuts for the purpose of flushing the pit do not seem advisable because they would simply move eutrophic conditions into a different semienclosed, shallow-water system (the braided channels) that would exchange water with the borrow pit. Additional channels would also divide the tidal energy such that, without maintenance dredging, the channels could silt in or occlude. The present tidal creek is very productive in its own right. It was used by subadult snook in summer and by small fishes all year. Small fishes were particularly abundant in the creek during low tides in winter.

Finally, we cannot predict *a priori* whether proposed changes will produce a net gain in the fisheries resources of the area because the system is already highly productive, and any changes, including nutrient reduction, may reduce total productivity. For example, enhancement of juvenile red drum habitat may reduce the juvenile snook population, and increases in the proportion of juvenile fish habitat might reduce the amount of habitat available to adult fishes. In this situation, where the borrow pit has been re-naturalized and already has a wealth of juveniles, subadults, and adults of fisheries species, the goal of restoration should be to work with the existing system and produce a better-balanced and more stable system with improved conditions for the current inhabitants.

ACKNOWLEDGMENTS

We thank M. Colaizzi (City of St. Petersburg), B. Henningsen (SWIM), and R. Whitman (Peninsula Design and Engineering, Inc.) for their assistance and the staff of Mangrove Bay Golf Course for providing access to the borrow pit. D. Leffler, J. Leiby, R. Matheson, R. McMichael, G. McRae, G. Nelson, and J. Quinn provided helpful comments on the manuscript. Partial funding for this study was provided by Florida recreational saltwater fishing license fees.

LITERATURE CITED

Gilmore, R. G., C. J. Donohoe, and D. W. Cooke. 1983. Observations on the distribution and biology of east-central Florida populations of the common snook, *Centropomus undecimalis* (Bloch). Florida Scientist 46: 313-336.

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

Kramer, D. J. 1987. Dissolved oxygen and fish behavior. Environmental Biology of Fishes 18: 81-92.

Lewis, W. M., Jr. 1970. Morphological adaptations of cyprinodontoids for inhabiting oxygen deficient waters. Copeia 1970: 319-326.

Peters, K. M., R. E. Matheson, and R. G. Taylor. 1998. Reproduction and early life history of common snook, *Centropomus undecimalis* (Bloch), in Florida. Bulletin of Marine Science 62: 509-529.

Peterson, M. S., R. E. Brockmeyer, Jr., and D. M. Scheidt. 1991. Hypoxia-induced changes in vertical position and activity in juvenile snook, *Centropomus undecimalis*: its potential role in survival. Florida Scientist 54: 173-178.

Peterson, M. S., and R. G. Gilmore, Jr. 1991. Eco-physiology of juvenile snook *Centropomus undecimalis* (Bloch): life-history implications. Bulletin of Marine Science 48: 46-57.

COAST 2050-A STRATEGIC COASTAL PLAN FOR LOUISIANA

L. Phil Pittman
Louisiana Department of Natural Resources
Coastal Restoration Division
P.O. Box 94396, Baton Rouge, Louisiana 70804-9396
Phone: (504) 342-9429
Fax: (504) 342-9417
E-Mail: philp@dnr.state.la.us

Bryan P. Piazza
Louisiana Department of Natural Resources
Coastal Restoration Division
P.O. Box 94396, Baton Rouge, Louisiana 70804-9396
Phone: (504) 342-6307
Fax: (504) 342-9417
E-Mail: bryanp@dnr.state.la.us

Bill Good, Ph.D.
Louisiana Department of Natural Resources
Coastal Restoration Division
P.O. Box 94396, Baton Rouge, Louisiana 70804-9396
Phone: (504) 342-6028
Fax: (504) 342-9417
E-Mail: billg@dnr.state.la.us

ABSTRACT

Coast 2050 is a multi-disciplinary approach to developing and implementing a strategic coastal plan for Louisiana's coastal zone and its valuable natural resources. The plan involves the collective effort of various federal, state and local agencies, as well as parish governments, landowners, environmental groups, and concerned citizens. This strategic plan has, as its mission, to "develop, by December 22, 1998, in partnership with the public, a technically sound strategic plan to sustain coastal resources and provide an integrated multiple use approach to ecosystem management."

A strategic coastal plan will result from the Coast 2050 initiative. This plan will include ecosystem management strategies designed to achieve desired objectives,

such as wildlife and fisheries production, flood protection, land loss reduction, and many others which will be identified during the development of the program. Once the plan is developed in accordance with an approved participation guide, the appropriate state and federal task forces and authorities will proceed to establish the plan, with or without amendments, as coastal policy. This policy is expected to form the basis for an amended Breaux Act Restoration Plan, the State's strategic plan, and coastal zone management guidance.

INTRODUCTION

The coastal zone of Louisiana was built by 16 deltaic episodes of the Mississippi River over thousands of years (Kolb and van Lopik 1958). As the river met the Gulf of Mexico and it's velocity decreased, the river deposited its sediment and built a delta. Over time, the river would change its course and build a new delta. As this progradational cycle was occurring on one delta, a degradational cycle would begin on a previously built delta (Gosselink 1984). Marine processes would eventually cause the land to sink and the sea to encroach on what was once land. This process of progradation and degradation has been taking place naturally for thousands of years. As man populated areas along the river ridge, converted the flood plain into agricultural fields, and developed the need to move commodities up and down the Mississippi, the need for control of the river emerged. By the 1940's massive flood control levees were erected along the length of the lower Mississippi to contain the river to its main channel, and these same levees in conjunction with the Old River Control Structure prevented the river from changing its course again. Sediment is now being lost as it is directed off the continental shelf instead of being used to build land. This means that the natural land loss process can no longer be subsidized by the land gain process. Land loss is now a major problem.

The current land loss rate in Louisiana's coastal zone is estimated to be somewhere between 25 and 35 square miles per year (Barras et al. 1994). Causes of land loss range from natural phenomena, such as wind and wave action, subsidence, sea level rise, saltwater intrusion, hurricanes and herbivory, to man-made causes such as oil and gas exploration, canalization, channelization and development. Effects, such as increased flooding, increased maintenance costs for roads, bridges and other infrastructure, and possibly even declining fish and shellfish stocks can be attributed to such land loss. Projections over the next 50 years indicate that such problems will not be alleviated but will probably become much more severe. It was clear to Louisiana's environmental resource agencies that a comprehensive planning effort to address the land loss was of utmost importance and urgency. The Coast 2050 planning initiative provides such a process.

On May 1, 1997, Governor Murphy J. "Mike" Foster stated in his May Day Proclamation that, "It is time for us to recognize that if we are to be truly successful in our efforts to restore our coast to a state of sustainable, productive health, that we must dedicate ourselves to the cause of protecting and restoring our coastal wetlands and barrier islands in order that we secure for ourselves and those who follow us the blessings and values that we and our forebears have enjoyed from these natural resources." In response to the Governor's proclamation a new comprehensive coastal planning initiative called Coast 2050 was begun.

BACKGROUND

Louisiana contains approximately 7.9 million acres of coastal wetlands (Zobrist et al. 1995) and a total of 7,791 miles of meandering shoreline that extends from the Pearl River on the Louisiana/Mississippi border westward to the Sabine River on the Texas/Louisiana border. (Lindstedt et al.1991). Louisiana's coastal zone is located in 19 southern parishes and forms an intricate interweaving of ecological systems. Coastal wetlands provide a wide array of benefits including fish and wildlife habitats, water quality enhancement, storm buffering, flood control, and economic values.

Coastal wetlands support an abundance of unique and diverse fish, wildlife, and plant species and are among the most productive natural ecosystems on earth. Louisiana coastal wetlands serve as spawning areas and nursery grounds for a myriad of shellfish and sport and commercial fish, including shrimp (*Penaeus* spp.), blue crabs (*Callinectes sapidus*), oysters (*Crassostrea virginica*), speckled trout (*Cynoscion nebulosus*), redfish (*Sciaenops ocellata*) and menhaden (*Brevoortia* spp.). These wetlands are also habitats for many fur-bearing mammals, including muskrat (*Ondatra zibethicus*), nutria (*Myocastor coypus*), otter (*Lutra canadensis*), mink (*Mustela vison*), beaver (*Castor canadensis*), opossum (*Didelphis virginiana*) and many others. Alligator (*Alligator mississippiensis*) farming and harvesting has become a valued product of Louisiana's coastal areas, with over 25,000 wild alligators being harvested each year (Coreil 1994). Historically this region has been recognized as one of the most important physiographic areas to migratory birds in North America (Gauthreaux 1971, Lowery 1974, Sprunt 1975, Gosselink et al. 1979, Moore et al. 1993). During spring and fall migration, millions of songbirds routinely use the gulf coast of Louisiana as an important stopover site before and after their trans-gulf migrations (Barrow et al. in press), and four to six million ducks and over 400,000 geese winter in Louisiana's coastal areas yearly (Bellrose 1976, USFWS 1984, Helmers 1992). Many species of invertebrates, amphibians, reptiles and mammals depend on wetlands for survival, including 11 threatened and endangered species which occur in coastal Louisiana.

Coastal wetlands also improve water quality and absorb the initial force of coastal storms and hurricanes (Mitsch and Gosselink 1993). Wetlands have been shown to remove organic and inorganic nutrients and toxins from the water that flows over them (Mitsch and Gosselink 1993) due to attributes such as reduction in water velocity, proximity of anaerobic and aerobic processes, high rates of vegetative productivity, diversity of decomposers and decomposition processes, high rates of sediment/water exchange, and accumulation of organic peat (Sather and Smith 1984). Vegetated coastal wetlands serve an extremely important surge protection function when tropical storms or hurricanes come ashore. Coastal wetlands absorb enormous amounts of water and dissipate wave energy that would otherwise do severe damage inland. During Hurricane Andrew, which crossed the Louisiana coastline on August 26, 1992, the Terrebonne basin system showed a decrease in storm surge from 2.0 m. (6.5 ft.) at the coast to about 0.15 m. (0.5 ft.) inland in the marshes east of the Atchafalaya River (Reed 1995). This ability of marshes to reduce storm surge provides vital flood protection for coastal communities.

Economically, Louisiana's wetlands contribute millions of dollars to the state economy. In Louisiana almost 900,000 sport fishing licenses are sold annually, and recreational fishing contributes $235 million yearly to Louisiana's economy (Cowan and Turner 1988). The sale of over 330,000 sport hunting licenses contributes $400 million to the State economy each year (Coreil 1994). Non-consumptive fish and wildlife activities exceeded $220 million in 1991 (Coreil 1994). Louisiana provides more domestic and commercial fishery landings than any other state in the nation, with an estimated 1.1 billion pounds of fish and shellfish harvested annually (USDOC 1996). The total value of Louisiana's commercial fisheries in 1991 was over $680 million. More than 40% of the nations wild fur harvest comes from Louisiana wetlands (Davis 1982). Cattle production in coastal areas exceeds $25 million per year (Coreil 1994). Alligator hides and meat from both wild and farm harvests exceeds $16 million annually (LCWCRTF 1997). In addition to fish and wildlife related revenues, coastal wetlands also protect natural gas production facilities worth $7.4 billion per year as well as oil and gas refineries which produce over $30 billion worth of petroleum products every year (LCWCRTF 1993). Louisiana's coastal wetlands contain 10 major navigation channels that move 400 million tons of waterborne commerce each year (USACE 1993), and coastal port facilities protected by the wetlands compose 25% of the nation's total exported commodities (LCWCRTF 1993).

Because Louisiana's coastal area is ecologically and economically among the world's richest estuarine regions it is of vital local, state, and national interest to protect this valuable resource. The Coast 2050 initiative is dedicated to this cause.

GENERAL ORGANIZATION/TEAM MEMBERS

Coast 2050 is a long range planning effort that aims at getting input on what the public wants their coastline to look like in the year 2050. The Coast 2050 team is a mix of local, state, and federal representatives, academia, and volunteers from the general public (Fig. 1). The Strategic Working Group (SWG) and Coastal Zone Management Working Group (CZMWG) were constituted by the Breaux Act Task force and the State Wetlands Authority. The SWG is composed of representatives from the U.S. Army Corps of Engineers (USACE), Environmental Protection Agency (EPA), Natural Resources Conservation Service (NRCS), U.S. Fish and Wildlife Service (USFWS), and National Marine Fisheries Service (NMFS), all Breaux Act agencies, as well as the Office of the Governor, Louisiana Department of Natural Resources (DNR), Division of Administration (DOA), Department of Wildlife and Fisheries (LDWF), Department of Environmental Quality (DEQ), Department of Transportation and Development (DOTD), and the State Soil and Water Conservation Commission of the Department of Agriculture and Forestry (SWCC), all state agencies. The SWG is responsible for overseeing the development of the Coast 2050 strategic plan.

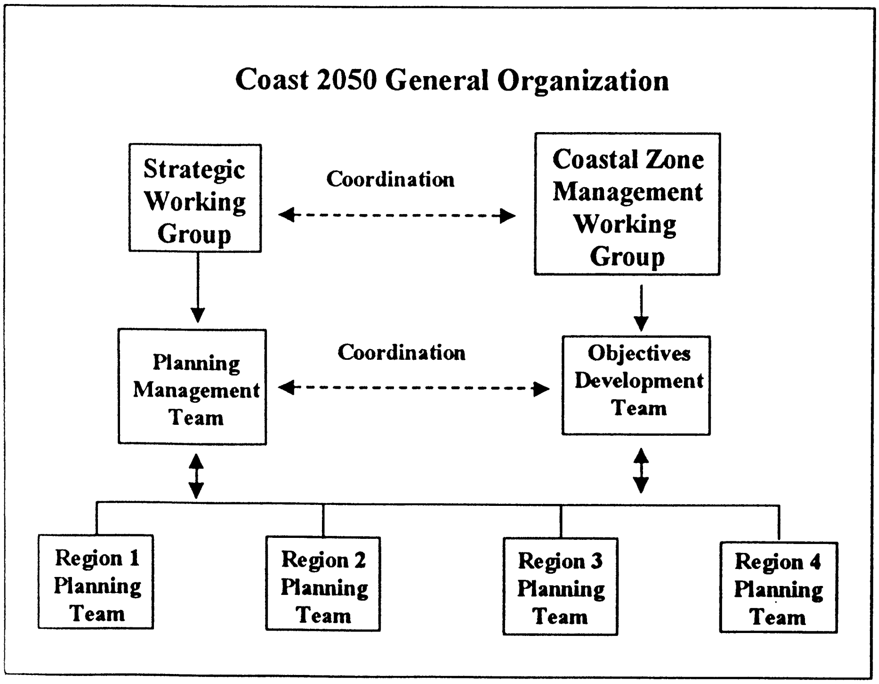


Fig. 1. General organization structure used in the Coast 2050 planning initiative.

The CZMWG is directed by the DNR and consists of government representatives from the 19 coastal parishes and parish Coastal Zone Management Advisory Committees (CZMAC). The CZWMG is responsible for coordinating public involvement into the Coast 2050 planning process.

The Planning Management Team (PMT) is responsible for authoring the Coast 2050 plan, and the Objectives Development Team is responsible for determining coastal use and resource objectives (i.e. what the public wants). Members of the ODT work closely with parish officials, local Coastal Zone Management and Louisiana Cooperative Extension Service (LCES) field agents, and the CZMACs, as well as attend town meetings to solicit information on what the citizens want their coastline to look like in the year 2050.

The four Regional Planning Teams (RPTs) were established to develop coastal strategies, review coastal use and resource objectives compiled by the ODT and meld them together. RPTs are composed of federal and state agency personnel, academic representatives from various Louisiana universities, parish government representatives, the LCES/LSU Seagrant staff, and volunteer participants from the general public. These teams are the regional experts that provide the PMT with technical information for their respective region and propose regional coastal strategies to the PMT.

THE COAST 2050 PLANNING PROCESS

The mission statement for Coast 2050 reads, "In partnership with the public develop, by December 22, 1998, a technically sound strategic plan to sustain coastal resources and provide an integrated multiple use approach to ecosystem management." (Coast 2050 Participation Guide 1997). To achieve this, Coast 2050 is attacking this problem from two sides and making coastal policy based on the common ground between them (Fig. 2). Together with the ODT, parish governments and the public identified coastal habitat and resource objectives that they felt were most important to them. These objectives reflect what the public wants its coast to look like in the year 2050. To assess ecosystem needs, the coast was divided into 4 broad geographic regions based on hydrologic region boundaries and geographic features (Fig.3).

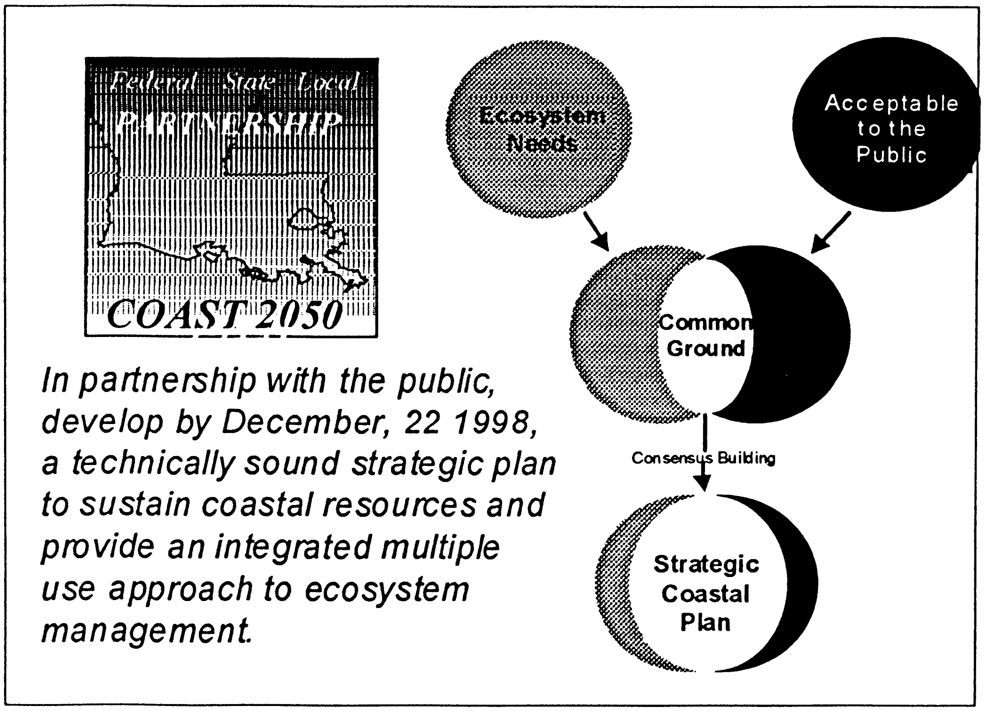


Fig. 2. Diagram showing how the ecosystem strategies and public objectives will be melded into the Coast 2050 plan.

Region 1 consists of the Pontchartrain basin. This basin occurs east of the Mississippi River, and north of the Mississippi River Gulf Outlet (MRGO). It includes the St. Bernard Delta lobe, Chandeleur Sound, and large expanses of contiguous bottomland hardwood and cypress swamp communities. Its marshes have been cut off from Mississippi River flow by the guide levees.

Region 2 includes the Breton Sound Basin, the Mississippi River Delta, and the Barataria Basin. It spans from the Mississippi River Gulf Outlet (MRGO) west to Bayou Lafourche. The marshes on the west side of the river have been totally severed from river flow by a series of guide levees and hurricane protection levees. On the east, the marshes north of Bohemia have also been severed from river flow. This area suffers from some of the highest land loss rates in the coastal zone (Fig. 4). Issues in Region 2 include balancing the ecological need for freshwater and sediment diversion into the basin with the needs of the sportfishing and oyster industry, as well as

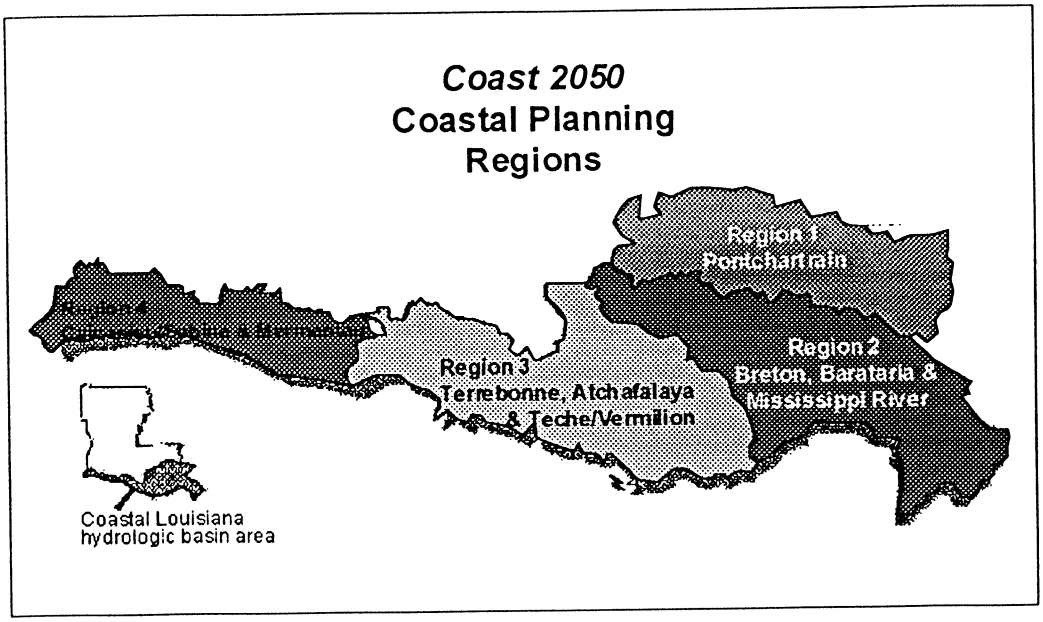


Fig.3. Coast 2050 coastal planning regions and the hydrologic basins contained in each

improving drainage and flood protection in the basin.

Region 3 includes the Terrebonne Basin, Atchafalaya Basin, and Teche Vermillion Basin. Terrebonne Basin is also suffering from the absence of natural riverine processes. The Atchafalaya and Teche Vermillion Basins are benefitting from sediments deposited by the Atchafalaya River and Wax Lake Outlets, which receive a portion of Mississippi River flow. Issues in the Terrebonne Basin include water quality, and the need for freshwater and sediment enrichment. Issues in Teche/Vermilion include edge erosion, the past destruction of ancient oyster reefs, water turbidity, and salinity.

Region 4 includes the Mermentau and Calcasieu Sabine Basins. The shorelines of this region are suffering from wave erosion and the lack of Mississippi and Atchafalaya River sediment available to be transported via long shore drift. Saltwater intrusion through the Calcasieu ship channel has caused much erosion in the Calcasieu/Sabine Basin. Timing, duration, and depth of water in the Mermentau area from upland runoff is another area of concern.

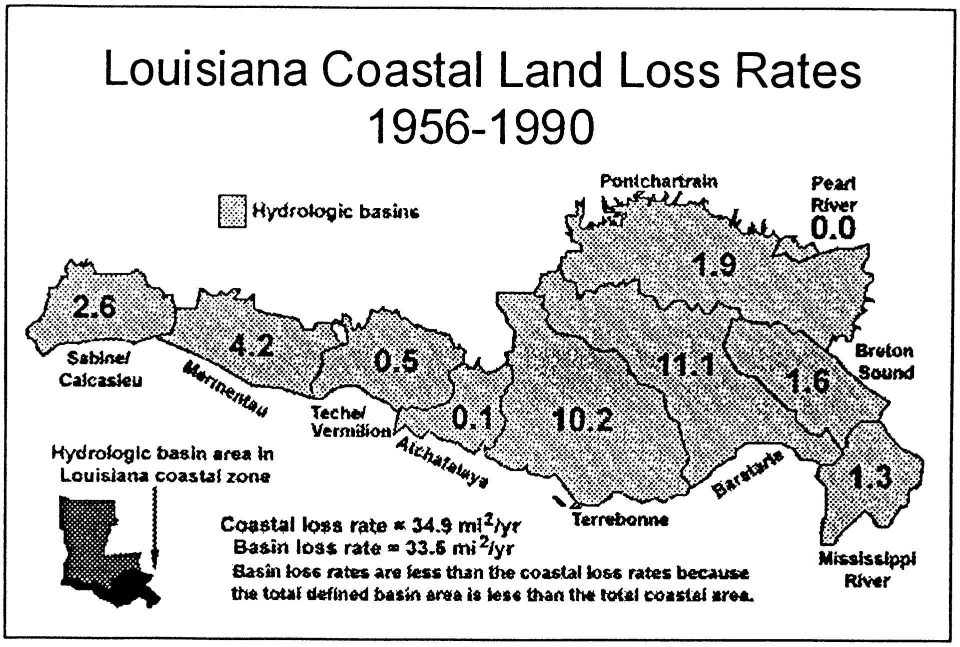


Fig. 4. Land loss rates in coastal Louisiana's nine hydrologic basins from 1956-1990. Data taken from Barras et al. (1994).

Regions were further divided into a total of 132 mapping units based on hydrologic boundaries and ecological features. Summaries of each mapping unit were developed that included land loss rate in the unit, habitat types, habitat shifts caused by land loss, fish and wildlife populations and trends, water supply and drainage information, strategies nominated by past plans, permitted activities and public and private infrastructure at risk from land loss.

Technical experts from state and federal governments, universities, private consulting firms, environmental special interest groups, and the general public (i.e. commercial fishermen, farmers, hunters, fishers, and residents) on each RPT worked together to develop strategies for their respective region that would be scientifically valid and beneficial to the ecosystem and still meet the public habitat and resource objectives for the region. Regional and local restoration strategies were compiled for individual mapping units or combinations of mapping units in each region.

Once the plan is written, state and federal authorities will try to establish the plan, with or without amendments, as unified coastal policy. This policy will form the basis of an amended Breaux Act Restoration Plan, Louisiana's Strategic Plan, and

Coastal Zone Management guidance. Coast 2050 is not about designing specific projects, however individual projects will come out of the strategies included in the plan for each region. Therefore Coast 2050 will be the coastal policy to be used when planning specific coastal restoration projects.

SIGNIFICANT ACHIEVEMENTS/DEADLINES

The Coast 2050 planning initiative has a deadline of December 1998; the whole process has to be completed in 18 months. As of this manuscript, much has been completed, and much is yet to be done (Fig. 5). In May of 1997 the SWG was

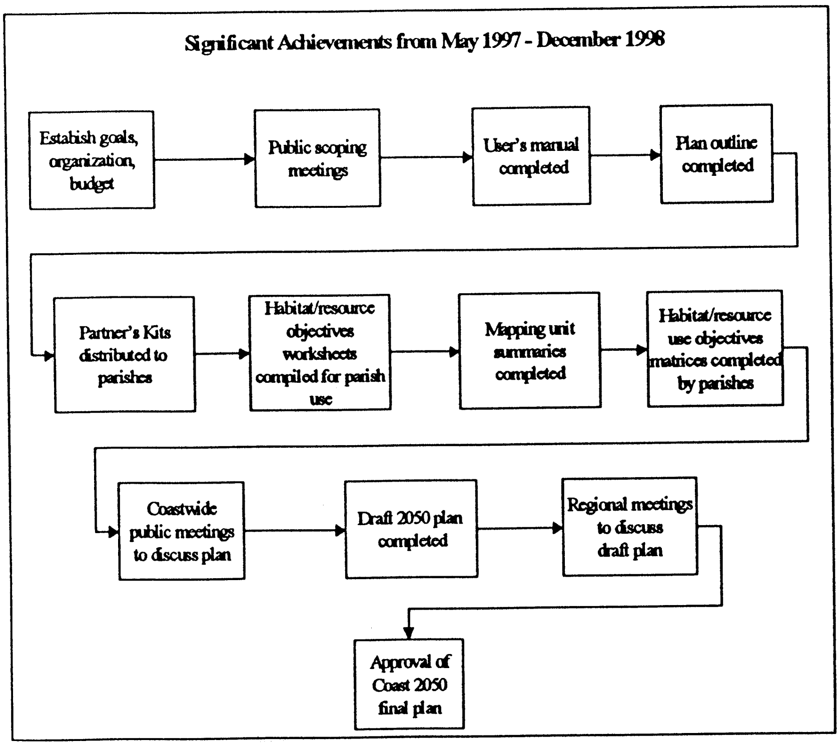


Fig. 5. Flowchart showing milestones in the 18 month Coast 2050 planning process. As of this manuscript, everything above the last two tiers has been completed. The deadline for the completion of everything in the last two tiers is December 22, 1998.

formed to begin the development of the Coast 2050 Initiative. Federal and state agency representatives established the mission statement, created a table of organization, time-line and approach, formulated a rough budget, and established geographical regions. Public scoping meetings were held throughout coastal Louisiana during July and August of 1997 to inform the public about Coast 2050 and receive their feedback. A participation guide which contained information on Coast 2050 organization, team member lists, points of contact, and a milestone table was completed and distributed coastwide in September of 1997. A plan outline was completed in October of 1997 and distributed coastwide.Partners kits which consisted of a slide show and other public relations materials were designed for parish representatives for use in their local presentations in December of 1997. In October RPTs began holding meetings to receive local input from the public and working toward strategy development. Mapping unit summaries for all units were completed in February 1998. The Planning Management Team (PMT) has been meeting monthly to receive and review input received from regional meetings and to discuss large-scale strategies. The SWG has been meeting quarterly to review the progress made to-date and to oversee the development of the strategic plan. In June 1998 a series of meetings will be held coastwide to give local parishes an update on the planning process and discuss the progress of the draft plan. By August of 1998 a draft strategic plan will be completed by the PMT. The PMT will hold a series of regional meetings in September 1998 to discuss the draft plan. Revisions resulting from input received during the regional meetings will be made in October 1998, and a leadership meeting involving all teams and working groups will also be held in that month. The final plan will then be available for approval by the CWPPRA Task Force and the State Wetlands Authority by the December 22, 1998 deadline.

CONCLUSION

The Coast 2050 plan is being developed in partnership with the public each step of the way. It is based on technically sound strategies that will sustain coastal resources. For the first time it will provide a coast-wide, integrated, multiple use approach to ecosystem management in Louisiana. Interested individuals can keep up with activities or peruse the participation guide or a slide show on Coast 2050 on our web page: www.lacoast.gov/Programs/2050/Index.htm.

LITERATURE CITED

Barras, J. A., P. E. Bourgeois, and L. R. Handley. 1994. Land loss in coastal Louisiana 1956-1990. National Biological Survey, National Wetlands

Research Center Open Report 94-01. 4 pp. 10 color plates.

Barrow, W. C., Jr., C. C. Chen, R. B. Hamilton, K. Ouchley, T. J. Spengler. 1998. Disruption and restoration of en-route habitat, a case study: the Chenier Plain. Studies in Avian Biology.

Bellrose, F. C. 1976. Ducks, geese, and swans of North America. Wildlife Management Institute.

Coreil, P. D. 1994. Wetland functions and values in Louisiana. Baton Rouge: Louisiana State University Cooperative Extension Service, Pub. 2519. 11 pp.

Cowan, J., Jr., and R. Turner. 1988. Modeling wetland loss in coastal Louisiana: Geology, geography, and human modifications. Environmental Management 12(6):827-838.

Davis, D. 1982. Economic and cultural consequences of land loss in Louisiana, *in* Proceedings of the conference on coastal erosion and wetland modification in Louisiana: causes, consequences, and options. Baton Rouge, LA. FWS/OBS-82-59.

Gauthreaux, S. A., Jr. 1971. A radar and direct visual study of passerine spring migration in southern Louisiana. Auk 88:343-365.

Gosselink, J. G., C. L. Cordes, and J. W. Parsons. 1979. An ecological characterization study of the Chenier Plain coastal ecosystem of Louisiana and Texas. 3 vols. U.S. Fish and Wildlife Service. FWS/OBS-78/9 through 78/11.

Gosselink, J. G. 1984. The ecology of delta marshes of coastal Louisiana: a community profile. U.S. Fish and Wildlife Service. FWS/OBS-84/09. 134 pp.

Helmers, D. L. 1992. Shorebird management manual. Manomet, MA: Western Hemisphere Shorebird Reserve Network. 58 pp.

Kolb, C. R. and J. R. Van Lopik. 1958. Geology of the Mississippi River deltaic plain, southeastern Louisiana. Technical Reports 3-483 and 3-484. Vicksburg, MS: U.S. Army Engineers Waterway Experiment Station.

Lindstedt, D. M., L. L. Nunn, J. C. Holmes, Jr., and E. E. Willis. 1991. History of oil and gas development in coastal Louisiana. Resource Information Series No.7. Baton Rouge: Louisiana Geological Survey. 74 pp. plus appendices.

Louisiana Coastal Wetlands Conservation and Restoration Task Force. 1993. Louisiana Coastal Wetlands Restoration Plan: Main Report, Environmental Impact Statement and Appendices. Baton Rouge: U.S. Army Corps of Engineers, U.S. Department of Agriculture, U.S. Department of Commerce, U.S. Department of the Interior, U.S. Environmental Protection Agency, and Louisiana Department of Natural Resources.

_____ 1997. The 1997 evaluation report to the U.S. Congress on the effectiveness of Louisiana coastal wetland restoration projects. Baton Rouge: U.S. Army Corps of Engineers, U.S. Department of Agriculture, U.S. Department of Commerce, U.S. Department of the Interior, U.S. Environmental Protection Agency and Louisiana Department of Natural Resources. 76 pp.

Lowery, G. H., Jr. 1974. Louisiana Birds. LSU Press, Baton Rouge, LA.

Mitsch, W. J. and J. G. Gosselink. 1993. Wetlands (Second edition). New York, NY. Van Norstrand Reinhold. 722 pp.

Moore, F. R., S. A. Gauthreaux, P. Kerlinger, and T. R. Simmons. 1993. Stopover habitat: management implications and guidelines. In D. M. Finch and P. W. Stangel, eds. Status and management of neotropical migratory birds., pp.58-69. USDA Forest Service. RM-229.

Reed, D. J., (ed). 1995. Status and historical trends of hydrologic modification, reduction in sediment availability, and habitat loss/modification in the Barataria and Terrebonne estuarine system. BTNEP Publication No. 20, Barataria-Terrebonne National Estuary Program, Thibidaux, Louisiana. 338 pp. plus Appendices.

Sather, J. H. and R. D. Smith. 1984. An overview of major wetland functions and values. Western Energy and Land Use Team, U.S. Fish and Wildlife Service. FWS/OBS-84/18. Washington D.C. 68 pp.

Sprunt, A. 1975. Habitat management implications of migration. In Proceedings of the Symposium on Management of Forest and Range Habitats for Non-game Birds., pp 81-86. USDA Forest Service, GTR WO-1.

U.S. Army Corps of Engineers. 1993. Water-borne commerce of the United States, calendar year 1993. Part 5 - National summaries. Fort Belvoir, VA: Water Resources Support Center, U.S. Army Corps of Engineers.

U.S. Department of Commerce. 1996. Fishery statistics. National Oceanic and Atmospheric Administration. National Marine Fisheries Service.

U.S. Department of the Interior, Fish and Wildlife Service. 1984. Vanishing Gulf coast wetlands. U.S. Government Printing Office.

Zobrist, E. C., T. Osborn, R. Ruebsamen. 1995. Hydrologic restoration of Point Au Fer Island, Louisiana: National Marine Fisheries Service efforts to restore wetland habitat. In Proceedings of the Coastal Zone '95. B. L. Edge ed., pp.31-32.

THE USE OF A HEP EVALUATION TO ESTABLISH RESTORATION NEEDS

Kevin C. Owen, Iroquois Pipeline Operating Company,
One Corporate Drive, Suite 600 Shelton, CT 06484

ABSTRACT

The accurate assessment of the impacts on natural resources due to construction activities is important in defining the needs and objectives of environmental restoration projects. The U.S. Fish and Wildlife Service developed the Habitat Evaluation Procedure (HEP) methodology to assist in predicting environmental impacts and establishing restoration goals. The HEP assessments include mapping habitat types and estimating the value of the habitats for evaluation species. The predicted changes in habitat acreage and value are then used to identify the type and magnitude of environmental mitigation required.

In 1987, the HEP methodology was used for a proposed hydroelectric project on the Susquehanna River in Harrisburg, Pennsylvania. The assessment examined potential impacts on islands and aquatic vegetation beds (AVB) in the river and predicted habitat restoration needs. Based upon the vegetation, the islands were assumed to be wetlands and were divided into scrub - shrub and forested wetlands. For the AVB, the habitat values were quite consistent between sample locations for the different evaluation species; with there being no difference between locations for some of the species. For the forested and scrub - shrub wetlands, the habitat values varied widely, with no recognizable trends. This variance weakened the subsequent impact prediction and restoration planning.

In 1991 and 1992, the islands were mapped using the Corps of Engineers (COE) wetland delineation procedures. The forests, which had previously been mapped as wetlands, were shown to consist of both wetland and upland habitat. In 1993, the HEP evaluation for the forested habitat was repeated using the wetland and upland habitat mapping based upon the COE procedures. The variability that was observed for forested wetlands in the 1987 study was found to be caused by the presence of both upland and wetland habitat on the islands. With the corrected habitat mapping, the habitat values in the 1993 study were more consistent for all evaluation species for the forested habitat type.

Through the proper mapping of habitats, the prediction of potential impacts on the islands and AVB was more reliable. The restoration that was required for each

impacted habitat type could then be identified. This study indicates the value of proper habitat identification and mapping in establishing restoration requirements and goals.

INTRODUCTION

The Susquehanna River passes over a low-level dam that was constructed in the early 1900's in the City of Harrisburg, Pennsylvania (the City). The City, in an effort to increase the area of the river available for certain types of water related recreation, has proposed to construct a higher dam upstream of the original dam. In 1987, the City submitted an application to the Federal Energy Regulatory Commission (the FERC) to construct the proposed dam with hydropower generating capacity (Acres, 1987).

Prior to the submittal of the FERC application, the City consulted with the regulatory and resource agencies reviewing the proposed project. In response to recommendations from these review agencies, the existing habitat conditions and potential impacts on wildlife were assessed using the Pennsylvania Modified Habitat Evaluation Procedure (PAM HEP). The Habitat Evaluation Procedure (HEP) methodology was developed by the United States Fish and Wildlife Service (USFWS) to assist in predicting environmental impacts and establishing restoration goals. HEP assessments include mapping habitat types and estimating the value of the habitats for evaluation species. The predicted changes in habitat acreage and value are then used to identify the type and magnitude of environmental mitigation required. The PAM HEP is a simplified version of the Habitat Evaluation Procedure (HEP) that excludes an assessment of fish habitat (Palmer et al., 1985).

The HEP evaluation (Acres, 1987; 1988) was conducted for forested habitat, scrub-shrub wetlands and aquatic vegetation beds (AVB). Subsequent to the initial habitat mapping of the project area, the Corps of Engineers (COE) procedures to delineate wetlands were issued. These procedures require the presence of hydric soils, wetlands hydrology, and hydrophytic vegetation to define wetlands (U.S. Army Corps of Engineers, 1987).

In 1991 and 1992, the habitat areas were reclassified using the COE procedures to delineate wetlands (Acres, 1993). The HEP evaluation was repeated in 1993 based upon the revised vegetative cover mapping. The results of the 1987 and 1993 HEP evaluations are reviewed in this paper. This review is intended to illustrate the value of assessment models in identifying project impacts and establishing restoration needs. The review also suggests these models can be helpful in highlighting shortcomings of the basic data collection and interpretation that should be addressed.

The evaluations for the AVB will be reviewed to illustrate the salient components of the HEP evaluation. However, this paper will concentrate on the HEP evaluations of the forested habitat in the project area. The results of the evaluation of the scrub-shrub wetlands are not included in the scope of this paper.

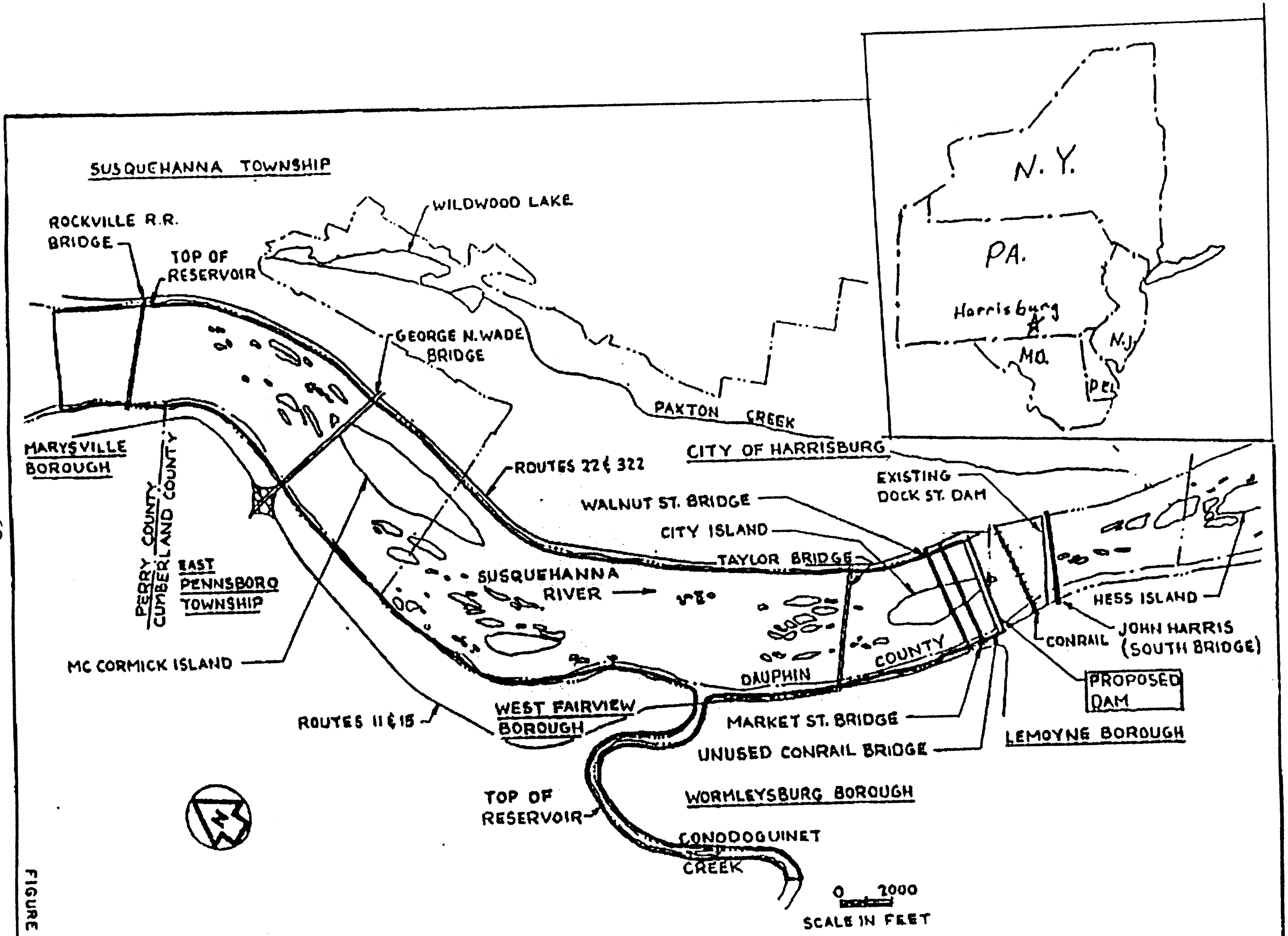
STUDY AREA

The area to be impacted by the proposed hydropower project is illustrated in Figure 1. The proposed dam and resulting reservoir would have impacted approximately 14 km of the Susquehanna River in the vicinity of Harrisburg, Pennsylvania. The project area extended from the Rockville Railroad Bridge upstream to the existing Dock Street Dam downstream (Figure 1). Approximately 5 km of a tributary of the river, Conodoguinet Creek, would also have experienced increased water levels due to the dam. The project area included over 60 vegetated islands, channel bars and, AVB. As indicated in Figure 1, the islands and channel bars were scattered throughout the project area and varied dramatically in size. These islands, channel bars, and AVB were used in the HEP evaluation reported in this paper.

MATERIALS AND METHODS

Prior to the initiation of the 1987 HEP project, the types and locations of the vegetated habitats within the project area were mapped. A base map of the project area was generated from survey data and aerial photographs. Aerial photo interpretation was used to map the distribution of the different habitat types. Vegetation surveys were then conducted in the field to verify the mapping and to identify the dominant plant species within each habitat type. For the purpose of the mapping, areas that had a prevalence of hydrophytic vegetation were assumed to be wetlands (Acres, 1987).

The HEP evaluation completed in 1987 was conducted following the procedures of Palmer et al., (1985). A HEP team was established consisting of one representative each from the USFWS, the Pennsylvania Game Commission (PGC), and an environmental consulting firm contracted by the City. For each of the previously mapped habitat types (i.e., forested wetlands, scrub-shrub wetlands, and AVB), the HEP team identified guilds of animals. A guild is a group of animal species that use the habitat resources in a similar manner. For each guild, an evaluation species was chosen for which an habitat suitability index (HSI) model was either published or for which there was so much published information that an HSI model could be readily developed (Acres, 1987; 1988).



PROJECT LOCATION MAP

FIGURE 1

Once the evaluation species and HSI models were established, the HEP team measured the required variables in the field. For each habitat type, five randomly selected locations were sampled. From the equations given in the HSI models, the HSI score for each evaluation species was calculated from the variables for each sampling location and the HSI scores were averaged. The HSI score was multiplied by the acreage of each habitat type determined from the vegetation cover type maps to establish the habitat units (HU) for the existing, or baseline, condition (Acres, 1987; 1988). The existing or baseline condition is referred to as the Target Year Baseline or TYB (Palmer et al., 1985).

Maps were prepared projecting the change in the vegetative cover types in the study area that would have been caused by the proposed project. These impacts would have resulted from the construction of the proposed dam and the increased area and depth of the reservoir. The HSI scores were multiplied by the acreage of the habitat types to establish the HU for the post-project construction condition. The City proposed certain mitigation measures that would have altered the post-project construction acreage. The HSI scores were then multiplied by the acreage projects for each habitat type after the mitigation was completed to derive the HU for the mitigated condition (Acres, 1987; 1988).

The post-project construction condition is referred to as the Target Year Construction or TYC. The conditions after the proposed mitigation is in place and fully functional are referred to as the Target Year Mitigation or TYM (Palmer et al., 1985). The USFWS and the PGC, because of the opposition of these agencies to the project, would not participate in, or approve, the HU calculations for the TYC and TYM.

In 1991 and 1992, the habitat areas were reclassified using the COE procedures to delineate wetlands (Acres, 1993). Sampling transacts were established on each island in the project area and data on soils, hydrology, and dominant vegetation were recorded at each sampling location along the transacts. The length and number of the transacts and the number of sampling locations along each transect varied with the size and biological complexity of the island or channel bar. With this revised mapping, the acreage of the different habitat types under the existing, post-project construction, and mitigated conditions were recalculated.

The HEP evaluation was repeated in 1993 based upon the revised vegetative cover mapping. The revised habitat acreage was used with the HSI scores determined in the 1987 study to calculate the HU for TYB, TYC, and TYM (Acres, 1993). The scope of this paper is limited to the HEP evaluations for the AVB and forested habitats.

RESULTS

The total acreage of the vegetated habitats are listed in Tables 1 and 2. Table I lists the areas calculated from the 1986 and 1987 mapping. The acreage calculated from the subsequent revised mapping in 1991 and 1992 is presented in Table 2. Since the HEP evaluation used acreage as the unit of area, metric units are not used in these, or subsequent, tables.

Table 1. Cover Type Acreage at Different Target Years - 1987 Mapping (Acres, 1988)

Cover Type	Baseline (TYB) Acreage	Post-Construction (TYC) Acreage	Mitigation In Place (TYM) Acreage
Aquatic Beds (AVB)	127	139	139
Scrub-Shrub Wetlands	49	6 (Original) 40 (Man-made)	60
Forested Wetlands	248	217	243

Table 2. Cover Type Acreage at Different Target Years - 1992 Mapping (Acres, 1993)

Cover Type	Baseline (TYB) Acreage	Post-Construction (TYC) Acreage	Mitigation In Place (TYM) Acreage
Aquatic Beds (AVB)	127	139	139
Scrub-Shrub Wetlands	49	6 (Original) 40 (Man-made)	60
Forested Wetlands	59	65	92
Upland Forests	197	127	127

The mapping reported by Acres (1987) used vegetation as the sole criterion to identify wetlands. Based upon a consideration of only the dominant vegetation, all habitats in the study area were classified as wetlands (Table 1). Using the COE (1987) wetland procedures, upland forested habitat was found to be present in the project area (Table 2).

The evaluation species selected by the HEP team for the AVB were wood duck, green backed heron, and great egret (Acres, 1988). The HSI scores for these species determined by the HEP team for the AVB are presented in Table 3.

Table 3. AVB - Mean HSI Score by Evaluation Species (Acres, 1988)

Sample Site	Wood Duck HSI Score	Green-backed Heron HSI Score	Great Egret HSI Score
Area 1	0.6	0.7	0.3
Area 2	0.7	0.7	0.3
Area 3	0.5	0.6	0.3
Area 4	0.6	0.9	0.3
Area 5	0.7	0.9	0.3
Mean HSI Score	0.6	0.8	0.3

The HU for the AVB calculated by the consultants for the City are presented in Table 4.

Table 4. Aquatic Vegetation Beds (AVB) - HEP Assessment (Acres, 1988)

Evaluation Species	TYB HU	TYC HU (change*)	TYM HU (change*)
Wood Duck	76	83 (+7)	83 (+7)
Green-backed Heron	102	111 (+9)	111 (+9)
Great Egret	38	42 (+4)	42 (+4)
Project Total	216	236 (+20)	236 (+20)

* change from the TYB HU value

As shown in Table 3, the HSI scores were quite consistent for each species among all the sampling locations. The low variability in the HSI scores suggests the AVB represent relatively homogeneous habitat conditions across the entire study area. This consistency tends to make the evaluation of the changes in the habitat conditions due to the project construction impacts and the proposed mitigation more reliable (Table 4). As indicated in Table 4, due to the expected increase in the acreage of the AVB, the value or availability of this habitat type to wildlife would be increased by the proposed project.

The evaluation species selected for the forested wetlands for the forested wetlands in the 1987 HEP assessment were American toad, great egret, and house wren (Acres, 1988). The HSI scores and HU determined for the forested wetlands are presented in Tables 5 and 6, respectively.

Table 5. 1987 Forested Wetlands - Mean HSI Score by Evaluation Species (Acres, 1988)

Sample Site	American Toad	Great Egret	House Wren
Area 1	0	0.8	0.4
Area 2	0.1	1.0	0.2
Area 3	0.5	1.0	0.1
Area 4	0.4	0.4	0.4
Area 5	0.8	0	0.6
Mean HSI Score	0.4	0.6	0.3

Unlike the HSI scores for the AVB, the HSI scores for the forested wetlands had a greater degree of variability. As shown on Table 5, at four sampling locations (Areas 1,2,3, and 4, the HSI scores for the American toad and house wren were relatively low (i.e. between 0 and 0.4, with one value being 0.5) and the score for the great egret was relatively high (0.8 to 1.0). For sampling Area 5, the reverse was true. The HSI scores for the American toad and house wren were relatively high (0.8 and 0.6, respectively) and the HSI score for the great egret was a zero.

Table 6. 1987 Forested Wetlands - HEP Assessment (Acres, 1988)

Evaluation Species	TYB HU	TYC HU (change*)	TYM HU (change*)
American Toad	99	87 (-12)	97 (-2)
Great Egret	149	130 (-19)	146 (-3)
House Wren	74	65 (-9)	73 (-1)
Project Total	322	282 (-40)	316 (-6)

* change from the TYB HU value

After the vegetative cover types were mapped in 1991 and 1992 using the U.S. Army Corps of Engineers (1987) wetlands delineation procedures, the HEP assessment was repeated (Acres, 1993). The sampling areas were divided between the upland and wetland forested locations. The HSI scores for the upland and wetland forest locations are presented in Tables 7 and 8, respectively.

Table 7. 1993 Upland Forests - Mean HSI Score by Evaluation Species (Acres, 1993)

Sample Site	American Toad	Great Egret	House Wren
Area 1	0	0.8	0.4
Area 2	0.1	1.0	0.2
Area 3	0.5	1.0	0.1
Area 4	0.4	0.4	0.4
Mean HSI Score	0.3	0.8	0.3

Table 8. 1993 Forested Wetlands - Mean HSI Score by Evaluation Species (Acres, 1993)

Sample Site	American Toad	Great Egret	House Wren
Area 5	0.8	0	0.6
Mean HSI Score	0.8	0	0.6

As indicated in Table 7, there was still a degree of variability in the HSI scores for the forested habitats. However, the recognition of the two types of forested habitat helped to account for some of the variability noted in 1987 (Table 5).

To indicate the significance of this remapping, the HEP evaluations were completed using the 1987 HSI scores and the acreage of the habitat types calculated from the 1991 and 1992 vegetation mapping. The results of this evaluation are presented in Table 9 for the upland forests and Table 10 for the forested wetlands.

Table 9. 1993 Upland Forests - HEP Assessment

Evaluation Species	TYB HU	TYC HU (change*)	TYM HU (change*)
American Toad	59	38 (-21)	38 (-21)
Great Egret	158	102 (-56)	102 (-56)
House Wren	59	38 (-21)	38 (-21)
Project Total	276	178 (-98)	178 (-98)

* change from the TYB HU value

Table 10. 1993 Forested Wetlands - HEP Assessment

Evaluation Species	TYB HU	TYC HU (change*)	TYM HU (change*)
American Toad	47	52 (+5)	52 (+5)
Great Egret	0	0	0
House Wren	35	39 (+4)	39 (+4)
Project Total	82	91 (+9)	91 (+9)

* change from the TYB HU value

The 1993 HEP evaluation suggested that the forested wetlands would probably be enhanced by the project (Table 10). The major potential adverse impacts would be experienced by the upland forests (Table 9).

DISCUSSION

In a HEP evaluation, like most impact assessment models, the correct mapping of the habitat types is important in developing model inputs, as well as achieving reliable results. As suggested by this paper, the proper mapping of habitats is vital to the accurate prediction of potential impacts of proposed construction projects. The accurate assessment of construction impacts on natural resources is important in defining the needs and objectives of environmental restoration projects. The restoration required for each impacted habitat type can then be more reliably identified through the use of the assessment model.

The variability in the HSI scores for the forested habitat reported in 1987 (Table 5) could have been attributed to a number of potential causes: selection of inappropriate animal guilds and evaluation species, insufficient number of sampling areas to adequately characterize the habitat, improper mapping of the habitat type and the failure to recognize both upland and forested wetlands.

The selection of the species guilds to be assessed and the evaluation species to be used to represent each guild is dependent upon the habitat types identified. Wading birds were known to feed in the AVB and nest in the forests on the islands in the project area. The great egret was chosen as the evaluation species for this guild nesting in the forests. However, the HSI scores for the great egret covered the entire scale for the HSI scores from zero to one (Tables 5, 7, and 8). This extreme variability for the HSI scores suggest that the great egret may not have been a proper evaluation species for the forested habitats.

The extreme variability in HSI scores in Table 5 suggests a representative number of areas was not sampled to adequately characterize the forested habitat. A much larger number of areas would have to be sampled to compensate for this variability in the HSI scores. From a statistical perspective, 15 sample areas would have been needed to estimate the population mean of the great egret HSI to within 0.2 for a 95% confidence interval in the Students' t-test. When the vegetation mapping was revised in 1991 and 1992, the inadequate sample size was made more apparent. As shown in Tables 7 and 8, there were only four samples in the upland forests and only one sample in a forested wetlands. The sampling protocol established for this HEP evaluation (i.e. five samples per habitat type) had been violated.

Based upon this HEP evaluation and other considerations, the City proposed to construct forested wetlands habitat in the river to mitigate the potential project impacts (Acres, 1987; 1988). As indicated in Table 6, the City in 1987, in proposing to create additional forested wetlands, appeared to be offering an appropriate mitigation package for the project. The mitigation proposed in 1987 would have compensated for most of the potential impacts identified on the forested wetlands.

However, the variability in the HSI scores (Table 5) indicated a lack of homogeneity in the forested habitat. The reliability of the HU calculations, and the resulting assessment of the scale of mitigation required (Table 6), was also weakened. The City, in 1987, did not have a method in place to distinguish the areas of low habitat value from the areas of higher habitat value. There was no way to demonstrate that areas of high habitat value would not be lost during the construction of the project and replaced by areas with low value habitat developed as mitigation (or vice-versa).

When the vegetative cover types were remapped in 1991 and 1992, both upland forests and forested wetlands were shown to be present (Acres, 1993). The presence of both upland and wetland forests suggested a reason for the variability in the HSI scores determined in 1987. Sampling Areas 1, 2, 3, and 4, which had relatively similar HSI scores, were found to be in upland forests (Table 7). Sampling Area 5, with the anomalous HSI scores, was located in a forested wetland (Table 8).

As shown in Tables 9 and 10, both the upland forests and the forested wetlands were under-sampled, as was also suggested by the variability in the HSI scores for the forested habitat determined in 1987 (Table 5). Despite this lack of data, as suggested in Table 10, the postconstruction conditions would have forested wetlands habitat that was, at the least, comparable to the existing conditions. As indicated in Table 10, the project would have actually increased the habitat value of the forested wetlands by increasing the acreage of this habitat type (Table 2).

Conversely, as indicated in Table 9, the upland forests would be significantly reduced by the proposed project. The mitigation measures proposed by the City would not have provided compensation for this reduction in upland forests. In fact, no mitigation had been offered to offset the potential impacts of the proposed project on this habitat type.

With the proper mapping of the habitat types, the actual impacts of the project could be more accurately identified. The resulting HEP evaluation then revealed that the mitigation offered by the City (i.e. creating additional forested wetland habitat) would not have compensated for the actual project impacts, the loss of upland forested habitat values. This study indicates the value of proper habitat identification and mapping and of the use of impact assessment models in establishing restoration requirements and goals.

ACKNOWLEDGMENT

The HEP evaluation and field work was conducted by Acres International Corporation (Acres) under contract to the City of Harrisburg, Pennsylvania. The project location map used to generate Figure 1 and the picture slides used by the author in the conference presentation were supplied by Acres. The opinions and conclusions expressed in this paper are those of the author and do not necessarily reflect the opinions and positions of either Acres or the City of Harrisburg.

LITERATURE CITED

Acres International Corporation. 1987. Before the Federal Energy Regulatory Commission, Application for License For Major Unconstructed Project, Dock Street Dam and Lock Project. Vol. 3, Exhibit E.3, pp. E.3.9-13 - E.3.9-14.

Acres International Corporation. 1988. Before the Federal Energy Regulatory Commission, Application for License For Major Unconstructed Project, Dock

Street Dam and Lock Project. Vol. 12, Response to FERC Further Additional Information Request No. 5, 3 pages plus tables.

Acres International Corporation. 1993. Harrisburg Hydroelectric Project - Water Level Alternatives. Tech. report prepared for the City of Harrisburg, PA. pp. 3-9 - 3- 1 0.

Palmer, J. H, M. T. Chezik, R. D. Heaslip, G. A. Rogalsky, D. J. Putnam, R.W. McCoy, and J. A. Arway, 1985. Pennsylvania Modified 1980 Habitat Evaluation Procedure Instruction Manual. Pennsylvania Game Commission.

U.S. Army Corps of Engineers. 1987. Corps of Engineers Wetlands Delineation Manual.Tech. Report Y-87-1. 100 pages plus appendices.

THE RE-MITIGATION OF COUNTY LINE ROAD (POLK COUNTY, FLORIDA)

Kevin M. Connor, MS - BCI Engineers & Scientists, Inc.
Art Wade - Polk County

ABSTRACT

County Line Road in west Polk County was historically segmented and did not provide the continuous connection between Interstate 4 and State Road 60 that exists today. In 1988, SWFWMD and FDEP issued permits approving the connection. These permits authorized 0.73 ha (1.8 acres) of wetland impacts along Segment 2 and 1.30 ha (3.2 acres) of wetland impacts along Segment 3. Pursuant to permit conditions, two mitigation areas were to be created: a 1.46 ha (3.6 acre) site for Segment 2 and a 3.85 ha (9.5 acre) site for Segment 3. However, initial construction resulted in only 0.49 ha (1.21 acres) of acceptable wetlands being created for Segment 2 and 0.51 ha (1.26 acres) for Segment 3. Subsequently, the areas were deemed unacceptable by SWFWMD and the County was subject to enforcement action.

Polk County contracted BCI and together with SWFWMD a mitigation plan was agreed upon which satisfied SWFWMD's requirements. The plan consisted of additional wetland creation within the two existing mitigation areas, such that a 1:1 ratio of compensation was achieved on-site, with the remainder being attained at two off-site mitigation areas (Sutton Road East and West). While most of the credit was accomplished through wetland creation, credit was also obtained through wetland enhancement and upland preservation and enhancement.

Many engineering challenges were encountered while designing the four mitigation areas. Technical challenges that were addressed include ingress/egress from the existing sites to avoid new impacts, and hard clays below the surface. Construction of the mitigation sites was completed in April 1998 with as-built submittal in May 1998.

INTRODUCTION

County Line Road is located in west Polk County along the Polk-Hillsborough County border. Historically, it has been a segmented road and did not provide a continuous connection between Interstate 4 and State Road 60. In the 1980s, Polk

County planned and designed the road to be the continuous connection that it is today. Currently, County Line Road is two lanes, but as part of this project, right-of-way was acquired for future expansion and four laning.

In 1988, the Florida Department of Environmental Protection (FDEP) and Southwest Florida Water Management District (SWFWMD) issued permits approving the road construction and associated wetland impacts. Segment 2 of the road impacted 0.73 ha (1.8 acres) of wetlands and Segment 3 impacted 1.30 ha (3.2 acres) of wetlands. As mitigation for these impacts, Polk County was required to construct two mitigation sites. One site is located along Segment 2 north of Medulla Road adjacent to Hamilton Branch. To compensate for the 0.73 ha (1.8 acres) of impacts, 1.46 ha (3.6 acre) of wetlands were to be created (2:1 ratio). The second mitigation site is located along Segment 3, north of Ewell Road adjacent to English Creek. To compensate for the 1.30 ha (3.2 acres) of impacts, 3.85 ha (9.5 acre) of wetlands were to be created (3:1 ratio). The road and mitigation site construction was completed in 1990.

In 1993, SWFWMD deemed the mitigation sites unacceptable and Polk County was subjected to enforcement actions. Polk County contracted BCI Engineers & Scientists, Inc. (formerly Bromwell & Carrier, Inc.) to assist with bringing the mitigation sites into compliance.

MATERIALS AND METHODS

BCI developed a plan to evaluate the mitigation sites to determine why they were unsuccessful and how to remedy the situation. The site evaluations consisted of three phases:

1) Install monitoring wells to assess groundwater elevations;
2) Perform soil borings; and
3) Determine the amount of acceptable wetlands at each site.

RESULTS

As a result of the evaluation, BCI determined several key items about the sites:

- Segment 2 had 0.49 ha (1.21 acres) of acceptable wetlands
- Segment 3 had 0.51 ha (1.26 acres) of acceptable wetlands
- Nuisance/exotic vegetation was abundant
- Monitoring wells showed permanent ground water was below existing ground elevations

- Hard clay was present 30 - 45 cm (12 - 18 inches) below a sandy surface layer at Segment 3

DISCUSSION

With these facts, BCI began developing strategies to provide the required mitigation. Several options were investigated including starting over and re-constructing each site, starting over at new sites owned by Polk County, constructing around the existing wetlands, and use of off-site locations for any needed additional area.

After negotiations with SWFWMD, a comprehensive mitigation plan was agreed upon. The mitigation plan called for:

- Credit for wetlands already created
- Creation of additional wetlands on-site (minimum 1:1 replacement on-site)
- Off-site wetland creation plus enhancement
- Upland preservation and enhancement

To achieve the 1.46 ha (3.6 acres) of mitigation for Segment 2, 0.49 ha (1.21 acres) of credit was received for the wetlands already created, an additional 0.26 ha (0.63 acres) would be created at Segment 2, and 0.09 ha (0.21 acres) of uplands would be preserved/enhanced at a 7:1 ratio, giving 0.01 ha (0.03 acres) of credit. The remainder of credit would be achieved at an off-site location.

Polk County owned two parcels of land on Sutton Road, approximately eight kilometers (5 miles) north of the County Line Road mitigation sites. The Sutton Road West parcel is approximately two-thirds uplands and one-third wetlands, most of which are associated with a small creek channel. To achieve the needed mitigation area, Polk County proposed to create additional wetlands adjacent to the creek floodplain. This also allowed greater floodplain storage for the proposed recreational park at the site. The remaining 0.70 ha (1.73 acres) of mitigation credit for Segment 2 was achieved at Sutton Road West.

To achieve the 3.85 ha (9.50 acres) of mitigation for Segment 3, 0.51 ha (1.26 acres) of credit was received for the wetlands already created, an additional 1.54 ha (3.81 acres) would be created and Segment 3 and 0.17 ha (0.42 acres) of uplands would be preserved/enhanced at a 9:1 ratio, giving 0.02 ha (0.05 acres) of credit. The remainder of credit would be achieved at off-site locations. An additional 0.30 ha (0.74 acres) would be created at Sutton Road West with the remainder at Sutton Road East.

The Sutton Road East property consisted of a deep borrow pit with steep slopes and an abandoned house with a yard of bahia grass and slash pine trees. The site supported 0.05 ha (0.12 acres) of wetlands along the fringe of the borrow pit. Polk County proposed to create 1.41 ha (3.48 acres) of wetlands at the site, enhance the existing 0.05 ha (0.12 acres) at a 4:1 ratio for 0.01 ha (0.03 acres) credit, and preserve/enhance 0.54 ha (1.33 acres) of uplands at a 9:1 ratio for 0.06 ha (0.14 acres) of credit.

In order to fill in the borrow pit at Sutton Road East, 12,225 cubic meters (16,000 cubic yards) of fill was needed. Most of the fill, approximately 8,400 cubic meters (11,000 cubic yards), was obtained from cuttings at Sutton Road East. An additional 3,055 cubic meters (4,000 cubic yards) was obtained from cuttings at Sutton Road West, and 770 cubic meters (1,000 cubic yards) was obtained from cuttings at Segment 3.

Excavation was a major component of the construction process and filling in the borrow pit provided a good disposal location of the excavated material. One of the biggest problems found during the site evaluations was that the sites lacked proper wetland hydrology. The sites were supposed to receive much of their hydrology from seepage. This did not happen because after construction of the road and its associated ditch and swale storm water system, the catchment areas funneling water to the sites were too small. As a result, it was determined that the sites needed to be excavated such that the permanent ground water would provide the hydrology. The evaluation also showed that at Segment 3 hard clay was present approximately 30 - 45 cm (12 - 18 inches) below the surface. In order to provide the planted vegetation with an adequate substrate to grow, the site was over-excavated. The overburden surface was pushed aside, the clay excavated, and then the overburden was replaced, such that approximately 15 inches of sand was placed above the clay for vegetation establishment. This process was also used at Sutton Road West, which also contained hard clay approximately 30 - 45 cm (12 - 18 inches) below the surface.

In 1996, the sites were permitted under the new Environmental Resource Permitting process. In 1997, the project was put out to bid. After the bids were received, project costs were higher than expected. As a result, minor changes were made. The most notable was that the original plan called for one and three gallon trees to be installed. The plan was modified and approved by SWFWMD for mostly one gallon size trees to be installed. The project was put out to bid again and was awarded to Phillips & Jordan, Inc.

In August 1997 site construction began, starting with the Sutton Road West and East sites. These sites were completed in December 1997. Earthwork at Segment 3 also began in November 1997 to provide fill material to Sutton Road East. Earthwork was completed at Segment 3 in March 1998. Much of the work at Segments 2 and 3 was delayed due to the large amount of rain in the winter of 1997/98. As slopes were being cut and contoured, weekly heavy rains would erode away the work. As a result, some areas had to be cut and contoured several times. The final planting of the sites was completed in April 1998 with as-built submittal in May 1998.

One of the requirements Polk County had in its construction plan was for pre-planting as-builts. This would allow for careful evaluation of the planting zones and acreage calculations. This proved valuable when it was discovered that Segment 3 was only 1.49 ha (3.69 acres), not the proposed 1.54 ha (3.81 acres). As a result, the Segment 2 creation area was increased by 0.05 ha (0.12 acres) from 0.26 ha (0.63 acres) to 0.31 ha (0.75 acres) in order to meet the acreage requirement.

Polk County is committed to providing successful mitigation and complying fully with all permit conditions. Monitoring and maintenance activities will begin this summer. BCI Engineers & Scientists, Inc. is honored to have been able to provide our environmental and engineering services to Polk County and assist them with this very important project.

SEED AND SEEDLING ECOLOGY OF THE INVASIVE NON-INDIGENOUS SHRUB *ARDISIA ELLIPTICA* THUNB.(MYRSINACEAE) IN SOUTH FLORIDA

John B. Pascarella[1] and Carol C. Horvitz[2]
Department of Biology
University of Miami
PO Box 249118
Coral Gables, FL 33124-0421 USA

[1]Department of Biology, Valdosta State University,
Valdosta, GA 31698.
[2]Author for correspondence.

ABSTRACT

The seed and seedling ecology of the invasive non-indigenous shrub *Ardisia elliptica* Thunberg (Myrsinaceae) were studied. We sampled seed banks and determined seed rain in subtropical forests, wet prairies, and non-indigenous dominated successional areas within Everglades National Park, Florida. Soil seed bank and seed rain was greatest in successional areas. Seed germination was best in moist, shadehouse conditions and lowest in sunny, field conditions. Fresh seeds were highly viable (77-100%). However, seed viability declined with storage greater than two months. There was no evidence of a long-term seed bank. Seedling morphology and biomass allocation patterns differed from a native congener *Ardisia escallonioides*, with *A. elliptica* investing more biomass in shoots and less in roots.

INTRODUCTION

The control of non-indigenous invasive species depends on knowledge of the species' life-history, including the role of soil seed banks in recruitment and storage (Lonsdale et al. 1988, Groves 1989). Removal of juvenile and adult members of a population may not result in elimination of a non-indigenous species if seedling recruitment occurs primarily through long-term soil seed storage and/or through seed dispersal from adjacent source areas.

In Everglades National Park (ENP), Florida, the invasive non-indigenous shrub *Ardisia elliptica* Thunberg, native to southeastern Asia in lowland coastal rain forest (Henderson 1959), was the target of an intensive hand removal and herbicide treatment in a subtropical forest, Paradise Key (Seavey and Seavey 1989). Although this removal program eliminated more than 44,000 individuals, substantial populations of the shrub remained in disturbed abandoned agricultural lands next to Paradise Key (Seavey and Seavey 1989, Whitteaker and Doren 1989).

In addition to its presence in Everglades National Park, *A. elliptica* occurs in a variety of habitats in south Florida: it is known from subtropical hardwood forests, abandoned agricultural lands, bayheads, and edges of wet prairies (Long and Lakela 1971, Austin 1978, Morton 1979). It is listed as a Category 1 invasive exotic (Florida Exotic Pest Plant Council 1995), indicating that it is invading and disrupting native plant communities in Florida. In an urban forest preserve (Matheson Hammock, Coral Gables, Florida), *A. elliptica* dominated some parts of the forest in a 1991 survey. It formed a monospecific thicket with more than 110 stems >1 cm dbh in a 5 m^2 plot; seedlings of native plants were not found(Pascarella, unpub. data).

We used soil seed bank samples, seed rain data, and laboratory, shadehouse and field studies of seed viability, germination, and seedling growth to evaluate the seed and seedling ecology of this species. We also compare our results to the ecology of a native congener, *Ardisia escallonioides* Schlectendahl and Chamisso.

STUDY SITE

The field study sites were near Paradise Key, Everglades National Park (ENP), Florida. We sampled six habitats within 7 km of each other. Two hardwood hammock sites (Hammock North and Hammock South) were in Paradise Key. In both sites, selective removal of adult and juvenile *A. elliptica* had occurred during 1987-1989 (Seavey and Seavey 1989). Hammock North (HN) had a greater abundance of non-indigenous vegetation than Hammock South (HS) due to previous human inhabitation. In each site, two 50 m transects were established. The Wet Prairie (PR) site was between Farm Road and the old road from Paradise Key. Common vegetation was sawgrass (*Cladium jamaicense*), prairie grasses (*Andropogon* sp.), and herbs. Three 10 m transects were established on the east side of the old road, two on the west side of Farm Road, and four on the east side of Farm Road. The Old Ingraham Highway (OR) is a narrow abandoned road (approximately 48 years old at the time of the study) found to the south of Paradise Key and extending to the Farm Road near the Hole-in-the-Donut and *Ardisia* thicket. The road was created by dumping marl fill from an adjacent canal and was surrounded by areas of wet prairie. Regenerating hammock vegetation dominated most areas. Five 10 m transects were established on the roadbed. The Hole-in-the-Donut (DO) was abandoned agricultural land that was rock plowed. Agricultural furrows were removed before natural succession proceeded. The vegetation consisted of a monospecific overstory of the non-indigenous tree *Schinus terebinthifolius* with an understory of *A. elliptica* (Pascarella and Horvitz, pers. obs.). Study sites were located to the west and east of Farm Road. Five 10 m transects were established on the west side and one 10 m transect was included from the east side because of similar vegetation. The *Ardisia* Thicket (AT) occurred on former agricultural lands. It differed from the Hole-in-the-Donut in that agricultural furrows were not removed before natural succession. The *Ardisia* Thicket was south and east of the Hole-in-the-Donut. Two 50 m transects were established in this area.

MATERIALS AND METHODS

In April 1990, we took 234 soil samples (25 cm by 25 cm by 5 cm depth) every two meters along these transects. The total surface area of soil sampled was 14.625 m^2. Because the total length of the transects varied among the different communities and outcrops of limestone prevented all samples from being taken, the number of soil sample replicates varied (Old Road = 25, Hole-in-the-Donut = 30, Wet Prairie = 46, Hammock North = 40, Hammock South = 43, Ardisia Thicket = 50). We placed soil samples in individual flats in the University of Miami (UM) shadehouse. Flats were watered daily and regularly rotated in position. Every two weeks until October 1990, we identified *A. elliptica* seedlings, counted, and removed them. We then sieved soil to detect any remaining *A. elliptica* seeds. Mean number of seeds in the soil seed bank was compared among habitats using a non-parametric Kruskal-Wallis ANOVA.

We placed seed traps every four m along the transects in November 1990 ($N = 138$ traps, each 30 cm in diameter and 0.071 m^2 in area). Total surface area covered by all traps was 9.74 m^2. The number of replicates of seed rain varied among communities (Old Road = 16, Hole-in-the-Donut = 19, Wet Prairie = 28, Hammock North = 27, Hammock South = 27, Ardisia Thicket = 27). Traps consisted of fiberglass screening material sewed into the shape of a collecting bag. We suspended them from a ring of wire and attached them to iron rebar with hose clamps. We censused traps approximately every two weeks from November 18, 1990 to January 26, 1991 during the peak fruiting period of *A. elliptica*. We recorded seeds that had been stripped of the outer fruit as "gut-passed by frugivorous birds" and seeds with intact fruits as "passive dispersal".

We used seeds from trees growing at the University of Miami Gifford Arboretum (Coral Gables, Florida) for all seed germination experiments. We removed seeds from the fruit pulp to simulate dispersal. In 1991, we conducted an incubation experiment, consisting of 30 petri dishes each containing 10 seeds placed in an incubator (14 hours light/8 hours dark at 29° C). To examine germination under a variety of environmental conditions, we placed 100 seeds in germination trays (seven trays) at the UM shadehouse (50% shade, watered daily) in December 1992. Simultaneously, we planted 50 seeds in an adjacent subtropical forest under heavy shade (four replicates) and in gaps (two replicates). In both cases, seed germination was monitored monthly for one year. Mean time to germination and mean percent germination per seed batch was compared among treatments using a non-parametric Kruskal-Wallis ANOVA. To test for long-term seed viability, we removed seeds from fruit pulp, dried them, and stored them in the lab at room temperature (23.5° C). We planted stored seeds every month in the shadehouse for six months beginning April 1, 1994. We monitored germination monthly until no new seedlings were noted. A regression of percent germination on months stored was calculated.

In the UM shadehouse, we grew 20 seedlings of *A. elliptica* and *Ardisia escallonioides* in individual pots under identical conditions (irrigation but no fertilizer). After six months, we harvested the surviving seedlings, separated them into stems, leaves, and roots, dried them to a constant weight, and weighed them to the nearest milligram. We measured stem height, root depth, root width, and calculated root:shoot ratios. We compared parameters between the two species using a u-test.

RESULTS

We found statistically significant variation in the number of *Ardisia elliptica* seeds per soil seed bank sample ($P < 0.001$, $H = 70.22$; df = 5). Four groups were distinguished in a pairwise comparison test (Dunn's Method). Most seeds were found in the Hole-in-the-Donut, Ardisia Thicket, and Hammock South, while few seeds were found in the other habitats (Figure 1). Most *Ardisia* seeds (76%) germinated in the UM shadehouse in June with the remainder germinating in July, September, and October. Eleven potentially viable *Ardisia* seeds were found in the soil at the end of experiment.

We found a total of 190 seeds of *A. elliptica* in seed traps from all areas except for the wet prairie (Figure 2). The *Ardisia* Thicket had 90% of all seeds. Seeds were found during all census dates in about equal numbers. Most seeds (164) were fallen from branches (86.3%); only 26 seeds (13.7%) were gut passed.

Seeds in the incubator began germinating within one week and 77% of the seeds had germinated by 6 weeks. The experiment was terminated because of fungal invasion in the petri dishes. There was a significant difference in mean months to germination ($P < 0.001$, $H = 216.7$, df = 2). Seeds planted in the shadehouse commenced germination rapidly (mean = 3 months) while seeds in the shaded field conditions did not begin germinating until 6 months and 8 months in the sunny field conditions (Figure 3). Total germination success was similar for both shadehouse and field shade treatments (53% vs 51%) while germination success was lowest under field sun treatments (9%), although the difference was only marginally significant ($P = 0.068$, $H = 5.03$, df =2)(Figure 3).

Viability (measured as percent germination) was significantly negatively related to months stored in the laboratory ($P < 0.01$, $r = -0.91$, n = 7). Seed viability was very high for seeds stored less than 2 months (from 100% for fresh seeds to 72% for 2 month old seeds), but rapidly declined for seeds older than 3 months (<22% for 3-6 month old seeds) (Figure 4).

Seedlings of the two *Ardisia* species differed in growth patterns(Table 1). Seedlings of *A. elliptica* were taller while seedlings of *A. escallonioides* had deeper and broader roots. Although biomass comparisons were statistically insignificant, *A. elliptica* invested more biomass in shoots than *A. escallonioides*, which invested more in roots. Root/shoot ratio was significantly higher in *A. escallonioides* than in *A. elliptica*. Total biomass was similar between the two species

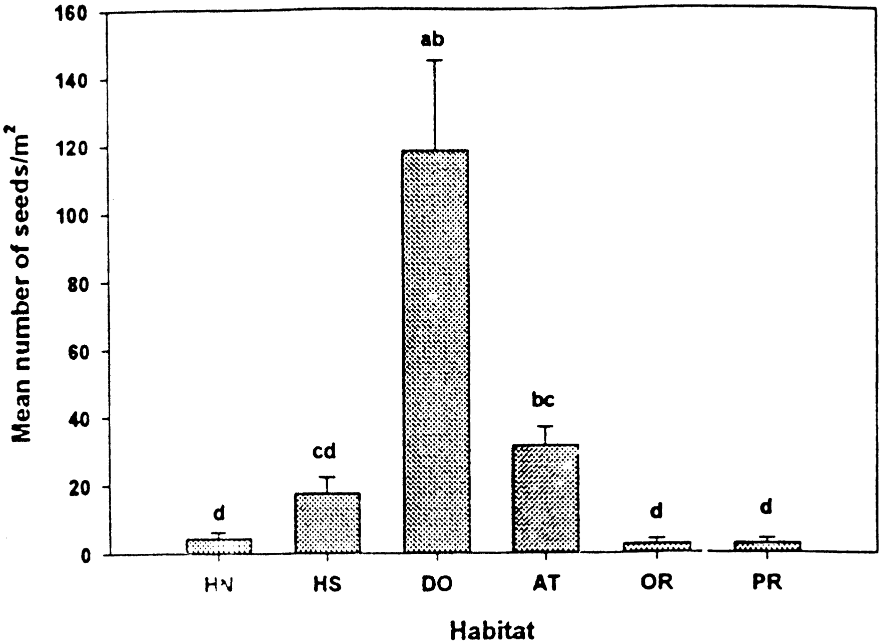


Figure 1. Soil seed bank (mean number of seeds/m^2) of *A. elliptica* in Everglades National Park. Different letters indicate statistically significant groups.

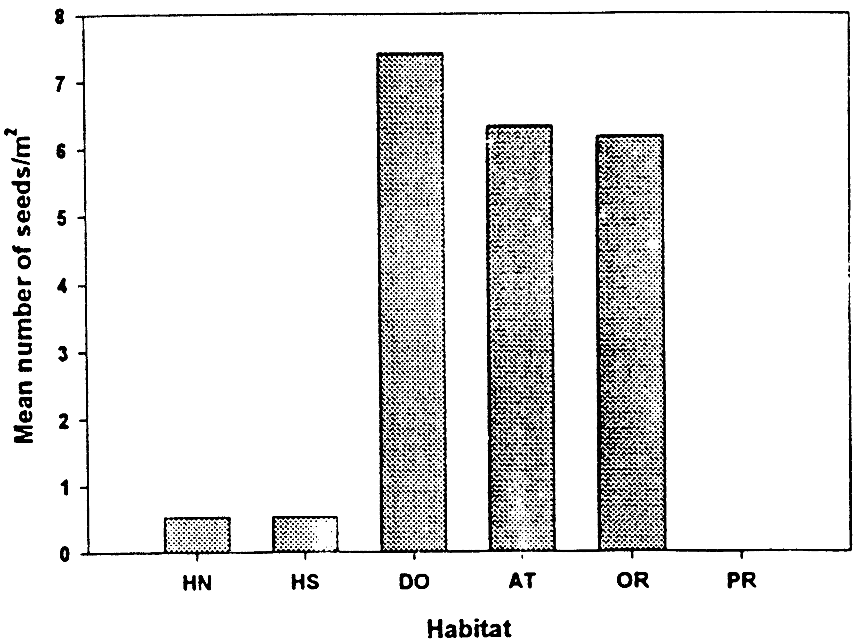


Figure 2. Mean seed rain (seeds/m^2) of *A. elliptica* in Everglades National Park

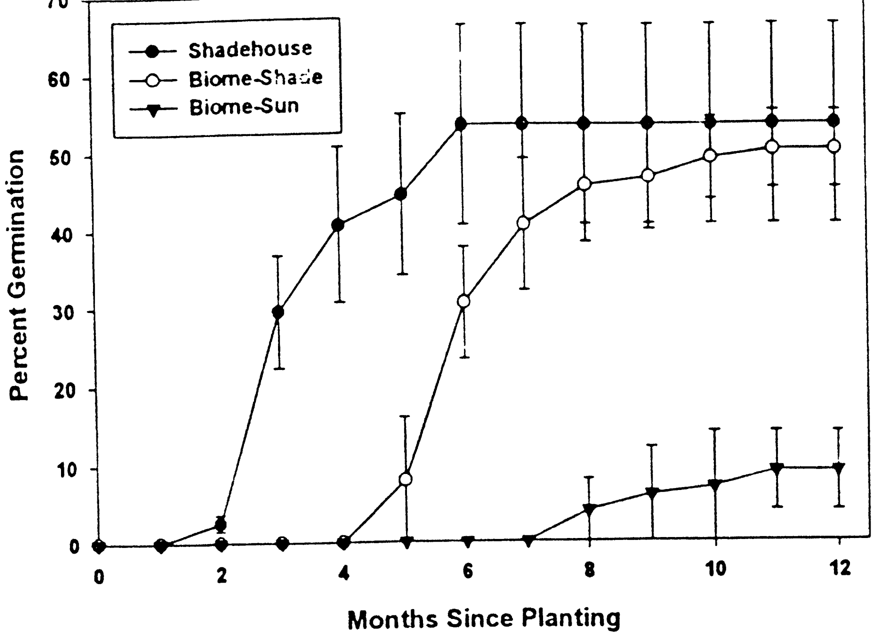


Figure 3. Phenology and percent germination of *A. elliptica* seeds in three environments.

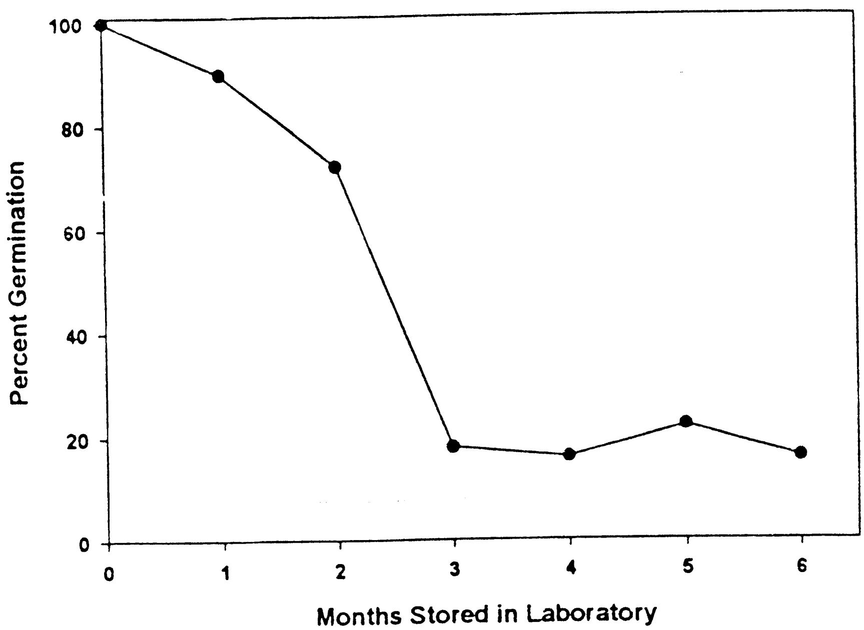


Figure 4. Percent germination of laboratory stored *A. elliptica* seeds.

TABLE 1. Comparison of six-month old seedling growth of the non-indigenous shrub *Ardisia elliptica* and a native congener *A. escallonioides* grown under identical shadehouse conditions. Sample size = seven seedlings/species. Comparisons of parameters were done using a u-test. **P* < 0.05, ** *P* < 0.01, N.S. = Not significant.

	A. elliptica	*A. escallonioides*	u-test
Stem Height (cm) ± 1 SD	3.89 ± 0.74	2.94 ± 0.50	**
Root Depth (cm) ± 1 SD	2.60 ± 1.55	6.21 ± 3.73	*
Root Width (cm) ± 1 SD	3.66 ± 1.30	6.51 ± 2.44	*
Number of Leaves ± 1 SD	4.86 ± 0.69	6.14 ± 2.19	N.S.
Total Biomass (mg) ± 1 SD	338.0 ± 96.8	346.3 ± 188.6	N.S.
Stem weight (mg) ± 1 SD	93.6 ± 16.5	88.3 ± 51.2	N.S.
Root weight (mg) ± 1 SD	110.7 ± 15.4	165.6 ± 29.8	N.S.
Leaf weight (mg) ± SD	133.7 ± 19.6	92.4 ± 25.9	N.S.
Root/Shoot Ratio	0.326 ± 0.0268	0.510 ± 0.0443	**

DISCUSSION

Ardisia elliptica is highly autogamous, capable of fruit set values of 60% without insect visitation (Pascarella 1997). Autogamous seeds are highly viable (Pascarella 1997). Reproduction can begin in less than two years and in plants smaller than one m tall (Pascarella 1997). In the Matheson Hammock population, the number of inflorescences was positively correlated with plant dbh ($P < 0.01$, $r = 0.63$, , n = 30, Pascarella, unpublished data). The strong correlation of plant size with inflorescence production and the highly autogamous breeding system indicates that fruit production should be greatest in the largest individuals. There are no known sources of predispersal seed predation (Pascarella, unpub. data), although recently, a seed-eating beetle has been found in a few populations (Tony Koop, University of Miami, personal communication). In contrast, the native congener *Ardisia escallonioides* has a specialized seed galling moth and can lose more than 90% of its potential fruit production to this insect (Pascarella 1996). Unlike *A. escallonioides*, which has an extensive rhizome network and can spread vegetatively (Pascarella 1995), *A. elliptica* does not have rhizomes and does not spread clonally (Pascarella, unpub. data). Thus, seed production and seed dispersal may be an important factor in the population dynamics of this invasive species.

Due to a long flowering season (May-February), ripe fruits are available from July-March (Pascarella, pers. obs.). In our study, most seed rain was due to passive dispersal of fruits falling off reproductive plants, suggesting that seed dispersal may be limited. As *A. elliptica* is a relatively new species in the flora, native frugivores may not utilize the fruits of this species or it may not be a preferred food item. However, native gray catbirds (*Dumatella carolinensis*, Mimidae) feed on *A. elliptica* fruits in ENP (C. Horvitz, unpub. data). Raccoon (*Procyon lotor*) scats with *A. elliptica* seeds have also been observed in ENP (Pascarella, pers. obs.). Because of limited seed removal, seed rain in this species is highly local as most seeds fall beneath parent plants. This localized dispersal may explain the formation of dense monospecific thickets of *A. elliptica* (Seavey and Seavey 1989). Seedlings are capable of growth under the relatively deep shade of adults (Horvitz, unpub. data). Although rare, seed dispersal by frugivorous birds and mammals is important in that a single established adult in a new area can develop into a thicket over time as well as reinvade previously weeded areas.

In the soil samples, most seeds of *A. elliptica* germinated in June, which corresponds to the beginning of the rainy season in south Florida. *Ardisia elliptica* has no long-term dormancy, as shown by the rapid germination response under ideal shadehouse conditions and the lack of long-term viability. In the field, ideal conditions are moist, shaded areas. In ENP, these conditions are most commonly found on the edges of hardwood hammocks that intergrade into wet prairies and bayheads. The lack of invasion of *A. elliptica* into pinelands in ENP may be related to lack of appropriate germination microsites in this drier, sunny habitat. In addition, *A. elliptica* may be intolerant of fire. Our study indicates that the habitat preference for moist, shady areas in both south Florida and other areas, such as Hawaii (Smith 1984), may be due to severe limitations on seed germination in more open, drier habitats. An approach such as used by Myers (1983) where seeds and seedlings are planted in a variety of natural habitats, in both dry and wet seasons, is needed to experimentally verify this hypothesis and to examine the response of the species to prescribed fire.

When adjusted for sample size, we found differences in both abundance of seeds in the seed bank and seed rain of *A. elliptica* in subtropical hardwood forests, wet prairies, and successional areas in ENP. These differences may be most parsimoniously explained by proximity of the sites to adult fruiting individuals. In subtropical hardwood forest areas where the adults of *A. elliptica* had been previously eliminated, we found substantial numbers of seeds in the seed bank, indicating that seed dispersal by frugivorous birds into these areas may currently be reintroducing the species. Although *A. elliptica* can occur in wet prairies, its low abundance in the seed bank may be due to limited seed dispersal as frugivorous birds may not use these primarily herbaceous areas. In successional areas, we found abundant seed rain, seeds in the seed bank, as well as abundant seedlings (Horvitz, unpub. data), indicating that the species is likely capable of maintaining itself and that succession towards a more native dominated habitat is unlikely.

Seed germination biology indicates that seeds germinate rapidly under wet, shaded conditions and that seed viability declines with age. Thus, seed banks are likely temporary (<1 yr.), arising from dispersal and declining with both germination and seed death. In contrast to *A. elliptica*, the native congener, *Ardisia escallonioides*, which has similar sized seeds, has strong short-term seed dormancy in the spring and early summer(6 months), followed by rapid germination in the fall (Pascarella 1995). Unlike *A. elliptica*, this species is more common in upland habitats such as the understory of pinelands. The difference in root to shoot ratios and larger root spread also support our hypothesis that *A. escallonioides* is more drought-tolerant than

A. elliptica.

Management of Category 1 invasive species in ENP emphasizes that only high value local areas or outlying populations that would enhance the spread of the species should be controlled (Whitteaker and Doren 1989). As most seed dispersal of *A. elliptica* is passive and local, elimination of outlying populations in high-quality natural areas surrounding the dense local populations at Hole-in-the-Donut and *Ardisia* Thicket may be the most effective way to control further spread into the previously cleared forest areas and to prevent new population establishment. The Hole-in-the-Donut is currently being restored to a wetland community (Doren et al. 1997). This will eliminate much of the seed source of *A. elliptica* and will help to contain further spread of the species in ENP.

ACKNOWLEDGEMENTS

We would like to thank Rick and Jean Seavey for their awareness of the invasive capacity of *Ardisia elliptica* in Everglades National Park. We also thank the members of the University of Miami Plant-Animal Interaction group for their assistance with this study. S. Florida Research Center provided permits to work in ENP. Funding to attend the meeting was provided by a Valdosta State University Faculty Research and Development Grant to J. Pascarella. This is contribution number 649 in the University of Miami Program in Tropical Biology, Ecology, and Evolution.

LITERATURE CITED

Austin, D.F. 1978. Exotic plants and their effects in southeastern Florida. Environmental Conservation 5:25-34.

Doren, R.F., D.L. Childers, T.V. Armentano, W.J. Platt, C. Horvitz, M. Norland, and B. Sparkman. 1997. Restoring wetlands on abandoned agricultural lands in Everglades National Park; A strategic plan for guiding the research, monitoring, and management of the "Hole-in-the-Donut" restoration program. South Florida Natural Resources Center Technical Report 97-002. 35 p.

Florida Exotic Pest Plant Council (EPPC). 1995. Florida exotic pest plant council's list of Florida's most invasive species, Spring 1995. 3205 College Ave., Ft. Lauderdale, FL.

Groves, R.H. 1989. Ecological control of invasive terrestrial plants. Pp. 437-461 *in* J.A. Drake et al., Biological Invasions: A Global Perspective. John Wiley and Sons, NY, NY.

Henderson, M.R. 1959. Malayan Wild Flowers (Dicotyledons). Caxton Press, Ltd. Kuala Lampur, Malaysia.

Lonsdale, W.M., K.L.S. Harley, and J.D. Gillett. 1988. Seed bank dynamics in *Mimosa pigra*, an invasive tropical shrub. J. Appl. Ecol. 25:963-976.

Long, R.W. and O. Lakela. 1971. A Flora of Tropical Florida. University of Miami Press. Miami, FL. 962 p.

Morton, J.F. 1979. Pestiferous spread of many ornamental and fruit species in South Florida. Proc. Florida Hort. Soc. 89:348-353.

Myers, R.L. 1983. Site susceptibility to invasion by the exotic tree *Melaleuca quinquenervia* in Southern Florida. J. Appl. Ecol. 20:645-658.

Pascarella, J.B. 1995. Effects of Hurricane Andrew on the population dynamics and mating system of the tropical understory shrub *Ardisia escallonioides* (Myrsinaceae). Ph.D. dissertation, University of Miami, Coral Gables, FL.

Pascarella, J.B. 1996. The biology of *Periploca* sp. (Lepidoptera: Cosmopterigidae): A gall maker on *Ardisia escallonioides* (Myrsinaceae). Florida Entomologist 79:606-610.

Pascarella, J.B. 1997. Breeding systems of *Ardisia* Sw. (Myrsinaceae). Brittonia 49:45-53

Seavey, R. and J. Seavey. 1989. Control of *Ardisia solanacea* in Paradise Key, Everglades National Park, 1987-1988. Unpublished document. 21 p.

Smith, C.W. 1984. Impact of alien plants on Hawaii's native biota. Pp. 180-210 *in* C.P. Stone and J.M. Scott, eds., Hawaii's Terrestrial Ecosystems: Preservation and Management. University of Hawaii, Honolulu, HA.

Whitteaker, L.D. and R.F. Doren. 1989. Exotic Plant Species Management Strategies and List of Exotic Species in Prioritized Categories for Everglades National Park. National Park Service, Southeast Region, Research/Resources Management Report SER-89/094, Atlanta, GA.

TREE SEEDLING ESTABLISHMENT ACROSS A HYDROLOGIC GRADIENT IN A BOTTOMLAND RESTORATION

R.K. Kolka[1], C.C. Trettin[1], E.A. Nelson[2], and W.H.Conner[3]

[1]Center for Forested Wetlands Research, USDA Forest Service, Charleston, SC 29414.
[2]Westinghouse Savannah River Technology Center, Aiken, SC 29802.
[3]Clemson University, Baruch Forest Science Institute, Georgetown, SC 29442.

ABSTRACT

Seedling establishment and survival on the Savannah River Site in South Carolina is being monitored as part of the Pen Branch Bottomland Restoration Project. Bottomland tree species were planted from 1993-1995 across a hydrologic gradient which encompasses the drier upper floodplain corridor, the lower floodplain corridor and the continuously inundated delta. Twelve species were planted in the three areas based on their flood tolerance and the hydrology of the area. Planted areas are separated by unplanted control strips to assess natural regeneration. A seedling survey conducted in 1997 showed that planted areas had significantly greater seedling densities than unplanted control sections. Water tupelo (*Nyssa aquatica*), green ash (*Fraxinus pennsylvanica*), sycamore (*Platanus occidentalis*), and persimmon (*Diospyros virginiana*) had the highest percent survival in the upper corridor while baldcypress (*Taxodium distichum*) had the best survival in the wetter lower corridor and delta. Water tupelo and green ash survival was low in wetter areas. Survival of planted species is dependent on hydrology, competition and herbivory although it is not possible to differentiate these effects from the available data.

INTRODUCTION

For over thirty years the bottomland hardwood system of the Pen Branch corridor and delta was used for the discharge of coolant water from a nuclear reactor. Prior to reactor start up, flow in Pen Branch was typically 1-2 m^3/s. Reactor operations raised the flow to as much as 10-12 m^3/s during reactor pumping (Nelson, 1996) and coolant waters were consistently 40-50 °C. By 1989, when the reactor was retired, this high-temperature and, elevated flow effluent had removed virtually all vegetation within the floodplain and eliminated the seed bank and root stock. By the early 1990's early successional vegetation covered the floodplain and delta with very little sign of the predisturbance bottomland forest.

In 1992, the USDA Forest Service began efforts to accelerate the restoration of the Pen Branch system to its previous bottomland state. The area was divided into three habitats or sections (Figure 1) based on hydrology and vegetation present: upper corridor (25 ha), lower corridor (16 ha), and delta (50 ha). Approximately 75% of the area was planted with native bottomland species using various site preparation techniques depending on the initial conditions present in the sections. The virtually unbroken thickets of black willow (*Salix nigra*) in the upper corridor were herbicided and burned to allow access and reduce overstory competition. The upper corridor was planted with cherrybark oak (*Quercus falcata var. pagodifolia*), swamp chestnut oak (*Q. michauxii*), water oak (*Q. nigra*), shumard oak (*Q. shumardii*), water hickory (*Carya aquatica*), pignut hickory (*C. glabra*), persimmon (*Diospyros virginiana*), sycamore (*Platanus occidentalis*), swamp tupelo (*N. sylvatica var. biflora*), green ash, water tupelo, and baldcypress (Table 1). The lower corridor was relatively open, and planting was done under the broken black willow canopy. The lower corridor was planted with cherrybark oak, swamp chestnut oak, green ash, water tupelo, and baldcypress (Table 1). The areas to be planted in the delta were herbicided to prevent competition from black willow on the ridges and cattails in the sloughs and planted with green ash, water tupelo, and baldcypress (Table 1). Planting was done in strips with unplanted, no site preparation control strips left between each planted area to assess natural regeneration (Figure 1). Species were selected based on their known tolerance to wet conditions and the hydrology of the corridor and delta (Table 1, Figure 2). Following each planting, surveys were conducted to monitor survival and growth (Dulohery et al., 1995). Understocked areas were replanted in 1995-1996. In the spring of 1996 a systematic pilot survey of seedling establishment was conducted, and the results of that survey were used to effectively design the 1997 survey.

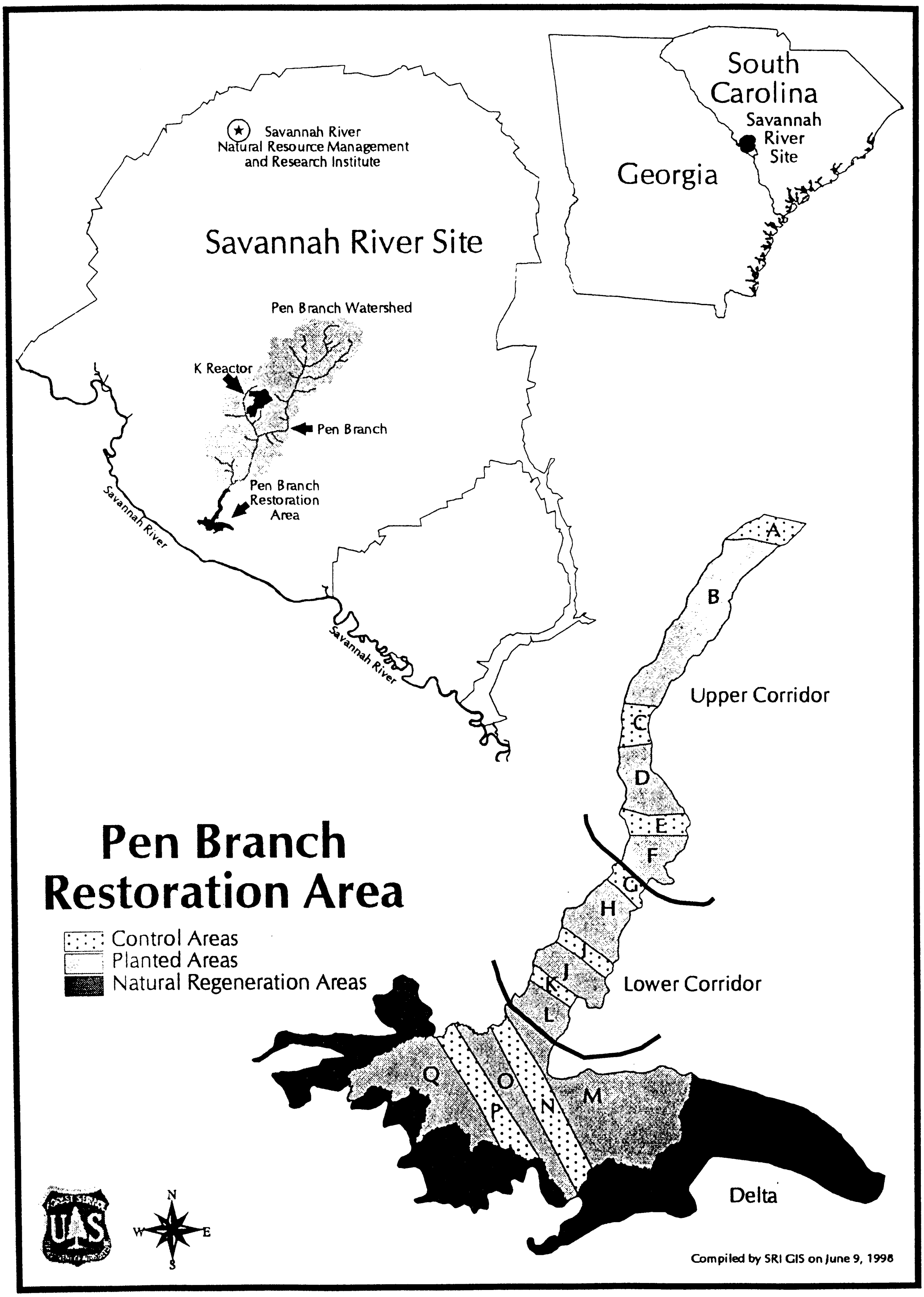


Figure 1. Location and research design of the Pen Branch Restoration Project.

The objective of this survey was to estimate the number of bottomland seedlings per hectare for each of the planted and control strips in the corridor and delta. A secondary objective was to survey the natural regeneration around the fringe of the impacted delta area.

METHODS

Statistical Design

Results of the spring 1996 regeneration survey were used as a guide to develop the statistical design of the 1997 survey. The goal of this survey was to estimate each strip mean with 90% confidence within ± 120 trees/ha (50 trees/ac). The number of plots needed is calculated using Equation 1 from Avery and Burkhart (1994),

$$n = 1/((E*2/(ts)*2) + 1/N) \qquad \text{[Eq. 1]}$$

Table 1. Percent distribution and total number of species planted in Pen Branch from 1993-1996. Note green ash, water tupelo and baldcypress were planted in all sections.

Species	Upper Corridor	Lower Corridor	Delta
Cherrybark Oak	22	7	0
Swamp Chestnut Oak	7	17	0
Water Oak	18	0	0
Shumard Oak	8	0	0
Water Hickory	14	0	0
Pignut Hickory	1	0	0
Persimmon	3	0	0
Sycamore	5	0	0
Swamp Tupelo	11	25	0
Green Ash	*9*	*25*	*10*
Water Tupelo	*1*	*12*	*60*
Baldcypress	*2*	*14*	*30*
Total (seedlings/ha)	**1831**	**1293**	**1012**

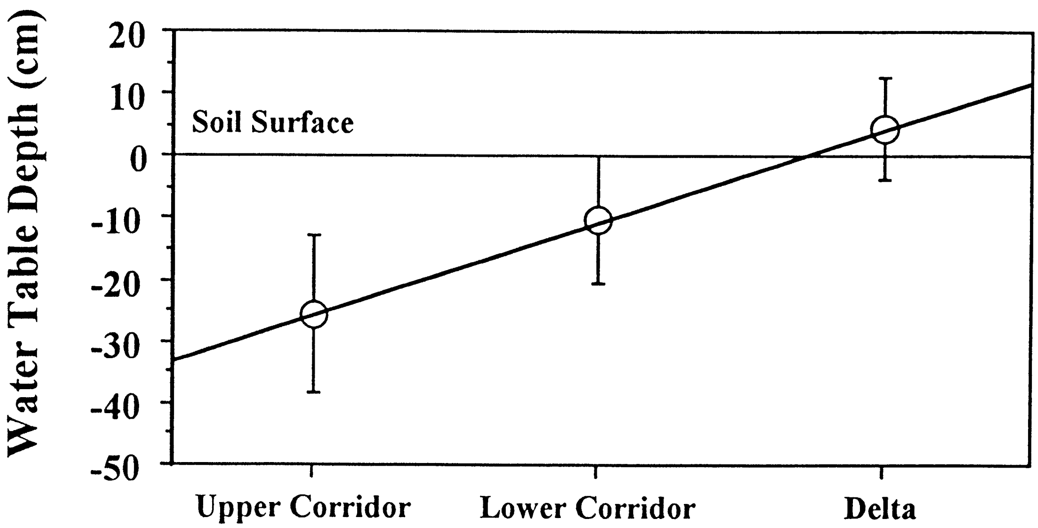


Figure 2. Hydrologic gradient in Pen Branch for 1993-1995 (± 1 SD).

where E is the allowable error (120 trees/ha), t is the t-distribution value for the given confidence level (0.90), s is the standard deviation, and N is the total number of possible plots.

The number of plots surveyed as a result of these estimates are shown in Table 2. We met our allowable error goal in all but one case (Table 2).

Although the design was developed to reach the desired level of accuracy for each strip, we also achieved a 95% CI of ± 120 trees/ha for the three planted and control section means. Results are comparisons of section means. We also sampled 63 plots in the natural regeneration zone around the fringe of the delta to assess the recovery of these less impacted areas.

Table 2. Number and location of 0.008 ha (0.02 ac) seedling survey plots for 1997. Error is the actual 90% confidence interval error attained for each strip.

Section	Strip	Number of Plots	Error seedlings/ha
Upper Corridor	Control A	17	62
	Planted B	47	116
	Control C	31	120
	Planted D	46	89
	Control E	25	118
	Planted F	57	77
Lower Corridor	Control G	14	128
	Planted H	38	120
	Control I	15	32
	Planted J	32	62
	Control K	11	0
	Planted L	26	62
Delta	Planted M	42	111
	Control N	35	17
	Planted O	29	114
	Control P	24	14
	Planted Q	39	35
TOTAL		**528**	

Field Design

The survey was conducted in April, 1997. Field crews of two or three tallied and identified all native bottomland species, including unplanted species typical of bottomlands such as red maple (*Acer Rubrum*), sweetgum (*Liquidambar styraciflua*), and river birch (*Betula nigra*) (Jones et al., 1994). Early successional species such as black willow, smooth alder (*Alnus surrulata*), wax myrtle (*Myrica cerifera*), and buttonbush (*Cephalanthus occidentalis*) were not tallied. Plots were 0.008 ha (0.02 ac) and placed 15 m (50 ft) apart along transects. The starting point of the transects were located at random intervals along the wetland boundary from established corners between planted and control strips (Figure 1). Transect bearing was perpendicular to the floodplain in the corridor and parallel to the long axis of strips in the delta.

Quality Control/Quality Assurance

We resampled 5.5% of the plots and found no significant difference in number of seedlings counted (paired t-test, $p > 0.10$). Correct identification of the seedling species occurred in 98% of the cases.

RESULTS AND DISCUSSION

Overall Seedling Survival

Seedling survival varied by species and by section (Table 3). Overall seedling survival increased as soils became more inundated, from 10% in the drier upper corridor to greater than 50% in the delta (Table 3, Figure 3). This gradient of survival was probably due to several factors including more herbivory from hogs, deer, and beaver in the open upper corridor and greater competition from herbaceous species, notably blackberry (*Rubus sp.*), which quickly became established after herbiciding and burning. Oak species had poor survival in the herbicided and burned upper corridor. Soon after planting, it was discovered that feral hogs were rooting up the oaks. It appeared that full canopy removal also allowed the hogs easy access to the seedlings. No site preparation in the lower corridor apparently led to more protected conditions for the oaks, leading to considerably greater survival (Table 3).

Persimmon, sycamore, green ash, and water tupelo had good survival in the drier upper corridor (Table 3). These species, especially sycamore and green ash, are fast growing and had broken through the herbaceous competition. Except for water tupelo, these species are also less water tolerant than some of the other species planted, and we would expect them to grow well in the relatively drier upper corridor.

Baldcypress is surviving extremely well in the wetter lower corridor and inundated delta (Table 3). Nearly 100% survival of any species is somewhat surprising. The obvious potential error in survival percentages is the counting of naturally regenerated volunteers as planted seedlings. This effect should be minimal because we subtracted the species density found in unplanted control sections from those in the planted sections. Natural regeneration of planted species was extremely low for all sections. Natural regeneration accounted for 58 stems/ha of baldcypress in the delta. Natural regeneration of other species in other sections was much lower. We do acknowledge that volunteers may comprise some small fraction of what was counted in the planted areas. It is possible that planted areas had nearer seed sources then unplanted controls and/or the site preparation techniques were more conducive to the establishment of volunteers.

Table 3. Percent survival of species planted in Pen Branch from 1993-1996.

Species	Upper Corridor	Lower Corridor	Delta
Cherrybark Oak	4	10	NP
Swamp Chestnut Oak	3	17	NP
Water Oak	4	NP	NP
Shumard Oak	0	NP	NP
Water Hickory	1	NP	NP
Pignut Hickory	15	NP	NP
Persimmon	35	NP	NP
Sycamore	42	NP	NP
Swamp Tupelo	7	NP	NP
Green Ash	42	9	18
Water Tupelo	54	15	24
Baldcypress	13	99	98
Overall	10	33	52

NP = species not planted

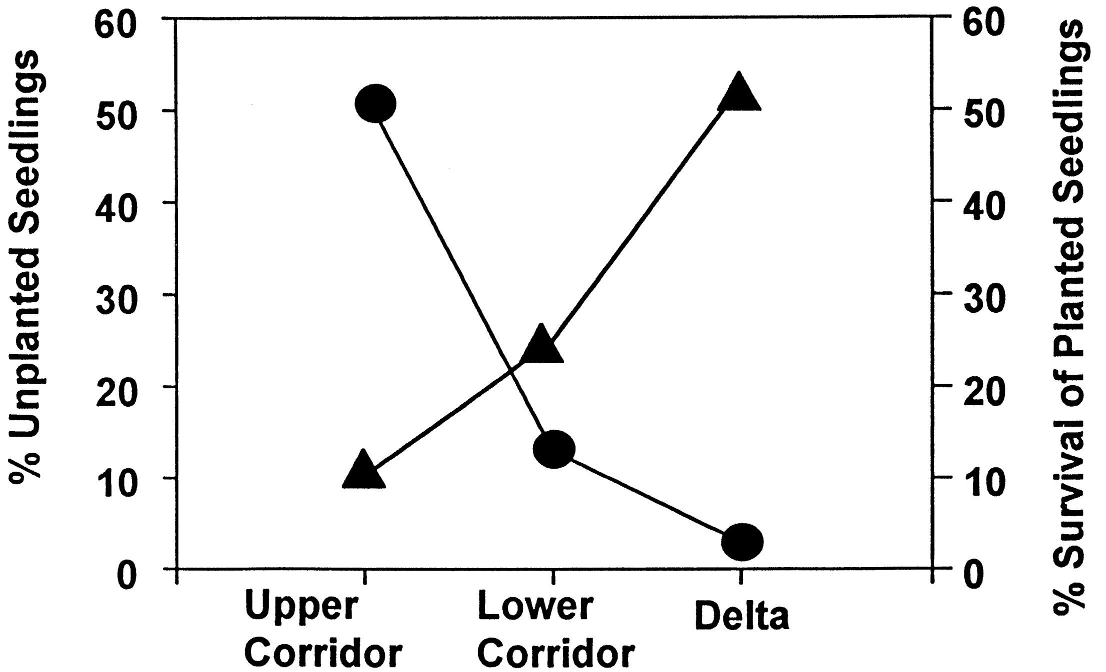


Figure 3. Distribution of unplanted seedling species (circles) and distribution and survival of planted seedlings (triangles).

Red maple was virtually unaffected by the herbicide and burning treatment in the upper corridor. It appears that site preparation in the upper corridor actually released red maple seeds. Red maple density was significantly greater in the upper corridor than in the lower corridor and delta (t-test, $p < 0.10$). Natural regeneration of red maple, sweetgum, and river birch is desirable and were counted as part of the overall bottomland seedling establishment. These species comprised about 50% of the bottomland species established in the upper corridor (Figure 3) of which, red maple represents the majority (95%) of unplanted seedlings. The percentage of red maple lessens as conditions become wetter, however it is not possible to differentiate the effects of site preparation techniques or nearness of seed sources from the hydrology.

Overall seedling establishment, including both unplanted native species and planted species, is significantly greater (t-test, $p < 0.10$) in the planted sections than in the unplanted control sections (Figure 4). We would certainly expect this result as much effort has been put forth to establish the planted seedlings. Bottomland seedlings established in the unplanted control sections included mainly red maple (51%) with river birch (17%), baldcypress (12%), sweetgum (6%) and sycamore (5%) also as important components. In planted sections, there were an average of 443 stems/ha, which falls within the range (330-900) reported for tree densities in unimpacted bottomland systems located on the Savannah River Site (Megonigal et al., 1997). Although we expect some seedling mortality to occur in the future, the 3-5 year old seedlings were well established, are above the herbaceous competition, and are growing vigorously. The main threat to their survival at this stage is from beavers. Often we observed planted seedling stumps that had been chewed by beavers.

Natural regeneration of the less impacted areas around the margin of the delta is highly variable. Mean stem density is 1750 ± 2410 stems/ha (1 SD), with a range of 0 to almost 10,000 stems/ha. Natural regeneration is comprised of mainly baldcypress (56%) with water tupelo (18%), red maple (16%), and sweetgum (8%) also as important components. Nearness to seed sources is obviously playing a very important role in the natural regeneration of the delta margin.

Survival of Baldcypress, Water Tupelo and Green Ash

Baldcypress, water tupelo, and green ash were planted in all three sections of Pen Branch. Comparison of these species within sections indicated no significant differences in survival in the upper corridor, however, baldcypress survival was significantly greater than either water tupelo or green ash in the lower corridor and delta (Figure 5).

Comparison of individual species across sections indicated no significant differences in survival of water tupelo or green ash. Baldcypress survival is greater in the lower corridor and delta than in the upper corridor (Figure 6). Although baldcypress is very tolerant of wet conditions (Hook, 1984) and survival increases as conditions become wetter, we can not attribute these differences to the hydrologic gradient alone. If hydrology was the only effect leading to survival, we would also expect water tupelo, a species also very tolerant to wet conditions (Hook, 1984), to have increased survival as conditions become wetter. We would also expect green ash, a species not as tolerant as baldcypress and water tupelo to wet conditions (Hook, 1984), to have a decreasing gradient of survival from the upper corridor to the delta.

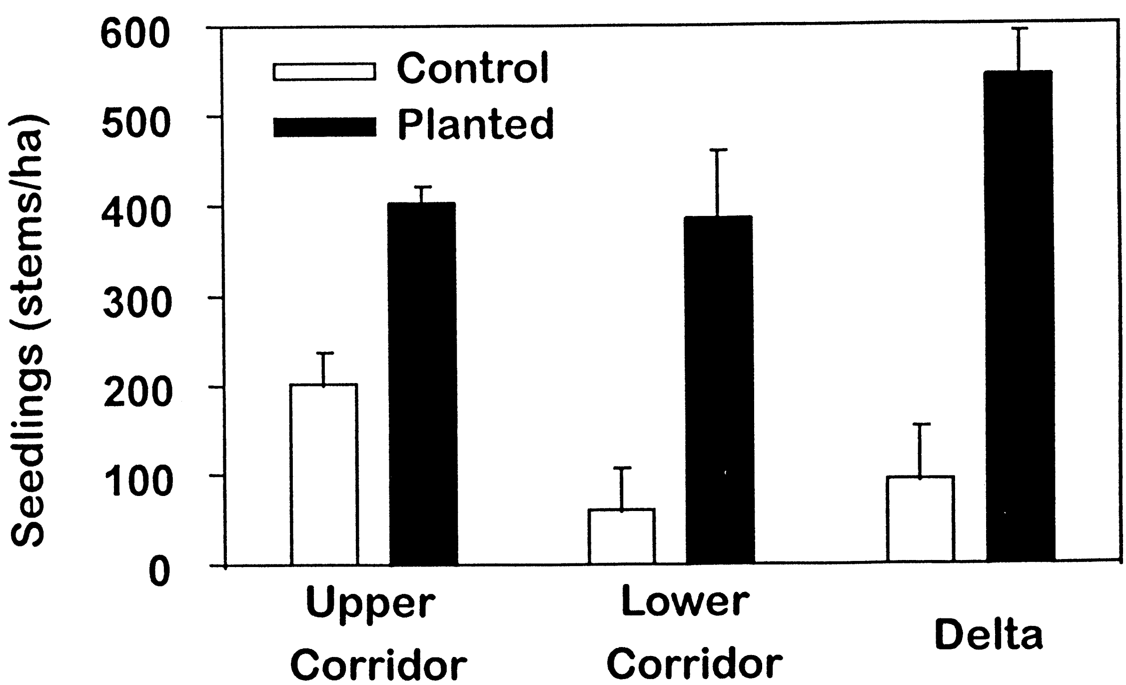


Figure 4. Overall seedling densities in the Pen Branch corridor and delta (error bars = 1 SD). Densities include unplanted bottomland species (red maple, river birch, and sweetgum).

The results indicate that hydrology is not the only factor controlling seedling survival of baldcypress, water tupelo, and green ash in Pen Branch. Herbivory and competition are also controlling survival. The effect of herbivory and competition were variable across sections depending on the site preparation method applied. Water tupelo is not as tolerant as baldcypress and green ash to shaded conditions (McKnight et al., 1981), and this may explain the low survival of water tupelo in the lower corridor. The lower corridor was planted directly under the scattered black willow canopy. However, green ash, a species tolerant of shaded conditions (McKnight et al., 1981), also had poor survival in the lower corridor. Degree of herbivory possibly explains the bulk of the variability in survival for these three species. Unfortunately, our study did not differentiate seedling mortality from the effects of hydrology, competition and herbivory.

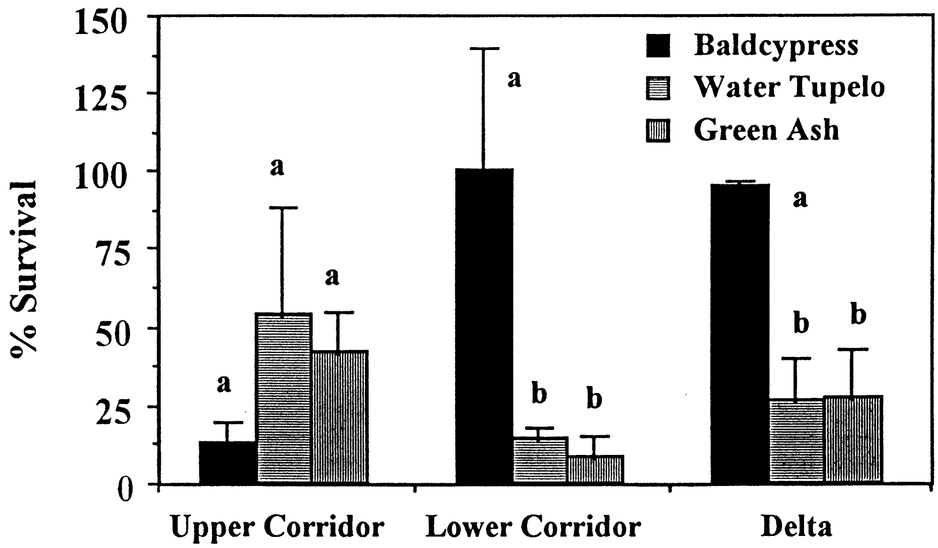


Figure 5. Comparison of species survival within sections of the Pen Branch corridor and delta. Error bars $\pm$ 1 SD, significance at the 0.10 level (t-test).

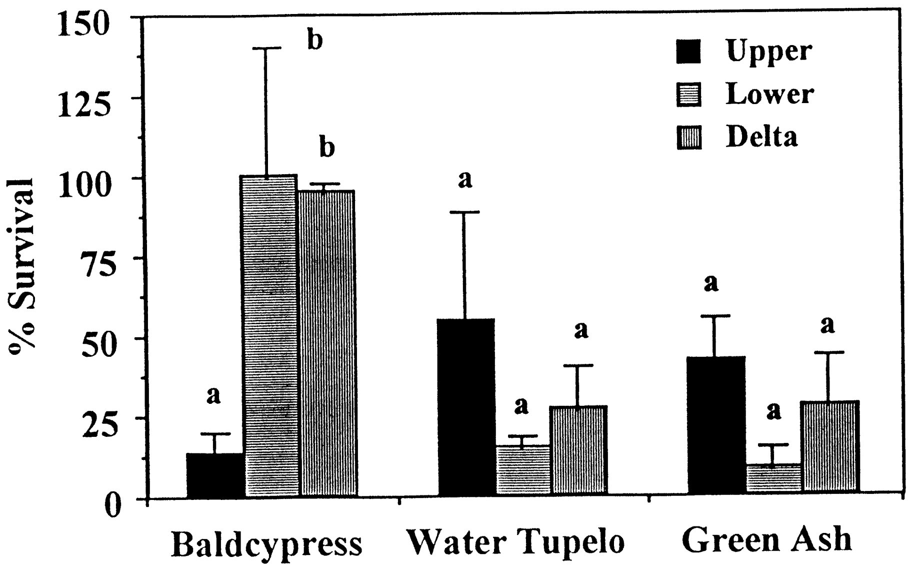


Figure 6. Comparison of individual species survival across sections of the Pen Branch corridor and delta. Error bars ± 1 SD, significance at the 0.10 level (t-test).

CONCLUSION - FURTHER RESEARCH

Seedling establishment in Pen Branch is variable depending on a number of factors. Hydrology is undoubtedly very important when considering seedling establishment, but site preparation methods, as they relate to competition and herbivory effects, are also important. We were not able to differentiate the effects of hydrology from competition and herbivory on seedling establishment in Pen Branch.

We have two ongoing studies designed to assess the effects of herbivory and competition on seedling survival and growth. In our study of competition effects, we replicated treatments of various canopy removal levels. Baldcypress, green ash, water tupelo, and swamp chestnut oak seedlings were planted under full mechanical removal of overstory, full herbicide removal of overstory, 60% removal of canopy, and intact black willow canopy (control). Plots were fenced to minimize herbivory. We recently took our fifth and final year of measurements on these plots. Preliminary analysis indicates that the mid-level, 60% canopy removal treatment had greatest survival and growth. It appears some canopy removal is desirable to allow light penetration to the seedlings without stimulating dense growth of herbaceous species which overtop the seedlings.

A second study is assessing the use of tree shelters on the survival and growth of baldcypress, green ash, water tupelo, and swamp tupelo seedlings. Tree shelters provide protection from herbivory. Replicated plots with and without tree shelters are in their fifth year of growth. Preliminary results suggest that tree shelters positively affect survival and growth of seedlings. Characterizing the magnitude of the positive response will allow us to assess the effect of herbivory on seedling survival. Results obtained from the competition and tree shelter studies will allow us to separate the effects of herbivory and competition from the effect of hydrology on seedling establishment.

LITERATURE CITED

Avery, T. E. and H.E. Burkart. 1994. Forest Measurements, 4th edition. McGraw-Hill, New York City, New York. 403 pp.

Dulohery, N.J., C.S. Bunton, C.C. Trettin, and W.H. McKee. 1995. Reforestation, Monitoring, and Research at Pen Branch: Restoring a Thermally-Impacted Wetland Forest. Establishment Report. USDA Forest Service. 55 pp.

Hook, D.D. 1994. Waterlogging tolerance of lowland tree species of the South. Southern Journal of Applied Forestry, 8(3): 136-149.

Jones, R.H., R.R. Sharitz, S.M. James, and P.M. Dixon. 1994. Tree population dynamics in seven South Carolina mixed species forests. Bulletin of the Torrey Botanical Club, 121(4): 360-368.

McKnight, J.S., D.D. Hook, O.G. Langdon, and R.L. Johnson. 1981. Flood tolerance and related characteristics of trees of the bottomland forests of the southern United States. p. 29-69. In Wetlands of Bottomland Hardwood Forests. J.R. Clark and J. Benforado, eds. Elsevier Scientific Pub. Co., Amsterdam, 401 pp.

Megonigal, J.P., W.H. Conner, S. Kroeger, and R.R. Sharitz. 1997. Aboveground production in southeastern floodplain forests: A test of the subsidy-stress hypothesis. Ecology, 78(2): 370-384.

Nelson, E. A. 1996. The restoration of Pen Branch: defining and measuring the progress of a thermally impacted stream becoming a functional wetland ecosystem. Abstracts from Symposium held on Feb. 13, 1996, Savannah River Site. WSRC-MS-96-0257X.

LAND IMPRINTING FOR RESTORING VEGETATION IN THE DESERT SOUTHWEST

Robert M. Dixon[1] and Ann B. Carr[2]

ABSTRACT

During the past two decades land imprinting has been used to restore perennial grasses on 20,000 hectares of degraded rangeland in southern Arizona, alone. Using homemade imprinters, several ranchers have profited from the greatly increased forage produced by the restored grasses. Elsewhere in the Desert Southwest, imprinter seeding has been directed to ecological restoration of native ecosystems. Seed mixes of early, mid and late seral species have been germinated in soil imprints at several locations in the Sonoran, Colorado and Mojave Deserts. V-shaped imprints funnel seed, rainwater, eroded soil and plant litter together where these resources can work in concert to germinate seeds and establish seedlings. Imprinting has established vegetation successfully on degraded land areas where annual precipitation ranges from 76 to 356 mm. The imprinting technology is currently being extended to the revegetation of steep slopes to control erosion and sedimentation. Future development will be directed to using imprinting in wetland restoration.

INTRODUCTION

Revegetation is needed in the Desert Southwest to replace the perennial grasses nearly eliminated by a combination of cattle grazing and drought (Roundy and Biedenbender 1995). Loss of the grass cover has greatly decreased rainwater infiltration and increased water runoff, erosion, flash flooding and sedimentation. Reduced recharge of upland aquifers has lowered water tables, thereby reducing or stopping the flow of springs and streams.

---

[1] Chair and [2] Vice Chair, The Imprinting Foundation, 1231 E. Big Rock Road, Tucson, AZ 85718. TEL: (520) 297-6165, FAX: (520) 323-8150

Currently, rangeland revegetation objectives are changing to meet the needs of society. This is especially true for public lands managed by the U.S. Forest Service and the Bureau of Land Management. There is greater emphasis on ecological restoration for biodiversity, wildlife and recreation. Revegetation approaches have changed to achieve these new goals. When increased grass forage production was the principal goal, classical agricultural wisdom was applied, whereby existing vegetation was destroyed using a variety of mechanical and chemical methods to eliminate competition with the grasses to be seeded. Often the final plant community was essentially a monoculture of an exotic grass such as Lehmann lovegrass (*Eragrostis lehmanniana*). Thus the complex, although degraded, existing ecosystem was converted to a simple agricultural ecosystem of exotics including Eurasian livestock—usually cattle. Exotic grasses were not only easier to establish in the degraded rangeland, but also could cope better with the intense grazing of large exotic ungulates, having coevolved with them in Africa and elsewhere.

RESULTS AND DISCUSSION

The new land treatment process, land imprinting, has been under development since 1976 when the first imprinter was fabricated in the machine shop at the USDA's Walnut Gulch Experimental Watershed which surrounds Tombstone, Arizona (Dixon and Simanton 1977). Development of this new method has been driven by two needs. First was the fact that the rangeland drill, often considered the best conventional method, was only marginally successful about one time out of ten. And next was the growing need for a method that would restore perennial grasses without destroying the existing vegetation; i.e., an effective method for interseeding the missing ecosystem component.

Imprinter seeding, when done properly, has been successful about nine times out of ten both for increasing forage production and ecological restoration. The greater success of imprinting relative to drilling was attributed to greatly improved control of rainwater at the soil surface (Dixon 1990). Ranchers, using homemade imprinters, have interseeded perennial grasses on some 20,000 hectares of degraded rangeland in southern Arizona, alone.

Failure of imprinter seeding has almost always been directly attributable to either poor imprints or poor seed or both. Poor imprints result from the use of substandard equipment and/or operating procedures. Poor imprints are those which are relatively shallow and/or unstable. Poor seed can have a number of causes, but perhaps the most common one is insufficient pioneer species in the seed mix to improve the microenvironment enough to help in the establishment of later seral species. These pioneer species serve as cover, nurse, mulch and green manure plants. They are especially needed where the land has been severely disturbed and the revegetation objective is ecological restoration.

Shallow or partial imprints result from a poorly designed imprinting roller, insufficient imprinter ballast or extremely hard soils. Ripping to soften extremely hard rangeland soils should not be done as an alternative to adding more imprinter ballast unless the imprinting pressure required for an adequate imprint exceeds 207kPa (30 psi.). Another alternative is to wait until a rain has softened the soil—an approach especially appropriate for fall seeding of rocky rangeland soils.

Imprint instability may be caused by initial tillage, lack of surface cover such as plant litter or gravel and coarse (sandy) soil texture. Time of imprinting also affects imprint stability. Fall imprinting is recommended because of the prevalence of gentle rains which settle and stabilize the imprint geometry. A rapid-growing cover crop of cool season annual grasses will further stabilize imprints before they are exposed subsequently to the highly erosive summer monsoonal rainfall. Special care should be taken to stabilize imprints in sandy soil which tends to be inherently unstable. Tillage prior to imprinting should be avoided as it regresses the secondary succession back to the starting point or to a thick stand of severely competitive pioneer plants. Such tillage, not only kills desirable plants, but also severely disrupts the soil ecosystem which otherwise would facilitate the establishment of perennial grasses and ecosystem restoration. Disruption of cryptogamic crusts, mycorrhizal fungi and invertebrate communities is especially harmful to the natural functioning of soil ecosystems. Tillage also accelerates oxidation of organic matter and the breakdown of soil structure.

Increasingly, imprinter seeding is being directed to ecological restoration of desert savannas instead of grass forage production. Good seed mixes are fundamental to the success of such projects. Failure or limited success is often the result of not using enough early seral species. They stabilize and improve the soil for later seral species. Two large scale projects in the

Sonoran desert near Tucson, Arizona will serve to exemplify the general approach to ecological restoration of severely degraded land through imprinter seeding (Dixon and Carr 1993).

In the first project, an 8-km stretch (240 ha) of severely disturbed floodplain along the Santa Cruz River was imprinter seeded during November 1987. The disturbed floodplain had been leveled, straightened, and walled to allow housing development within most of the outlying historic floodplain. Revegetation was required to mitigate the hydrologic effects of reshaping the floodplain including accelerated flow of floodwater, floodplain erosion, and downstream sedimentation. In the second project, a strip of severely disturbed land in the foothills of the Tucson mountains was seeded during November 1991. The 80-ha strip, 11 km in length, was disturbed during the installation of a large underground aqueduct by the Central Arizona Project to supply irrigation water to the San Xavier Indian Reservation. Complex native seed mixes, which were used at both project locations, included early, mid and late successional species to help accelerate the secondary succession toward a stable plant community with biodiversity equal to or greater than relatively undisturbed nearby areas. Thick stands of exotic weeds, present at the time of imprinting at both locations, served well in the roles of cover, nurse, mulch and green manure. Imprinting converted these weeds into a water saving, soil enriching mulch partially imbedded in the faces of the imprint. Many species within the seed mix responded rapidly to the imprinted seedbeds and seedling cradles. Consequently, plant communities at both locations are progressing rapidly toward the biodiversity goal. The V-shaped imprints funnel resources together at the imprint bottom where they can work in concert to germinate seeds and establish seedlings. The imprints also protect small seedlings against the desiccating effects of strong winds and hot sunlight to help them get their roots down before they have to face the severe macroclimate above. The relatively long life of imprints and natural seed dormancy greatly increase the chances for imprinting success relative to conventional drilling of seed. Thus, imprints and seeds can last through several years of drought and still function to germinate seeds and establish seedlings when the rains finally come.

Land imprinting arose from extensive infiltration studies which found that degraded/desertified land surfaces become smooth and sealed and as a consequence shed most of the rainwater instead of infiltrating it (Dixon, 1995). Imprinting was conceived as the most benign method possible for restoring the surface microroughness and macroporosity to, in turn, accelerate infiltration and revegetation processes (Dixon and Simanton 1977). Imprints are formed by downward acting forces (much like foot and hoof prints) without soil surface inversion, uprooting of plants, covering of plant materials, and destruction of cryptogamic crusts, mycorrhizae and soil invertebrates. Imprints are well-firmed and well-formed V-shaped pockets which funnel rainwater, plant litter, splash-eroded soil and seeds together where these resources can work in concert to germinate seeds and establish seedlings (Dixon and Carr 1994). Since the imprints are small closed microwatersheds (usually about 30-cm square) they do not bleed resources downslope as do the furrows of conventional methods such as drill seeding. Thus, imprinting goes a long way toward achieving the long held conservation goal of holding soil and water resources in place to maximize biomass production while maintaining and building topsoil for sustainable productivity indefinitely into the future.

SUMMARY AND CONCLUSIONS

Imprinter seeding is highly successful in the Desert Southwest for increasing grass forage production for livestock and for restoring ecosystems for biodiversity, wildlife and recreation. However, these goals are often somewhat incompatible and thus cannot be achieved to the maximum degree on the same land area at the same time.

Correct use of the new imprinting technology requires a marked departure from conventional agricultural wisdom for growing annual crops and the application of ecological principles for accelerating secondary succession of plant communities following land disturbances.

Common problems and mistakes which limit the success of imprinter seeding include:

1. Poorly designed imprinting roller.
2. Prior tillage for weed and brush control.
3. Inadequate ballast for a full-tooth imprint.
4. Operating imprinter in wet soil that sticks to the imprinting teeth.
5. Failure to rip soil that has been deeply compacted by heavy equipment.
6. Seed mix with insufficient early seral species present to accelerate the secondary succession

Finally success in imprinter seeding as in most no-till methods requires perseverance and the belief that it will work if done properly (Orchard 1996). It's not so much a question of whether imprinter seeding will work, but rather how can it be made to work?

Making imprinting work in a variety of new situations may require minor modifications in the standard equipment and operating procedures. Imprinting equipment is currently being adapted for use on steep slopes (Dixon and Carr 1997). This entails the development of an imprinting tooth with a curvilinear triangular cross section to increase water and seed storage space on steep slopes. Additionally, crawler tractors with self-cleaning triangular track pads can be readily adapted to imprinting steep slopes. An imprinting roller clamped to the dozer blade will imprint the space between the tracks. This same arrangement can be easily adapted to wetland restoration.

REFERENCES

Dixon, R.M. 1990. Land imprinting for dryland revegetation and restoration. In: Environmental Restoration: Sciences and Strategies for Restoring the Earth. Edited, John J. Berger. Island Press, Washington, D.C. p. 14-22.

Dixon, R.M. 1995. Water Infiltration control at the soil surface: Theory and practice. Journal of Soil and Water Conservation 50 (5): 450-453.

Dixon, R.M. and A.B. Carr. 1993. Recent revegetation projects in southern Arizona using land imprinting. Abstracts Eighth Wildland Shrub Symposium Shrub Research Consortium, University of Nevada, Las Vegas. Oct. 19-21. p. 30-31.

Dixon, R.M. and A.B. Carr. 1994.Land imprinting for low-cost revegetation. Erosion Control 1 (3) : 38-43.

Dixon, R.M. and A.B. Carr. 1997. Infiltration regulation for erosion control: Principles and practices. Proceedings International Erosion Control Association, 28th Conference. Nashville, Tennessee. p. 155-164.

Dixon, R.M. and J.R. Simanton. 1977. A land imprinter for revegetation of barren land areas through infiltration control. Arizona-Nevada Acad. Sci. and Am. Water Resources Assoc. 7: 79-88.

Orchard, Julie. 1996. No-till takes perseverance. No-Till Farmer, Lessiter Publications, October. p. 8.

Roundy, B.A. and S.H. Biedenbender.1995.Revegetation in desert grassland. In: Desert Grasslands. Edited, Mitchel P. McClaron and Thomas R. Van Devender. University of Arizona Press, Tucson, Arizona. p.265-303.

CAPE FLORIDA STATE RECREATION AREA WETLANDS RESTORATION

Gary R. Milano
Miami-Dade Department of Environmental Resources Management(DERM)
33 SW 2nd Avenue Miami, FL 33130-1540

ABSTRACT

During the 1950's, over 809 square meters (0.20 acres)of wetlands on the south end of Key Biscayne, Florida were destroyed through the placement of over 765,000 cubic meters (one million cubic yards) of dredge fill and approximately 3.2 km. (two miles) of bulkheaded shoreline associated with a failed development. Those wetlands, which are essential to the general health of the coastal marine and estuarine ecosystem, were replaced with a dry land forest of invasive exotic Australian pines (*Casuarina equisetifolia*). The passage of the northern eye wall of Hurricane Andrew (1992) over Cape Florida State Recreation Area completely leveled the forest of invasive exotics that dominated the created uplands. In the aftermath of the storm, the Florida Department of Environmental Protection, Division of Parks and Recreation (FDEP) developed a draft conceptual recovery and restoration plan for the park. A major objective of the plan was to restore, to the extent possible, the historic vegetation types present on this portion of Key Biscayne prior to the addition of fill material. The vegetation types included beach dune, coastal strand, maritime hammock, interior isolated freshwater wetland and a large tract of tidally connected mangrove wetland in the northwest portion of the park. Three hundred and forty-four square meters (0.085acres) of historical wetlands are being restored at the park, through cooperative efforts of federal, state and local agencies. DERM was identified as the lead agency for the implementation and execution of the wetlands restoration plan. The restoration plan has involved the removal of exotics, removal of portions of the bulkhead and fill, placement of a protective limerock barrier, elevation grading, creation of isolated freshwater wetlands, tidal pools, flushing channels, and the planting of wetland vegetation.

INTRODUCTION

Cape Florida is located ten miles southeast Miami, Florida, on the southern tip of Key Biscayne, a natural barrier island (Figure 1). During the early 1950's, approximately 1538 square meters (0.38 acres) of natural vegetation on the south end of Key

Biscayne were filled with dredged Biscayne Bay bottom for development purposes. The area became populated by a dense upland forest of invasive exotic Australian pines *(Casuarina equisetifolia)* and twenty nine other invasive exotic species. In addition, approximately 3.7 km (two miles) of concrete bulkhead was installed to contain approximately 2,295,000 cubic meters (three million cubic yards) of fill which added 1.53 M. (five feet) of elevation to the area. In 1966, the State of Florida acquired the 1644 square meter (0.406 acre) tract of land, and in 1969 designated it a State Recreation Area. Upon acquisition, only 109 square meters (0.027 acres) of the park's uplands supported natural plant communities. The cultural resources of the park included five documented pre-Columbian and historic sites. Most prominent of the historic sites is the Cape Florida Lighthouse, built in 1825.

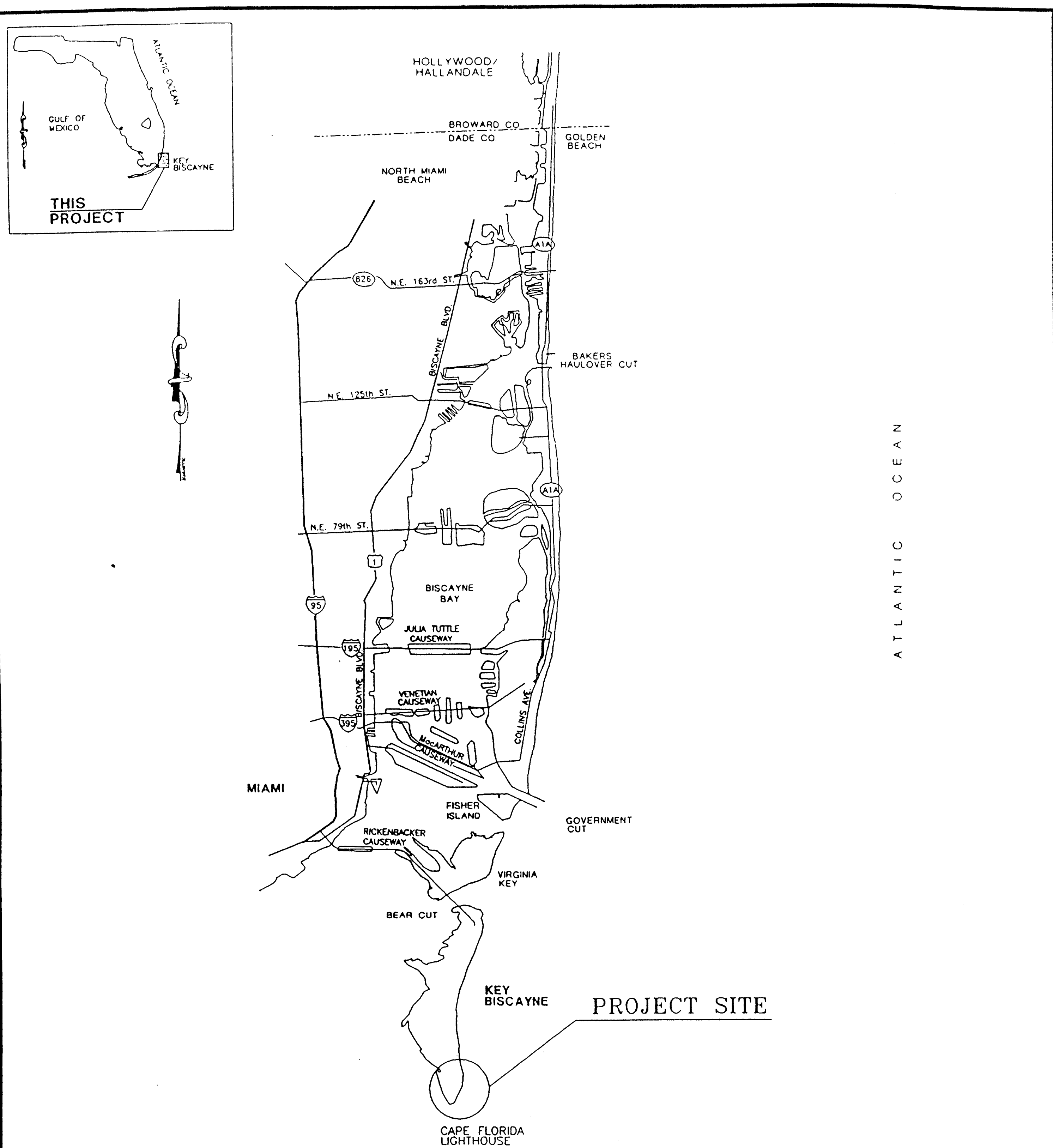
The passage of the northern eye wall of Hurricane Andrew (1992) destroyed the Australian pine forest that covered approximately 1538 square meters (0.380 acres) of the Park. In the aftermath of the storm, the Florida Department of Environmental Protection, Division of Park and Recreation (FDEP) developed a recovery and restoration plan for the park. A major objective of the plan was to restore, to the extent possible, the historic vegetation types present on this portion of Key Biscayne. The vegetation types included beach dune, coastal strand, maritime hammock, isolated freshwater wetland and a large tract of tidally connected mangrove wetland in the northwestern portion of the park (Figure 2). The ecological importance of coastal wetlands as habitat and as a vital link in the main food web has been well documented (Idyll et al., 1968; Odum et al., 1982). Miami-Dade County Environmental Resources Management (DERM) assisted in the development of the restoration plan and was identified as the lead agency for the implementation and restoration. The plan called for the creation of 40.5 square meters (0.010 acres) of freshwater isolated wetlands and 304 square meters (0.075 acres) of tidally connected wetlands. This paper presents a review of the elements involved with the design and restoration of the coastal wetlands at the park.

Restoration Plan Development

The restoration plan was developed through review of historical documents (1926 aerial photograph and personal observations) and field investigations of site characteristics. Field investigations included topographic, biological, geotechnical, hydrological, and archaeological reviews of the site.

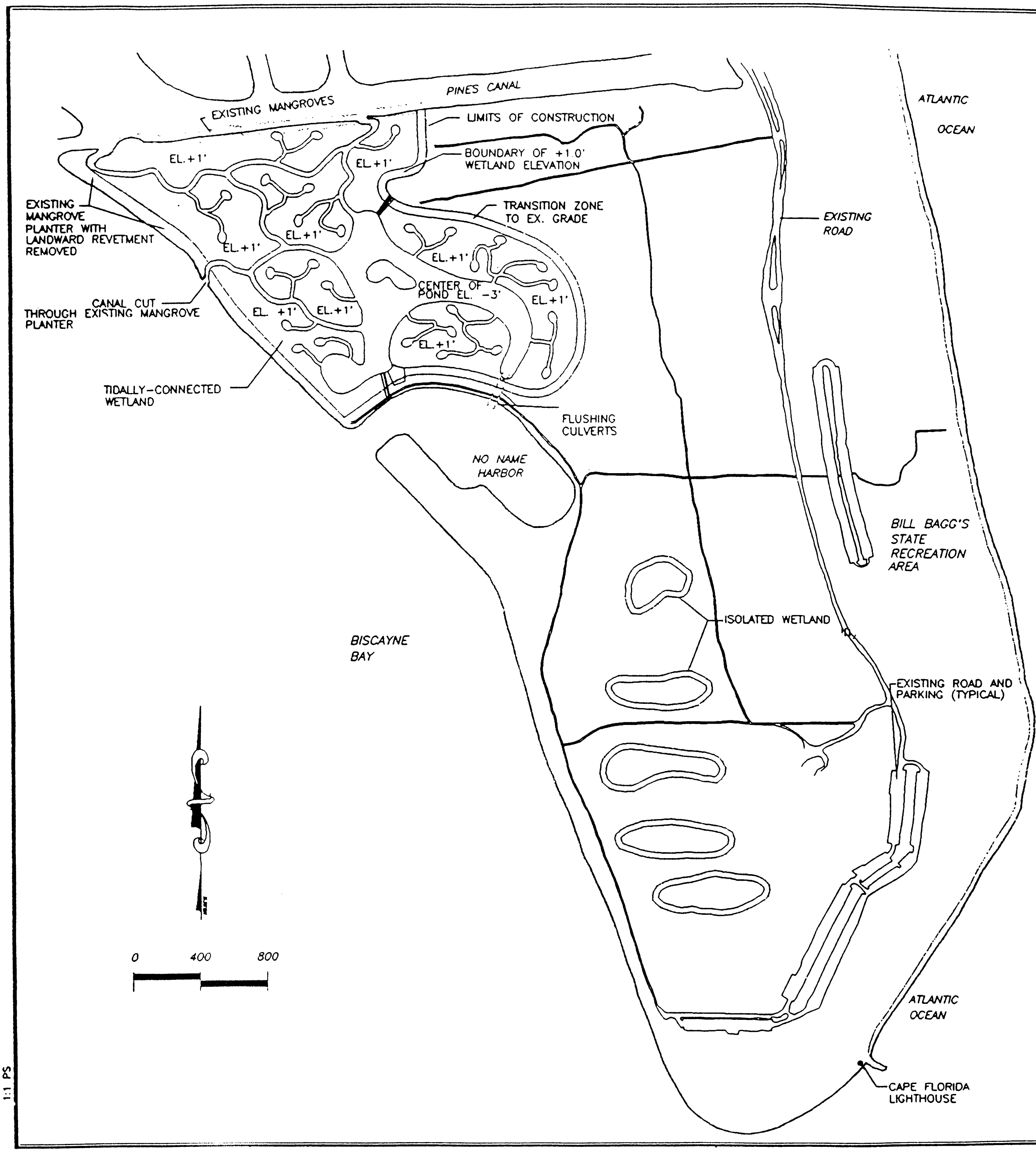
A topographic survey of the restoration area was prepared for the planning, design and construction phases of the project. The restoration area was surveyed topographically using the photogrammetric mapping method (Coastal Technology

Figure 1



ATLANTIC OCEAN
GULF OF MEXICO
KEY BISCAYNE
THIS PROJECT
HOLLYWOOD/ HALLANDALE
BROWARD CO
DADE CO
GOLDEN BEACH
NORTH MIAMI BEACH
A1A
826
N.E. 163rd ST.
BISCAYNE BLVD.
BAKERS HAULOVER CUT
N.E. 125th ST.
N.E. 79th ST.
1
95
BISCAYNE BAY
JULIA TUTTLE CAUSEWAY
195
VENETIAN CAUSEWAY
395
BISCAYNE BLVD
COLLINS AVE.
McCARTHUR CAUSEWAY
MIAMI
FISHER ISLAND
GOVERNMENT CUT
RICKENBACKER CAUSEWAY
VIRGINIA KEY
BEAR CUT
KEY BISCAYNE
PROJECT SITE
CAPE FLORIDA LIGHTHOUSE
ATLANTIC OCEAN

Figure 2


PINES CANAL
EXISTING MANGROVES
LIMITS OF CONSTRUCTION
BOUNDARY OF +1.0' WETLAND ELEVATION
TRANSITION ZONE TO EX. GRADE
EXISTING MANGROVE PLANTER WITH LANDWARD REVETMENT REMOVED
CANAL CUT THROUGH EXISTING MANGROVE PLANTER
EL.+1'
CENTER OF POND EL. -3'
TIDALLY-CONNECTED WETLAND
FLUSHING CULVERTS
NO NAME HARBOR
ATLANTIC OCEAN
EXISTING ROAD
BILL BAGG'S STATE RECREATION AREA
ISOLATED WETLAND
BISCAYNE BAY
EXISTING ROAD AND PARKING (TYPICAL)
0
400
800
ATLANTIC OCEAN
CAPE FLORIDA LIGHTHOUSE
1:1 PS

Corp., 1994). This cost-effective method was employed after the site was cleared of all exotic vegetation. The resulting topographic map, with contours at 0.15 meter (½ foot) intervals was super-imposed on a 2.54 cm = 61 meters (11, = 2001) scale aerial photograph of the restoration site.

A comprehensive biological assessment was conducted to document on-site and surrounding biological communities, to define biological goals and objectives, to identify environmental concerns, and to make specific recommendations concerning construction activities associated with the restoration.

Soil characteristics within the restoration area were determined by excavating 28 test pits at selected locations. A 152.5 meter (500') rectangular grid system was established in the footprint of the 304 square meter (0.075 acre) tidally connected wetlands in the northwest portion of the park. Test pits excavated by backhoe, located at each node of the grid, were sampled to analyze trends in vertical and horizontal distribution of soil strata. A soil-classification report for the site was developed to detail soil characteristics (e.g. type, grain size distribution, and color) and provide information applicable to developing marketing and spoil disposal strategies. In addition, ground penetrating radar and electronic surveying were used to provide data on subsurface conditions (Technos, Inc. 1994). These evaluations were used to locate the five historical isolated wetlands that had been filled to +1.98 meter (+6.5 feet) in the early 1950's.

Wave energy, tidal regime, current velocity and bathymetry surveys were conducted to assist in the development of design components such as .flushing canals (number, size, and depth), culverts (number, size, and elevation) and open water areas within the tidally connected wetlands. The final design for these components were evaluated using the Dynamic Estuary Hydrodynamic Model developed by the Environmental Protection Agency. Groundwater monitoring wells were installed and equipped with recorders to monitor seasonal fluctuations of groundwater. This was used to design elevations and contours of the five isolated freshwater wetlands which were restored in the park.

A two-phase archaeological monitoring work plan, was conducted at the restoration site by a qualities archaeologist. Phase I included the evaluation of a series of trenches throughout the restoration area, and Phase II consisted of daily observations of the excavation work during the restoration process. Archaeological evaluation during the excavation phase of the project revealed a 1,000 year old (B.P.) Human jawbone, along with an assortment of primitive conch shell tools. This is the oldest evidence of human habitation in this area (Zaminillo, 1997).

Restoration Implementation

The wetlands restoration plan was implemented via two separate Miami-Dade County construction contracts and three privately funded efforts. Federal, state and local environmental resource permits were obtained for all restoration work. The first element of the plan was to stabilize approximately 0.8 km, (one half mile) of high energy shoreline on the western boundary of the restoration area. This was accomplished through the first construction contract, which was executed in December 1993 and consisted of the installation of 16,535 metric tons (18,230 tons) of natural limerock boulders [30.5 cm. (12") to 76.2 cm. (30") in diameter] to create a 3.6 meter (12') wide x 1.22 meter (4') high rip rap revetment along the western boundary. The existing remnant concrete bulkhead along this shoreline was reduced to 0 National Geodetic Vertical Datum (NGVD) and utilized as additional material at the toe of the rip rap revetment. Limerock boulders were also placed along the seaward base of the concrete bulkhead for habitat and structural purposes. A 30.5 meter (100 foot) intervals along the bulkhead, a 1.53 meter (5') wide notch was cut to 0.3 meters (-1') NGVD to enhance flushing along the stabilizing structure. A 7.63 m. (25 foot) wide red mangrove (*Rhizophora mangle*) planter was installed at 0.4 meter (+1.3') NGVD elevation along the length of the rip rap revetment. A temporary limerock/filter fabric containment wall was installed on the landslide edge of the mangrove planter to contain the upland fill. The back wall was eventually recycled and utilized in the second construction contract to stabilize three flushing channels and two overlooks. The total cost of the first contract was $650,000, and was funded by the Florida Inland Navigation District and the Miami-Dade County Biscayne Bay Environmental Enhancement Trust Fund.

The second construction contract was executed in January 1996 and is expected to be completed by October 1998. The contract was subdivided into eight wetland components: Five 8.0 square meters (0.001 acre) freshwater isolated wetlands and three unequal areas [162 square meters (0.040 acres) , 64.75 square meters (0.016 acres) and 76.9 square meters (0.019 acres): 304 square meters (0.075 acre) total] of tidally connected wetland. The completed contract will result in:

Removal of 7,650 cubic meters (10,000 cubic yards) of solid waste

Removal of 344,250 meters (450,000 cubic yards) of dredge spoil material

Creation of 304 square meters (0.075 acre) of tidally, connected red mangrove (*Rhizophora mangle*)

Creation of a four square meter tern nesting island [1.4 m. (+4.51) elevation]

Creation of 16.2 square meters (0.004 acres) of open water area (-0.9' m. NGVD)

Installation of three floating water craft barriers at flushing connections

Installation of network of intertidal flushing creeks

Creation of 40.5 square meters (0.010 acres) of freshwater isolated wetlands

The total cost of the second contract is 1.9 million and was funded by the USDA Forest Service, South Florida Water Management District, Miami-Dade County Environmental Resources Management (DERM), Miami-Dade Water and Sewer Department, and the Village of Key Biscayne.

DISCUSSION

A total of 497,250 cubic meters (650,000 cubic yards) of fill material was excavated and transported to various locations in close proximity to the restoration site. The fill material was subdivided into three classifications. Type A consisted of approximately 229,500 cubic meters (300,000 cubic yards) of beach quality material. Approximately 76,500 cubic meters (100,000 cubic yards) of Type A material was recycled onto public beaches on Key Biscayne and Virginia Key, Florida. Private developers were responsible for the excavation and removal of approximately 153,000 cubic meters (200,000 cubic yards) of Type A material at no cost to the public. Type B, consisted of approximately 153, 000 cubic meters (200, 000 cubic yards) of high quality sand fill which was suitable for dune restoration or construction filling. Type C consisted of approximately 114,750 cubic meters (150,000 cubic yards) of material mixed with mulch, sandy humus, peat, and silt which was unsuitable for construction fill unless mixed with Type A or B material.

The 304 square meter (0.075 acre) marine wetlands [0.3 m. (+1.01) NGVD] are tidally connected to Biscayne Bay through three 3.7 m. (12') wide x -0.6 m. (2') NGVD flushing canals and a series of four [1.2 m. (4') diameter] culverts. The main canals (7.9 m. (26') wide x -0.9 m. (3') NGVD) interconnect with a 20.2 square meter (0.004 acre) open water area (0.9 m. (-3.0') NGVD] and twenty eight shallow pools [21.3 m. (70') diameter x 0.46 m. (-1.5') NGVD] via 6.1 m. (20') wide "feeder" canals [-0.3 m. (-1.0') NGVD]. The open water area and shallow pools provide low. energy habitat areas for larva, invertebrates, and juvenile fish. Additionally, the open shallow areas provide a sanctuary for wading birds.

The tidally-connected wetlands were planted with *Rhizophora mangle* on three-foot centers utilizing the construction contract and volunteers. *Avicennia germinans* and *Laguncularia racemosa* were not installed and recruited into the site through the tidal creeks. *Borrichia fructescens* and *Spartina spartinae* were planted around the edges of the wetland site above 0.76 m. (+2.5') NGVD. The five 8.1 square meter (0.002 acre) isolated freshwater wetlands [0.15 m. (+0.5') NGVD] were planted with *Acrostichum danaeifolium, Cladium jamaicensis, Eleocharis cellulose,* and *Spartina spartinae.*

Success criteria of the project are based on planting survivability and information regarding habitat use by fauna. To date, a 100% survival of wetland species is being realized. wildlife observations conducted by FDEP, and local environmental groups have documented an influx of fish and birds into the restoration area.

To date, 40 species of bird have been recorded using the saltwater wetlands, including 18 species shorebirds 7 species of egrets and herons and 4 species of terns Additionally, 12 species of birds have been documented utilizing the freshwater wetlands. Recently, a 4-foot crocodile was observed resting on the banks of the restoration area.

Public restoration dollars were maximized in this project through a resourceful spoil disposal plan, which reduced fill disposal distances and marketed fill to local developers. Fill materials of beach quality were recycled back onto local beaches. An estimated 2.8 million dollars were saved utilizing creative and resourceful project implementation strategies (e.g., phasing, spoil disposal strategies and resourceful construction strategies) . In addition, it should be noted that over 1500 volunteers have assisted with revegetating portions of tile restoration area with native wetlands vegetation, providing considerable cost savings.

The cost-effective restoration techniques reviewed in this paper have been developed through the DERM Biscayne Bay Coastal Habitat Restoration Program. Since 1985, DERM has restored and enhance- approximately 300 acres of coastal wetlands, created 38 acres of tropical hardwood hammock, created 15 acres of coastal strand community, created over one mile of dune community, enhanced and restored fourteen islands and stabilized over seven miles of unstable shoreline (Milano in prep).

ACKNOWLEDGMENTS

I wish to thank Dr. Susan Markley, Mr. Stephen Blair, Mr. Craig Grossenbacher, Mr.

Brian Flynn, Ms. Gwen Burzycki, Mr. Leonel Salgueiro and Ms. Valda Chin for their valuable assistance and support. I am grateful to Mr. David Crewz and Mr. Kevin Peters, Florida Department of Environmental Protection (FDEP), Marine Research Institute for their input and helpful critique of this work.

I also wish to thank the following Florida Department of Environmental Protection staff for their invaluable assistance: Dr. Renate Skinner, Mr. Dana Bryan, Mr. Lee Niblock, Mr. James Ross, Mr. Louis Scruggs, Mr. Stephen Gubitti, Ms. Elizabeth Carter-Golden, and Mr. Hank Smith.

My appreciation is extended to Mr.. Larry Resnick (deceased), Mr. Hernan Garces, Mr. John Ritsema and Mr. Anthony Canton, Miami-Dade Public Works Department, for their technical assistance with regard to contracts.

I express my appreciation to Coastal Systems International. (Mr. Harvey Sasso and Mr. Paul Lin) for their collaboration and support.

Thanks to the FDEP, South Florida Water Management District, and the Florida Inland Navigation for their continued funding support.

Special thanks are also extended to the many individuals in the community who volunteered their valuable weekends to plant native vegetation.

LITERATURE CITED

Coastal Technology Corporation. 1994. Cape Florida Wetlands Restoration Project: A report to Miami-Dade County Department of Environmental Resources Management.

Idyll, C.P., D.C. Tabb, and B. Yokel 1968. The value of estuaries to shrimp. Pp 83-90 In Newsome, J.D. (ed.), Marsh and Estuary Management Symposium, Louisiana State University Press, Baton Rouge.

Milano, G. R. (in preparation). Island restoration and enhancement in Biscayne Bay, Florida.

Milano, G. R. (in preparation). Restoration of coastal wetlands in Miami-Dade County, Florida.

Odum, W.E., C.C. McIvor, T.J. Smith III. 1982. The ecology of the mangroves of South Florida: A community profile. U.S. Fish and Wildlife Service, Office of Biological Services. FWS/OBS-81/24. Washington, D.C.

Technos Inc. 1994. Geophysical investigation of the near surface soils at Bill Baggs State Park, Key Biscayne, Florida. Technos project No. 94-118.

Zaminillo, J. 1997. Archaeological. monitoring and testing of the wetland restoration project at Bill Baggs Cape Florida State Recreation Area. Key Biscayne, Florida Archaeological and Historical Conservatory, Technical Report #166.

WINTER PERFORMANCE OF PLANTS IN TREATMENT WETLANDS

Thomas C. Holt, Brian K. Maynard, William A. Johnson, and John T. Bushoven
Department of Plant Sciences, University of Rhode Island
Kingston, RI 02881, U.S.A.

ABSTRACT

The use of wetland plants for constructed wetlands and filter strips is growing in popularity for the treatment of nonpoint source pollution. Most of the current literature on nutrient reduction by treatment wetlands evaluates summer performance. However, in temperate climates many plants go dormant in winter. A gap exists in the literature with regards to the year around performance of plant species used in treatment wetlands. We report on the ability of different wetland plants to take up nutrients from wastewater after periods of winter frost. Wetland mesocosms were built in temperature-controlled greenhouses to simulate winter temperature regimes that cause many wetland plant species to go dormant. Plants grown in triplicate wetland cells were fed an artificial wastewater and monitored for nitrogen and phosphorus reduction. They were then destructively harvested before and after periods of light (-2 °C) and heavy frost (-4 C). Growth rates, nutrient uptake, allocation of above and below-ground growth, and nutrient content of plant tissue were evaluated for Scirpus validus, Pontederia lancifolia, Iris pseudacorus, and Canna flaccida. Canna and Iris showed the best performance at removing nutrients from wastewater prior to the onset of frost, followed by Pontederia and Scirpus. After exposure to frost Pontederia was the most severely affected in both plant growth and nutrient uptake, while Scirpus seemed unaffected by frost.

INTRODUCTION

Degraded water quality is a growing concern across the nation. Non-point source pollution, such as runoff from barnyards, milking parlors, aquaculture facilities, and nurseries loads watersheds with nutrients, sediments, and pesticides, leads to the eutrophication of lakes and streams, and adds nitrates to drinking water supplies (Baker, 1992; USEPA, 1993).

Treatment wetlands have emerged as effective, low cost methods of water treatment which have the potential to reduce agricultural non-point source pollution and contribute to agricultural sustainability.

Plants play an important role in treatment wetlands by taking up nutrients and providing an extensive root zone which supports microbial attachment as well as the filtration of particulates (Brix, 1997; Gumbricht, 1993). Wetland plants also have the ability to transport air into their roots through aerenchyma tissue (Brix, 1987; Steinberg and Coonrad, 1994). Some of this oxygen leaks out of the roots, aiding in the decomposition of organic matter and the growth of nitrifying bacteria (Brix, 1987; Brix, 1993; Reddy et al., 1989). Nitrification, an aerobic process, is usually the limiting factor in the transformation of nitrogen in a wetland. Once ammonia is oxidized to nitrate, anaerobic denitrification readily takes place in wetlands (Reed and Brown, 1995).

Several studies have shown that a significant linear relationship exists between plant biomass and nutrient removal from wastewater (Kadlec and Knight 1996; Tanner 1996). Plants with higher biomass have a larger reserve for the storage of nutrients, as well as more root surface area for microbial attachment and mechanical filtration. This suggests that differences in environmental conditions that affect plant growth and biomass production would also affect nitrogen and phosphorus removal from wastewater.

Although there exists a significant amount of information on the ability of different plant species to remove nutrients from wetlands during the summer, few studies have measured the ability of different species of wetland plants to remove nutrients from wastewater throughout the winter season. The objective of this study is to determine the growth rate, nutrient uptake potential, and tissue nutrient content of different species of wetland plants before and after exposure to light (-2 ^{0}C) and heavy freezes (-4 ^{0}C).

METHODS

Vegetative propagules of yellow canna (Canna flaccida), yellow iris (Iris pseudacorus), bulrush (Scirpus validus), and pickerelweed (Pontederia lancifolia) were planted individually in 24-liter cylindrical wetland cells (0.07 m^2 x 0.4m) fitted with a central basal inflow (30mm i.d.) and peripheral surface outflow (30mm i.d.). Each cell was filled to within 3 cm of the top with 4-6 mm gravel while maintaining the water level 3cm below the top of the gravel. The interstitial water volume of each cell was 8 liters before planting. Unplanted wetland cells served as controls.

Experiments were carried out at the Agricultural Experimental Station at the University of Rhode Island (Lat. 42° 29'N). Plants were established from 10/1/97 to 2/28/98 in a heated greenhouse (mean temperature 22 ± 6 ^{0}C, range 14-38 ^{0}C) with

N-P-K fertilizer solution under 1000 watt metal halide lighting systems. Tap water was added daily via the central basal inflow to replace water lost by evapotranspiration. All plants had grown to maturity, flowered and seeded prior to the start of the experiment. On 3/1/98 three replicates of each species were destructively harvested and biomass determined (above- and below-ground tissues). Remaining cells were arranged randomly in replicates of three in temperature controlled greenhouses with one-third in a heated house (mean 24 $\pm$ 7 ^{o}C), range 16 to 38^{o} C (Fig. 1a), and the rest in a cool house (mean 11 $\pm$ 6 ^{o}C), range -4 to 22 ^{o}C (Fig. 1b). Half of the cells in the cool house were covered with polyethylene film on the coldest nights so that the minimum temperature did not fall below -2 ^{o}C. Air temperature was gradually lowered so that frost did not occur until 4/10/98 (Fig. 1b).

Cells were batch fed weekly via the central basal inflow tube an artificial wastewater solution 90 mg/l ammonia and 15 mg/l phosphate from ammonium sulfate, potassium phosphate, magnesium sulfate, and trace elements. From 4/16/98 to 4/22/98 (days 43 to 50, Fig. 1) water samples were collected twice, just after nutrient solution was added on day 1, and 7 days later. Samples were collected at the peripheral surface outlet by adding water through the central basal inflow tube to displace existing solution. A dye tracer used in a preliminary experiment verified that mixing between nutrient solution and displacement water did not occur. Water was analyzed (APHA, 1989) for nitrogen (ammonia and nitrate) and phosphate reduction (Table 1). Plant material was harvested, oven dried to determine biomass, and plant tissues were digested in acid and total tissue nitrogen and phosphorus content determined (Table 1). Biomass from the final harvest (4/23/98) was compared to the earlier harvest (3/1/98) to determine biomass increase in biomass during the experiment (Table 2). Total plant removal of nitrogen and phosphorus from the wetland cell was estimated by multiplying the increase in biomass by the nitrogen and phosphorus content of the plant tissue (Table 2). A Student's t-test was used to determine significant differences among biomass, tissue nutrient content, and nutrient removal from solution.

RESULTS

Nutrient uptake and biomass increase varied widely for the different species. Canna cells performed similar to iris cells which performed better than pickerelweed and bulrush cells in nutrient reduction (Table 1). After the first frost there were no significant differences among the nutrient removal of canna, iris, and bulrush, all of which performed better than pickerelweed.

Canna and iris cells in the cool greenhouse treatment had significantly smaller increase in biomass and estimated N and P removal from solution than those in the

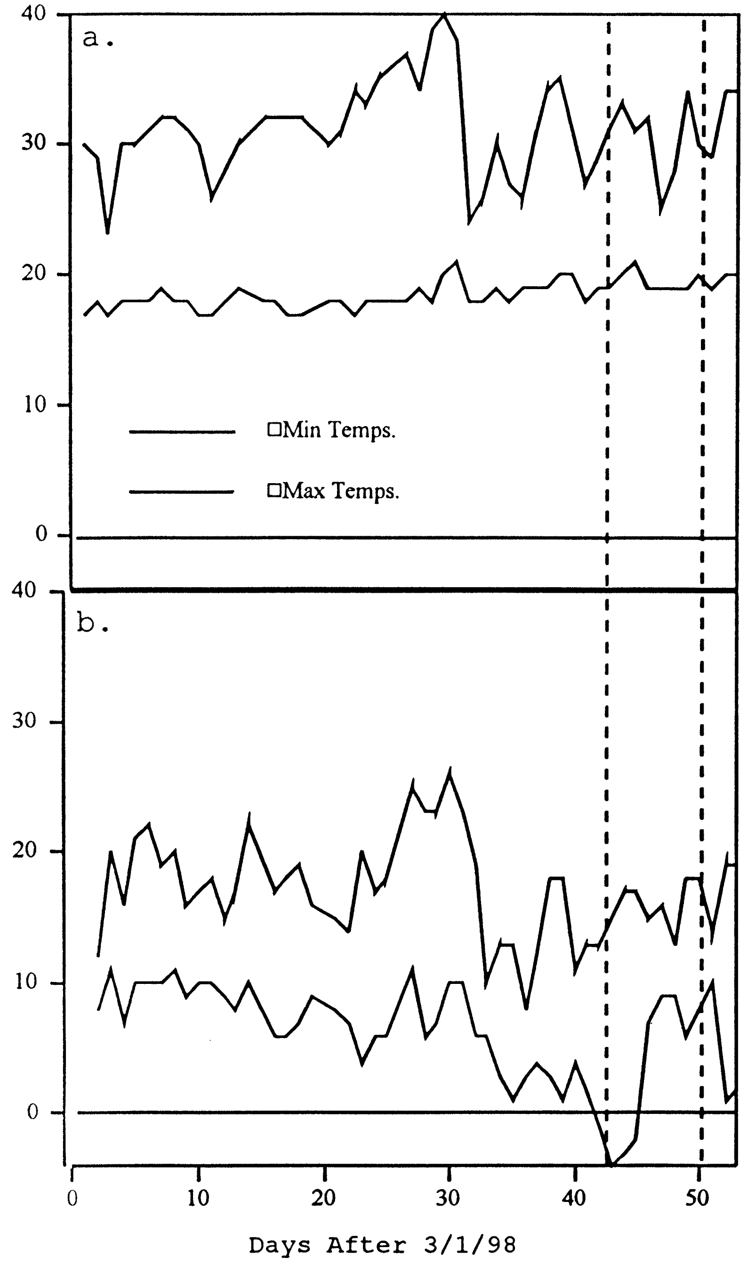


Fig. 1 The daily max/min temperatures (a) warm greenhouse (b) cool greenhouse during the experimental period.

Table 1: Comparison of nitrogen and phosphorus uptake by different wetland plant species at different temperatures from 4/16/98 to 4/22/98. Means represent three replicates with standard deviation in parentheses.

Plant species and temp. ranges	% Nitrogen removed	% Phosphorus removed	Plant Biomass (g)	Nitrogen content of plant tissue (mg/g dw)	Phosphorus content of plant tissue (mg/g dw)
temp. range 16-38 C					
canna	94 (2.5)	82 (3.2)	1096 (100)	26 (1.8)	3.4 (0.1)
yellow iris	91 (2.9)	84 (3.8)	1058 (105)	32 (1.9)	3.7 (0.2)
bulrush	69 (2.0)	49 (2.0)	675 (42.8)	18 (1.2)	2.4 (0.2)
pickerelweed	80 (2.1)	63 (2.3)	348 (38.0)	39 (2.6)	5.1 (0.3)
no plants	17 (0.5)	13 (2.0)			
temp. range -2-22 C					
canna	80 (2.9)	51 (3.6)	880 (59.1)	23 (2.1)	3.2 (0.2)
yellow iris	80 (3.0)	62 (2.7)	891 (50.3)	28 (0.9)	3.7 (0.3)
bulrush	50 (3.1)	44 (3.2)	685 (34.6)	19 (1.1)	2.3 (0.2)
pickerelweed	16 (2.0)	40 (3.1)	245 (30.4)	35 (3.2)	5.0 (0.3)
no plants	10 (1.9)	14 (1.3)			
temp. range -4-22 C					
canna	72 (3.6)	46 (2.3)	852 (48.5)	22 (2.3)	3.0 (0.2)
yellow iris	81 (4.1)	63 (4.1)	872 (53.7)	29 (2.7)	3.5 (0.3)
bulrush	62 (3.2)	47 (3.8)	725 (50.1)	18 (1.9)	2.3 (0.2)
pickerelweed	11 (4.0)	28 (3.3)	220 (32.2)	35 (3.0)	4.7 (0.3)
no plants	12 (1.1)	16 (0.5)			

Wetlands cells were batch fed a nutrient solution consisting of 90 mg/l NH_4 and 15 mg/l PO_4 with a retention time of 7 days.

Table 2. Estimates of nitrogen and phosphorus removal from 3/2/98 to 4/28/98 based on nutrient tissue composition at harvest. Means represent three replicates with standard deviation in parentheses.

Plant species and temp. ranges	Increase in biomass(g) from 3/2/98 to 4/28/98	Estimated nitrogen removal based on N tissue contents (g)	Estimated phosphorus removal based on P tissue contents (g)
temp. range 16-38 C			
canna	316 (32.7)	8.2	1.07
yellow Iris	298 (37.0)	9.5	1.10
bulrush	105 (10.1)	1.9	0.25
pickerelweed	108 (12.5)	4.2	0.55
temp. range -2-22 C			
canna	100 (8.2)	2.3	0.32
yellow iris	131 (9.1)	3.6	0.48
bulrush	115 (7.9)	2.2	0.27
pickerelweed	5 (1.1)	0.2	0.03
temp. range -4-22 C			
canna	88 (7.6)	1.94	0.26
yellow iris	112 (8.2)	3.20	0.39
bulrush	155 (15.9)	2.80	0.36
pickerelweed	-20 (2.1)	- 0.7	0.09

heated greenhouse treatment (Table 2). Above ground tissues of both species were visually damaged by frost with canna showing more visible leaf damage than iris. No damage was detected in underground tissues of either species. Estimated uptake of N and P by canna was 3 to 4 times higher, and by iris 2 to 3 times higher, in the heated than in the cool greenhouse treatment (Table 2).

Prior to frost, pickerelweed had the lowest increase in biomass (348 g). However, pickerelweed had the highest mean tissue nutrient content (39 mg/g DW N, 5.1 mg/g DW P). Pickerelweed performed very poorly at nutrient uptake once subjected to frost. Above-ground tissues were severely damaged by both light and heavy frost. Pickerelweed was the only species that lost biomass once subjected to frost (-20 g) (Table 2) and removed similar amounts of nutrients (11% N, 28% P) as the no plant control (12% N, 16% P). Upon harvest it was noted that roots growing from the rhizome had died and that new roots were starting to grow out of the rhizome in both cool greenhouse treatments. This was the only species in the experiment that exhibited visible root damage following frost.

Bulrush cells removed the fewest nutrients from solution and had the lowest biomass of any species tested prior to the onset of frost (Table 1). They also produced the lowest tissue nutrient content (18 mg/g DW N, 2.4 mg/g DW P). Even though the height of the bulrush canopy was close to 3m, the stems were hollow. Bulrush also did not produce as much below-ground biomass as canna and iris. Conversely, bulrush seemed unaffected by frost, with a biomass of 725 g and nutrient uptake of 62% N and 47% P in the coolest treatment and biomass of 675 g and nutrient removal of 69% N and 49% P in the heated greenhouse treatment. In the coolest treatment bulrush increased in biomass more than the other species, but because of its low tissue N and P levels, estimated removal of N and P uptake lagged behind canna and iris (Table 2). Bulrush exposed to frost was similar in appearance to bulrush growing in the heated greenhouse.

DISCUSSION

Results of this experiment show that different wetland plant species are affected differently by frost. Overall, canna and iris removed nutrients best both before and after frost. Bulrush was less affected by frosts, but still lagged behind iris and canna in nutrient uptake. Pickerelweed performed very poorly with the onset of frost.

The no plant control cells removed 14%-25% of the nitrogen and 15%-35% of the phosphorus in comparison to planted cells. Most of the phosphorus probably adhered to the microfilm of the gravel (Breen, 1990; Reddy and D'Angelo, 1997) while the nitrogen either adhered to the gravel or was lost through ammonia volatilization or bacterial nitrogen transformations (Breen, 1990; Reddy and D'Angelo, 1997). More nitrogen would be lost through microbial transformations if a carbon source had been added to the wastewater (Zhu and Sikora, 1995).

Because of their small size, water temperatures in our cells were probably lower than they would be in large scale wetlands. Four frost events in the range of -2C to -4C produced thin layers of ice which melted each day, but reformed at night. Larger wetlands presumably would have stored more heat energy during the day and not have had ice formation. It is therefore possible that the results of this experiment could be transferred to treatment wetlands exposed to air temperatures well below -4C.

All plants, with the exception of bulrush, had lower tissue nutrient content in treatments exposed to frost. This could be due to a reduction in young succulent growth and active meristematic tissue that is higher in tissue nutrient content than more mature growth (Kadlec and Knight, 1996; Mitsch and Gosselink, 1986). This young succulent growth is often the first to be damaged by frost. Bulrush did not appear to have any reduction in new tip growth following frost.

The results of these experiments might have differed in larger systems or with different starting material. For example, vegetative growth rates might have been affected by differences in initial propagule vigor, number of root meristems, or ecotypic growth characteristics (Daniels, 1991; McNaughton, 1966). Edge and container-effects might also be exaggerated in smaller wetland cells (Busnardo et al., 1992; Tanner, 1994). Shoots in smaller wetland cells are exposed to more solar radiation than in larger scale wetlands where more shading would occur. Growing roots deflect downward when they reach the edge of the smaller wetland cell. This creates a deeper root system which increases contact with the wastewater (Breen and Chick, 1995). However, the relative growth potentials, biomass, and tissue nutrient content seen in our research should be broadly indicative of species performance in other systems exposed to frost.

When wetland plants go dormant and leaf litter falls back into the wetland, much of the nutrients taken up by plants are released back into the water column upon decomposition (Hammer, 1997; Kadlec and Knight, 1996). However some of the nutrients may be stored in rhizomes of plants or deposited into detritus material that eventually gets converted into soil (Kadlec and Knight, 1996). By using plant material that does not go dormant in winter, or by combining multiple species plantings in a wetland where different species of plants enter dormancy at different stages in the fall, a sudden release of nutrients back into the wetland might be reduced or avoided.

Our research suggests that high-performing canna and iris could be good alternatives to commonly used species such as bulrush or cattail. In our study canna and iris were superior to bulrush in nutrient uptake and biomass production before the onset of frost and had similar growth rates and nutrient uptake after frost. Canna and iris also have significant ornamental value in foliage and flower. This could be important in areas where aesthetics should be considered. One reason iris and canna may not be more commonly used in treatment wetlands is that they do not typically form large monocultures in natural wetlands (personal observations). However monocultures can be created if plants are established in higher densities and weeds are removed or excluded. This is especially true in smaller scale wetlands. These ornamental species could also be planted on the fringe of larger scale wetlands. The less invasive growth habits of iris and canna may also allow for mixed species wetlands to be easily established and maintained.

ACKNOWLEDGEMENTS

The authors acknowledge the assistance provided by Jiang Zhongchun and the laboratory and supplies provided by Richard J. Hull and Micheal W. Sullivan.

REFERENCES

ASIWPCA. (1985). America's clean water: The State's evaluation of progress, 1972-1982. Association of State and Interstate Water Pollution Control Administrators, Washington, DC.

Baker, L.A. (1992). Introduction to nonpoint source pollution in the United States and prospects for wetland use. Ecol. Eng. 1: 1-26.

Breen, P.F. (1990). A mass balance method for assessing the potential of artificial wetlands for wastewater treatment. Water Res. 24(6): 689-697.

Breen, P.F. and A.J. Chick. (1995). Rootzone dynamics in constructed wetlands receiving wastewater: A comparison of vertical and horizontal flow systems. Water Sci. Technol. 32: 281-209.

Brix, H. (1987). Treatment of wastewater in the rhizosphere of wetland plants: The root zone method. Wat. Sci. Technol. 19: 107-118.

Brix, H. (1993). Macrophyte-mediated oxygen transfer in wetlands: transport mechanisms and rates. In G.A. Moshiri (ed.), Constructed Wetlands for Water Quality Improvement. Lewis Publishers, Boca Raton, FL, pp. 391-398.

Brix, H. (1997). Do macrophytes play a role in constructed treatment wetlands? Wat. Sci. Technol. 35(5): 11-17.

Busnardo, M.J., R.M. Gersberg, R. Langis, T.L. Sinicrope, and J.B. Zedler. (1992). Nitrogen and phosphorus removal by wetland mesocosms subjected to different hydroperiods. Ecol. Eng. 1:287-307.

Daniels, R.E., (1991). Variation in the performance of Phragmites australis in experimental culture. Aquat. Bot., 42: 41-48.

Gumbricht, T. (1993). Nutrient removal processes in freshwater submersed macrophyte systems. Ecol. Eng. 2: 1-30.

Hammer, D.A. (1997). Creating Freshwater Wetlands. Lewis Publishers, Boca Raton, FL.

Kadlec, R.H., and R.L. Knight. (1996). Treatment Wetlands. Lewis Publishers, Boca Raton, FL.

McNaughton, S.J. (1966). Ecotype function in the Typha community. Ecol. Monographs 36(4): 297-325.

Mitsch,W.J. and Gosselink, J.G. (1986). Wetlands. Van Nostrand Reinhold, New York pp. 277.

Reddy, K.R, E.M. D'Angelo and T.A. DeBusk. (1989). Oxygen transport through aquatic macrophytes: The role in wastewater treatment. J. Environ. Qual. 23: 907-913.

Reddy, K.R and E.M. D'Angelo (1997). Biogeochemical indicators to evaluate pollutant removal efficiency in constructed wetlands. Wat. Sci. Technol. 35: 1-10.

Reed, S.C. and D. Brown. (1995). Subsurface flow wetlands: A performance evaluation. Water Environ. Res., 67: 244-248.

Steinberg, S.L. and H.S. Coonrod. (1994). Oxidation of the root zone by aquatic plants growing in gravel-nutrient solution culture. J. Environ. Qual. 23: 907-913,

Tanner, C.C. (1994). Growth and nutrition of Schoenoplectus validus in agricultural wastewater. Aquat. Bot. 47: 131-153.

Tanner, C.C. (1996). Plants for constructed wetland treatment systems - A comparison of the growth and nutrient uptake of eight emergent species. Ecol. Eng. 7: 59-83.

U.S. Environmental Protection Agency. (1993). Guidance specifying management measures for sources of nonpoint pollution in coastal waters. U.S. Environmental Protection Agency, Office of Water. Washington DC. EPA-840-B-92-002.

Zhu, T. and F.J. Sikora. (1995). Ammonium and nitrate removal in vegetated and unvegetative gravel bed microcosm wetlands. Wat. Sci. Technol. 32: 219-228.

REPRODUCTIVE SUCCESS OF FLORIDA GRASSHOPPER SPARROWS IN SUMMER AND WINTER BURNED AREAS

Dustin W. Perkins[1], Peter D. Vickery[1,2]
Mark D. Scheuerell[3] and Bill Pranty[4]

[1]Department of Forestry and Wildlife Conservation, University of Massachusetts, Amherst, Massachusetts 01003
[2]Center For Biological Conservation, Massachusetts Audubon Society, South Great Road, Lincoln, Massachusetts 01773
[3]Department of Zoology, University of Washington, Box 351800, Seattle, Washington 98195-1800
[4]8515 Village Mill Row, Bayonet Point, Florida 34667-2662

ABSTRACT

The endangered Florida grasshopper sparrow (*Ammodramus savannarum floridanus*), an endemic subspecies to the Florida dry prairie, is now limited to only four known breeding locations. We think that improved management of this taxon is necessary for its recovery. Reproductive success was determined from 28 nests during the 1996 and 1997 breeding seasons at Three Lakes Wildlife Management Area in Osceola County. Based on the Mayfield method, nest success was calculated to be 22.3% in summer burned areas and 10.9% in winter burned areas. Annual productivity per pair was estimated to be between 2.78 and 3.48 fledglings in summer burned areas, and between 1.48 and 1.85 fledglings in winter burned areas. Depredation was the major cause of nest failure during this study. Low nest success rates were also recorded at Avon Park Air Force Range in Highlands and Polk counties, a site which only conducts winter burns. We estimated that in winter burned, and possibly also in summer burned areas there were not enough young being produced to maintain current population levels. Fires historically occurred in the summer, however prescribed fires are often conducted in winter to improve cattle forage and reduce nesting losses. Previous research has demonstrated that summer burns extend the length of the breeding season for Florida Grasshopper Sparrows. We think that these data combined with previous research gives reason to increase the frequency of prescribed burns to aid in the recovery of this unique taxon.

AN ENVIRONMENTAL ANALYSIS OF A POND NETWORK IN THE TOWN OF LINCOLN, MASSACHUSETTS.

David A. Krauss: Department of Biology, Boston College, Higgins Hall, 140 Commonwealth Ave, Chestnut Hill, MA 02167-3811

ABSTRACT

In 1996 the Lincoln Pond Commission in association with the Lincoln Land Conservation Trust commissioned a study of several large ponds in the Town of Lincoln, MA. As a follow up to that study, a survey of the smaller ponds in Lincoln was conducted during the summer of 1997. The results of an ecological assessment of thirty-seven ponds in the town are analyzed and discussed.

Lincoln is located in eastern Massachusetts and covers roughly 26 square kilometers. It is primarily a residential town with very little commercial zoning, no industry and a large proportion of conservation land. Lincoln has diverse ecosystems; the dominant community is eastern hardwood forest, but there are also significant areas of coniferous forest, meadows, pastures and hayfields. A network of natural and man-made ponds runs throughout the town involving several watersheds. A number of "species of concern" use the Lincoln ponds including rare species like the Spotted Salamander *(Ambystoma maculatum)* and common species with rapidly expanding populations such as Purple Loosestrife *(Lythrum salicaria)*.

The effects of the surrounding environments on the diversity and overall health of ponds in Lincoln were studied. Diversity in the communities surrounding ponds was reflected in the diversity of organisms found in the ponds themselves. Management strategies facilitating human use of ponds were largely inconsistent with those intended to maximize wildlife usage. Man-made ponds were generally found to be less diverse but more stable than natural ponds.

INTRODUCTION

In 1996 the Lincoln Pond Committee (LPC) in association with the Lincoln Land Conservation Trust (LLCT) commissioned a study of several large ponds in the Town of Lincoln, MA. As a follow up to that study, another study of the smaller ponds in Lincoln was commissioned for the summer of 1997. An ecological assessment of thirty-seven ponds in the town was conducted and an analysis of the results is published in this document.

Lincoln is located at approximately 71^0 10' W, 34^0 20' N in the Commonwealth of Massachusetts and covers roughly 26 square kilometers. It is primarily a residential town with very little commercial zoning and no heavy industry. There is still a significant proportion of farmland, made up of small privately or communally run farms, and a considerable amount of land is set aside as conservation land. Aside from the areas maintained as pasture land, the dominant ecosystem in Lincoln is hardwood forest, dominated primarily by oaks *(Quercus* sp.) and maples (Acer sp.). There is also a considerable proportion of coniferous forest dominated by White Pine *(Pinus strobus)* and Eastern Hemlock *(Tsuga canadensis)*. Wetland environments in Lincoln include rivers, streams, bogs, freshwater marshes, and ponds.

For a relatively small town, Lincoln has a large number of ponds, with all or part of 120 different ponds falling within the Town borders. The five largest ponds were assessed in the 1996 study along with 7 others. In 1997 37 smaller ponds were surveyed in order to create a baseline so that environmental changes through time at these and other ponds could be studied in the future. The ponds in Lincoln have formed through a variety of circumstances. Many are man-made, some are glacial kettleholes, a number are vernal pools, while others are simply depressions in drainage basins. Human usage of the ponds varies widely, from none at all on some isolated or privately held ponds, to intense on some public recreational ponds.

A number of "species of concern" are also present in and around the ponds of Lincoln. Many of these organisms are rare or protected species such as the Pink Ladyslipper *(Cypripaedium acaule*), Spotted Salamander *(Ambystoma maculatum),* and Eastern Bluebird *(Sialia sialis).* Other species of concern include common invasive with expanding populations, such as Purple Loosestrife *(Lythrum salicaria),* Buttonbush *(Cephalanthus* occidentalis), that may be displacing other species.

The purpose of this study was therefore multifold. First, to assess the environmental quality of the 37 ponds selected for the study. Secondly, to provide a baseline against which to compare other ponds and future environmental changes. Thirdly, to identify and evaluate any environmental patterns existing in the pond ecosystems in the Town of Lincoln. Finally, to evaluate the potential for future scientific research and the development of management strategies for Lincoln's ponds.

METHODS

A list of ponds to be surveyed was generated by the Lincoln Pond Committee. A total of 37 ponds were surveyed during the summer of 1997, and the results of these surveys are discussed below.

The basic field method involved a species level survey of the plants and animals present at each pond. Ponds were visited only once. A map was drawn showing the relative proportions of each of eleven different wildlife habitats present at the ponds border and within the 100ft. wetlands buffer zone. A transect was conducted around the circumference of each pond and all of the plant species encountered within 3 meters of the ponds' edge were identified using the guides and keys listed in the bibliography.

Although this survey was designed to be a botanical survey, all vertebrate species that were found to occur in the ponds' border zones were also listed. Direct observations and tracks and signs of animals were counted. These coincidental observations were included in order to generate as much information about the ponds as possible. Breeding birds were distinguished from non-breeding species. As a measure of invertebrate diversity, the species of Butterflies (Super-Family: Papilionoidea) and Dragonflies and Damselflies (Order: Odonata) present at the ponds were also listed. These groups were chosen because they are easy to spot and identify, and do not require the setting of traps. No attempt was made to census any species, only the presence or absence of species was noted.

The proportion of different plant communities surrounding the pond as well as the different forms of vegetation covering the pond were noted and mapped. The maps were then drawn on computer using Deneba "Canvas version 3.5.2" on a Macintosh Power PC 7200/120. The maps generically break up surrounding communities into hardwood forest, coniferous forest, brush, meadow, and lawn. Similarly, pond covering vegetation is listed as water lilies, duckweed, floating algae, and emergents. These maps, along with the detailed species list and watershed maps for each pond are available in Krauss, 1997.

In order to look for trends and patterns among the ponds of Lincoln, a series of statistical analyses were carried out. These analyses were a set of correlation analyses comparing different pond variables and were conducted using Statview SE+graphics on the computer described above. The variables used to indicate pond health include plant, vertebrate, breeding bird, and insect diversity, as well as total biodiversity. These diversity measurements are simple species richness measurements based on the survey data. Percentage of the pond covered by floating vegetation, emergent vegetation, Purple Loosestrife, Buttonbush, and open water were among the variables that could influence pond biodiversity. The number of different environments present along the pond border and in the buffer zone were also considered as possible determinants of biodiversity. Pond size was also considered to be a possible determinant of diversity and was based on the map area of ponds as calculated from the "Trail Map for Lincoln, Massachusetts."

RESULTS

Two hundred and sixty-three species of plants were found in and around the ponds surveyed in this project. Ten species of butterfly, 16 species of dragonfly (including damselflies), 6 species of amphibian, 4 reptiles and 8 mammals were also encountered. Of the 53 species of birds found around the ponds, 23 of them bred within the pond buffer zones. The number of plant species at individual ponds ranged from 17 to 84 with an average of 42.6. The number of vertebrate species ranged from 2 to 19 with an average of 9.7. The average number of breeding bird species was 1.4, ranging from 0 to 8.

Overall biodiversity varied widely from pond to pond and so a variety of statistics were computed to determine whether there were any important correlates of diversity acting around the ponds of Lincoln. All diversity measures described below are simply species richness measures and do not take into account the proportions of different species in each category. When discussing habitat or coverage diversity the diversity measure is simply the number of different environments present from the list in Appendix A. All statistics were computed using the data gathered on the 35 ponds surveyed in June. The results of all statistical tests conducted are reported in Table 1.

A variety of significant relationships emerged from these analyses. Plant diversity was central among these relationships. Plant diversity was found to be a significant determinant of animal diversity being significantly correlated with insect diversity [butterflies and dragonflies only(Fig.1)] and vertebrate diversity (Fig.2). Such a relationship is to be expected as animals ultimately depend on the vegetation around them for food and shelter.

Environmental diversity within the buffer zone (Fig.3) and at the pond borders (Fig.4) were both significantly correlated with plant diversity. This may seem like a spurious correlation since several of the potential pond environments are defined by their plant communities. However, since each of these plant communities (meadow, brush, hardwood forest, coniferous forest) reflect different sets of environmental conditions that can vary within a few meters, such as drainage and disturbance regimes, the statistic is not meaningless. The strong positive correlations shown by these statistics indicate that where ponds form under diverse physical conditions, rather than uniform conditions, they will have a more complex vegetation.

The Relationship Between Plant Diversity and Insect Diversity

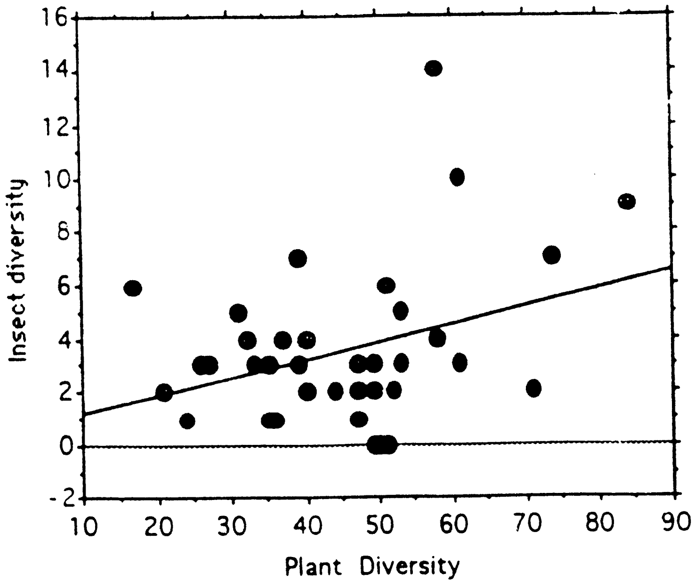


Figure 1. The number of plant species at a pond is a significant determinant of the number of insect species at the pond (DF = 1, 34; F-value = 4.3; p-value = 0.046). Both aquatic and terrestrial plants are included in this statistic, and only butterflies, dragonflies, and damselflies are considered in the insect species measure.

The Relationship Between Plant Diversity and Vertebrate Diversity

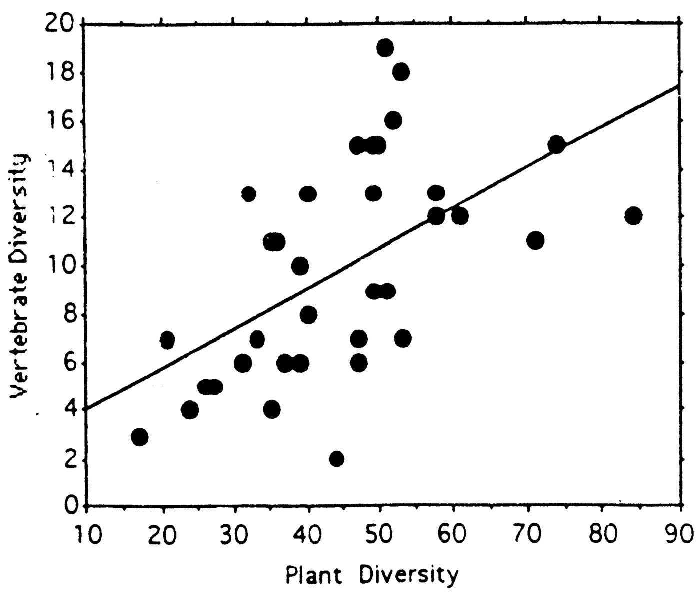


Figure 2. The number of plant species at a pond is a significant determinant of the number of vertebrate species at the pond (DF = 1, 43; F-value = 15.96; p-value = 0.003). All species of plants and vertebrates are included in this statistic.

Table 1

Variable	Pond Size F-value	P-value	Diversity of Border F-value	P-value	Diversity of Buffer F-value	P-value	Diversity of Pond Cover F-value	P-value
Plant Diversity	1.5	0.24	11.03	0.002	5.7	0.02	0.53	0.47
Vertebrate Diversity	2.1	0.15	3.03	0.09	4.1	0.048	3.28	0.08
Breeding Bird Diversity	4.2	0.049	0.01	0.92	0.32	0.57	1.25	0.27
Insect Diversity	1.5	0.23	0.39	0.53	0.08	0.78	1.78	0.68
Total Biodiversity	2.66	0.112	7.57	0.009	5.22	0.03	0.99	0.33
% open water	0.01	0.91	0.02	0.9	0.46	0.5	11.45	0.002
% Floating Cover	0.25	0.62	0.14	0.72	0.003	0.95	4.75	0.04
% Emergent Cover	0.03	0.88	0.12	0.73	0.5	0.48	2.97	0.09
% Loosestrife	0.0004	0.98	0.41	0.53	0.3	0.58	0.004	0.95
% Buttonbush	2.22	0.14	0.06	0.8	0.07	0.79	2.33	0.14
Pond Cover Diversity	0.06	0.81	0.01	0.91	0.02	0.89		
Border Diversity	0.69	0.41						
Buffer Diversity	1.97	0.17						

Variable	Plant Diversty F-value	P-value	% Floating Cover F-value	P-value	% Emergent Cover F-value	P-value	% Purple Loosestrife F-value	P-value
Plant Diversity	****	****	0.88	0.36	1.44	0.24	0.76	0.39
Vertebrate Diversity	15.96	0.0003	1.48	0.23	1.75	0.2	0.15	0.7
Breeding Bird Diversity	1.58	0.22	0.95	0.34	0.02	0.88	0.04	0.84
Insect Diversity	4.3	0.046	1.48	0.23	0.03	0.85	1.03	0.32
Total Biodiversity	****	****	1.62	0.21	0.15	0.23	0.37	0.54

In table one are compiled the results of the linear regression analyses performed in this investigation. As all analyses were performed onf a sample of 35 ponds all have 1,34 degrees of freedom. Any p-value less than 0.05 was considered significant. Those few tests that showed significant correlations are illustrated in accompanying figures.

The Relationship Between Environmental Diversity in a Pond's Buffer Zone and It's Plant Diversity

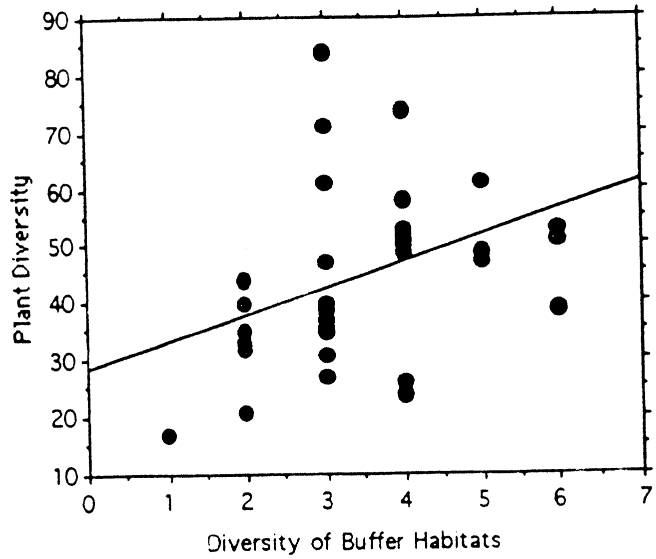


Figure 3. The number of different habitats within the ponds' 100 ft. buffer zone is significantly correlated with the plant diversity at each pond (DF = 1, 34; F-value = 5.7; p-value = 0.02).

The Relationship Between Environmental Diversity on a Pond's Border and It's Plant Diversity

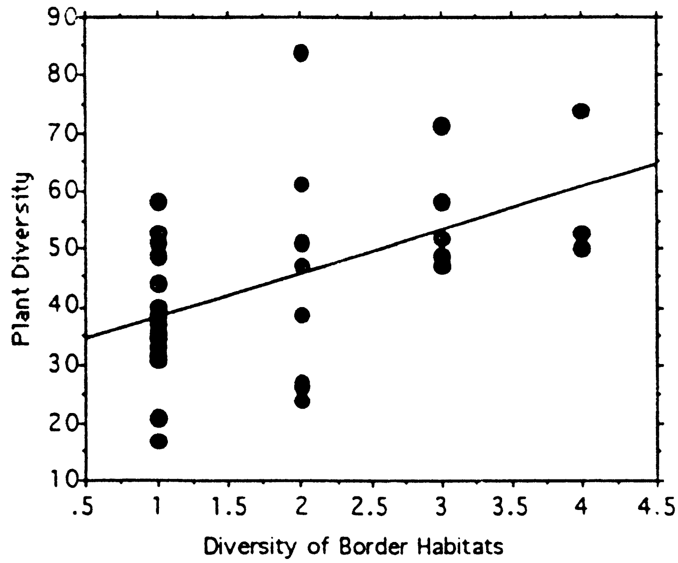


Figure 4. The number of different habitats at each pond's border is significantly correlated with the plant diversity of the ponds pond (DF = 1, 34; F-value = 11.03; p-value = 0.002). There is some auto-correlation because different habitats are defined by plant communities.

By inference one would expect the diversity of vertebrates and insects to also be correlated with the diversity of buffer zone and pond border environments as well. In fact, vertebrate diversity does correlate significantly with buffer zone diversity (Fig.5), but while vertebrate diversity does increase as pond border diversity increases this relationship falls short of statistical significance ($p=0.09$). Insect diversity was not significantly correlated with either above measure of environmental diversity. This result may be due to the fact that Odonatates are primarily influenced by the aquatic environment, rather than the terrestrial environment. Total biodiversity was strongly correlated with both measures of environmental diversity (Fig. 6 & 7). As total biodiversity is strongly influenced by plant diversity (most of the species found at all ponds are plants) this result is not surprising.

The Relationship Between Environmental Diversity in a Pond's Buffer Zone and It's Vertebrate Diversity

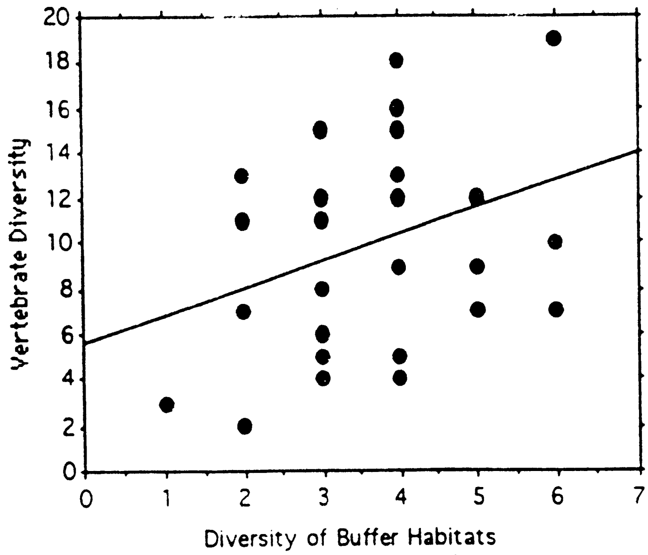


Figure 5. The number of different habitats within the ponds' 100 ft. buffer zones is significantly correlated with the vertebrate diversity at each pond (DF = 1, 34; F-value = 4.1; p-value = 0.048). A greater habitat diversity in the area around a pond allows more different species of vertebrates to use the pond.

The Relationship Between Environmental Diversity in a Pond's Buffer Zone and It's Total Biodiversitv

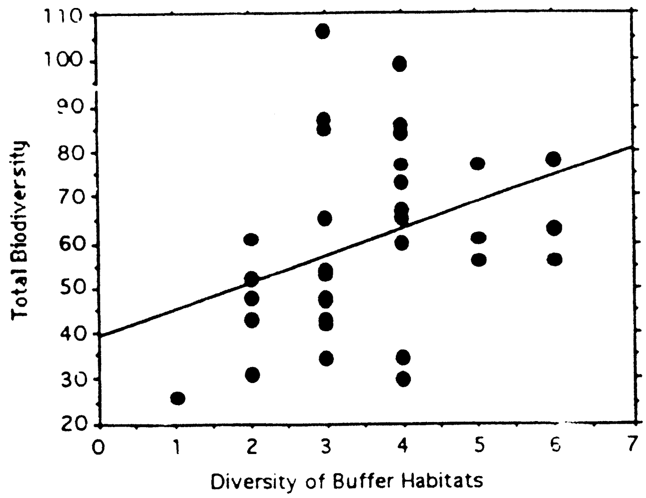


Figure 6. The number of different habitats within the ponds' 100 ft. buffer zones is significantly correlated with the total biodiversity at each pond (DF = 1, 34; F-value = 5.22; p-value = 0.03). The greater the environmental diversity around each pond results in greater biodiversity at each pond. The total diversity measure is heavily influenced by plant diversity.

The Relationship Between Environmental Diversity on a Pond's Border and It's Total Biodiversity

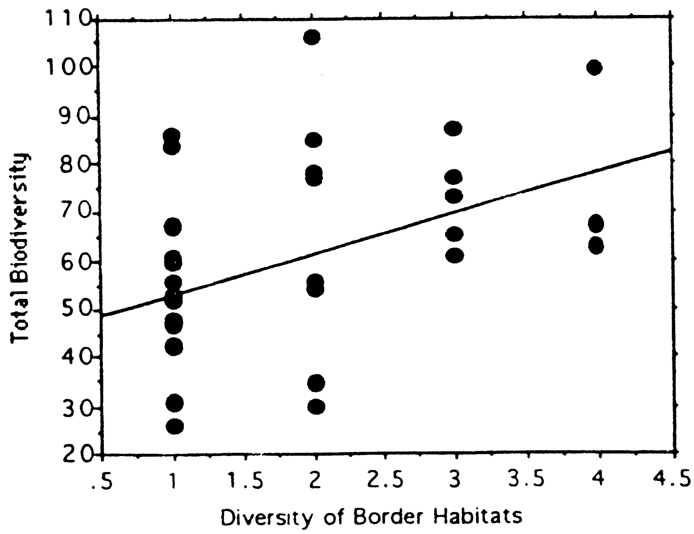


Figure 7. The number of different habitats along the ponds' border is significantly correlated with the total biodiversity at each pond (DF = 1, 34; F-value = 7.57; p-value = 0.009). The greater the environmental diversity around each pond, the greater the biodiversity at each pond. The total diversity measure is heavily influenced by plant diversity.

Pond size did not turn out to be an important correlate of biodiversity. The size of the ponds, which ranged from approximately 30 to 11,000 square meters, was not significantly correlated with any measure of environmental diversity or with plant, vertebrate or insect diversity (Table 1). Pond size was, however, significantly correlated with breeding bird diversity (Fig. 8). Since pond size does not correlate with habitat diversity and none of the habitat diversity measures correlate with breeding bird diversity, it seems logical to conclude that the relevant issue here is territory size. Smaller ponds simply do not have enough space to allow a high diversity of breeding birds. This conclusion may be in error, as it is only the presence of two outlying points that create significance. When they are removed from the analysis the level of results of the regression analysis drop well below significance (DF = 1,32; F-value = .43; p-value = .51).

Breeding Bird Diversity as a Function of Pond Size

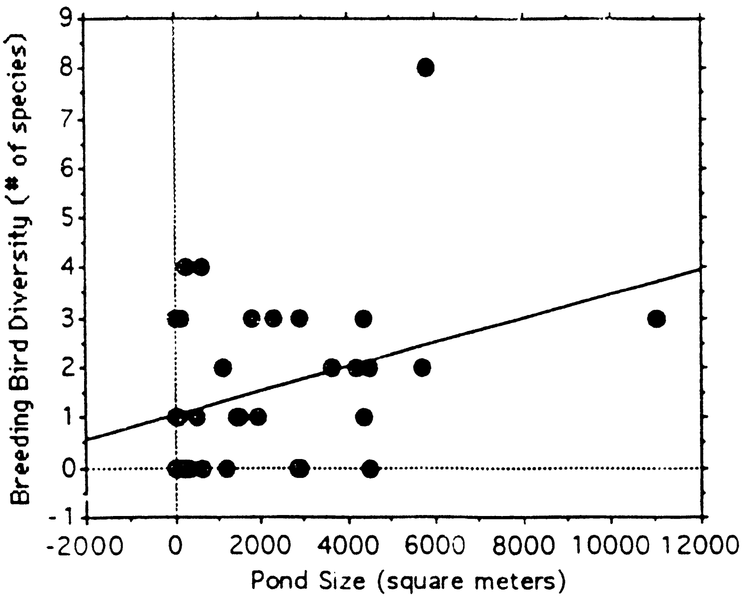


Figure 8. The number of breeding bird species found at a pond is significantly correlated (DF = 1, 34; F-value = 4.2; p-value = 0.049) with the size of the pond. When the outlying points are removed from the analysis the relationship disappears (DF 1, 32; F-value = 0.43; p-value = 0.51).

Although the aquatic vegetation was included in the plant diversity measure, it was also treated seperately. Very few patterns emerged with respect to aquatic vegetation in the Lincoln ponds. The diversity of pond cover, measured as a number of species, was correlated with the percentage of pond surface covered with floating vegetation (Fig. 9). This phenomenon suggests that the diversity of emergent vegetation is relatively consistent from pond to pond and it is the variations in the floating plant biota that primarily account for differences in aquatic vegetative diversity between ponds. The diversity of pond cover was found to have a strong inverse correlation with the percentage of open water in a pond (Fig.10). This correlation suggests that ponds are rarely filled in by monocultures. Instead, as open water is covered up a variety of new species enter the pond increasing overall diversity.

Purple Loosestrife and Buttonbush are both highly invasive species that are a concern for pond managers. Although Purple Loosestrife is common throughout Lincoln, it did not occur in particularly large quantities anywhere. The fact that no significant relationship was found between the occurrence of Purple Loosestrife and any diversity measure indicates that it does not seem to be displacing other species in the Town. Buttonbush was not found at enough ponds in this survey to perform any statistical analyses. Likewise, Water Chestnut was encountered too infrequently to draw any conclusions.

The Relationship Between the Diversity of Pond Cover and the Percentage of Floating Plants on a Pond

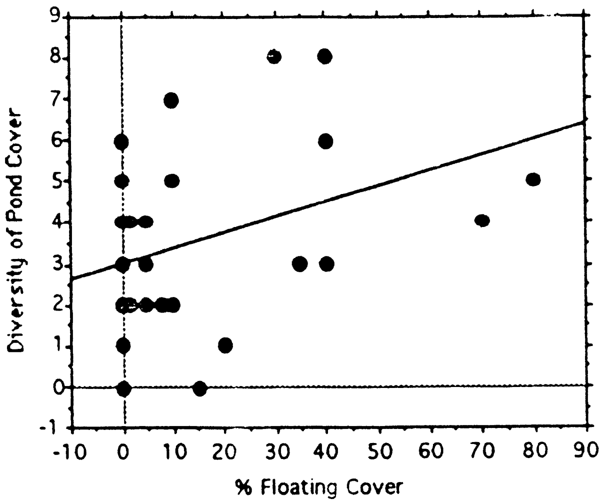


Figure 9. As the percentage of floating cover on a pond increases the diversity of the pond's cover also increases (DF = 1, 34; F-value =4.75; p-value = 0.04), suggesting that the domination of ponds by monocultures is unusual.

The Relationship Between the Diversity of Pond Cover and the Percentage of Open Water on a Pond

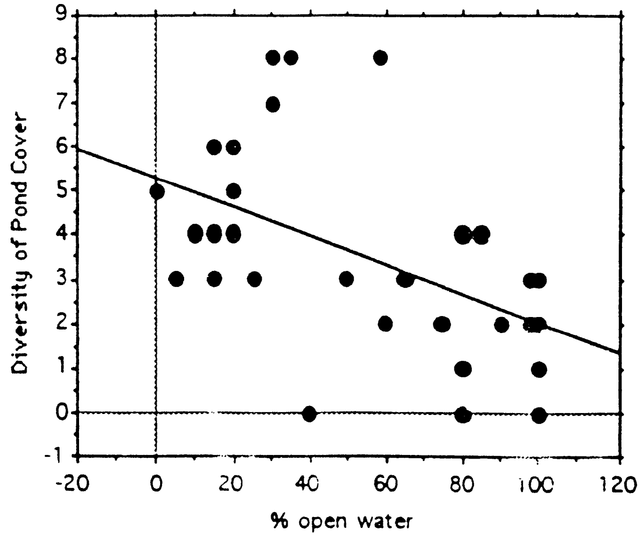


Figure 10. As the percentage of open water on a pond decreases the diversity of pond cover increases (DF = 1, 34; F-value = 11.45; p-value = 0.002), demonstrating that large expanses of open water indicate low diversity environments.

DISCUSSION

Total biodiversity at Lincoln ponds is strongly influenced by the diversity of plant life at the ponds. This can be of use when designing management strategies for conservation lands. If maintaining maximum biodiversity is the goal of conservation in Lincoln (which is by no means certain) then managing to increase plant diversity is the most effective way to achieve it. By creating environments that foster a high botanical diversity, increases in zoological diversity will follow. Although such a desire is often the goal of modern conservationists, it is not always desirable. It may be preferable to maintain classic examples of a particular ecosystem, such as old growth hardwood forest,, with the species that require the specific conditions therein. If preserving large areas of specific communities is the conservation goal then a sub-maximum biodiversity must be expected.

As plant diversity turned out to be one of the most important aspects of the pond environments it was important to determine what factors influenced it. Surprisingly, pond size does not play an important role in determining the biodiversity of a pond. Larger ponds have more area and longer perimeters and are likely to intersect more different sets of physical conditions, thus creating more niches and allowing more species to exist. This is a classical relationship between size and biodiversity.

Although larger ponds often had more species than smaller ponds in Lincoln, statistical analysis showed that this relationship was not significant. The fact that pond size was significantly correlated with the diversity of breeding birds probably reflects an areal factor rather than a niche diversity factor. There is simply room for more breeding territories along a larger pond and a greater variety of species are likely to occupy them due to stochastic processes.

Diversity of habitats around each pond, both immediately touching the pond (border habitats) and within the ponds, 100ft. buffer zones (buffer habitats) was the primary factor determining plant diversity and hence animal diversity at each pond. This is a difficult correlation to deal with as it may be a spurious autocorrelation. Habitats were defined by their plant communities, e.g. hardwood forest and meadow. Since a greater collection of plant communities will naturally have a greater number of plant species, this correlation could be dismissed as meaningless. However, different plant communities grow in different locations because of specific local conditions that allow a particular set of plants to thrive. If the number of habitats (plant communities) is taken as a measure of the variance of physical condition found at a pond (such as disturbance regimes, soil structure, ground water flow, etc.) then this statistic has meaning. Ponds with a greater variety of environmental conditions will tend to have higher biodiversity. This fact can be an important consideration in planning management strategies.

It is the nature of the pond ecosystem to eventually fill with plants, then sediments forming a meadow and ultimately succeed to forest (Ricklefs, 1990). Several of the ponds surveyed here are in advanced stages of succession. As plant stems increase in the open water new physical habitats are created both above and below the surface, creating conditions favorable to some species. At the same time, submergent species may be shaded out and habitat for some species lost. These relationships are complex and a detailed investigation of them was well beyond the scope of this study. However, two interesting relationships did emerge.

As the proportion of open water on a pond decreases the diversity of pond cover (the number of both emergent and floating species) increases. This relationship may seem self evident, but it brings out an important point. As a pond fills with vegetation, typically, it is filled by a wide variety of species rather than being taken over by a monoculture of one species. It was also found that as the diversity of pond cover increased the proportion of floating cover also increased. An increase in pond cover diversity was not noted to correlate with an increase in the proportion of emergent cover in a pond. Taken together, these facts suggest that floating cover is likely to have a greater variance than emergent cover. Or put another way, if the pond is taken over by a monoculture (or just a few species) it is more likely to be a monoculture of an emergent species than a floating species. This conclusion, however, seems counter-intuitive when the fact that there are more emergent species overall and that floating species such as duckweed and water lilies tend to form monocultures. It therefore seems logical to suggest that floating species tend to take a more significant role later in the filling process when a wide variety of aquatic plants have had an opportunity to become established in a pond. Investigation of this hypothesis is currently planned as a long term research project in Lincoln.

Purple Loosestrife is a highly invasive introduced species that is of great concern in New England. Although it was found to be widespread among the Lincoln ponds, statistical analysis showed that it is not choking out other species as it is not in any way related to a decrease in biodiversity at the ponds. This finding is disbelieved by many of the Pond Committee members who wanted this study to show that Purple Loosestrife was excluding native plants. Such a result would have justified efforts to eradicate the plant from town. Although this author agrees that removing Purple Loosestrife is desirable the findings of this study can not provide the sort of data necessary to describe the invasive nature of the plant.

The Lincoln pond network provides some excellent opportunities for academic research. The wide variety of surroundings, use patterns, sizes, and water flow regimes would allow many interesting experiments and observations to be conducted. The presence of so many ponds on conservation land managed by research friendly organizations creates the potential to set up long term research projects investigating ecological processes and environmental management strategies that could provide highly valuable data. Establishing a relationship between an academic institution with an active environmental studies program and the LPC, LLCT, and LCC could be a valuable asset to the town. Several such institutions exist in the area and in conjunction with the local schools could create an outstanding long term research facility.

In terms of environmental management, the most important point to come out of this study is the need to develop a set of goals for the Lincoln ponds. If preserving examples of plant communities is the goal then allowing succession to occur or instituting disturbance regimes according to the community desired would be the appropriate course. If increasing biodiversity is the goal then a fairly constant management to promote a variety of communities will be necessary. Balancing these decisions with the need for human access to the ponds for recreational use is always a difficult policy decision. Unfortunately no clear relationship between human pond usage patterns and biodiversity emerged from this study. Several ponds will need human intervention within the next decade or two if they are to remain ponds. The development of a clear set of conservation goals is the first crucial step towards developing a comprehensive management strategy for the Lincoln Ponds.

ACKNOWLEDGMENTS

This project was funded by the Lincoln Land Conservation Trust. The assistance of the Lincoln Pond Committee and Conservation Commission was invaluable. Buzz Constable, Nadie Rice, Tom Walker, Jane Layton, Sue and Chris Klem all provided assistance. Geoff McGean of the Lincoln Conservation Commission provided logistical support that aided in the completion of this project.

LITERATURE CITED

Brown, Lauren 1979. Grasses An Identification Guide. Houghton Mifflin Company, Boston. 240 pp.

Burt, Richard & Richard P. Grossenheider 1976. A Field Guide to the Mammals of North America North of Mexico. Houghton Mifflin Company, Boston. 289 pp.

Carpenter, Virginia 1991. Dragonflies and Damselflies of Cape Cod. The Cape Cod Museum of Natural History, Brewster, MA. 79 pp.

Cobb, Boughton 1963. A Field Guide to the Ferns and Their Related Families. Houghton Mifflin Company, Boston. 281 pp.

Conant, Roger 1975. Reptiles and Amphibians: Eastern/Central North America. Houghton Mifflin Company, Boston. 429 pp Hallowell, Anne C. & Barbara G. Hallowell 1981.Fern Finder. Nature Study Guild, Berkeley. 62 pp.

Klem, Chris 1996. Report of the Lincoln Pond Committee to the Lincoln Land Conservation Trust and The Lincoln Conservation Commission. Internal Report.

Klots, Alexander B. 1951. A Field Guide to the Butterflies of North American East of the Great Plains. Houghton Mifflin Company, Boston. 349 pp.

Krauss, David A. 1997 An Ecological Analysis of Ponds in the Town of Lincoln, MA. Lincoln Conservation Commission, Lincoln, MA. 203 pp.

National Geographic Society 1987. Field Guide to the Birds of North America. National Geographic Society, Washington, D.C. 464 pp.

Newcomb, Lawrence 1977. Newcomb's Wildflower Guide. Little, Brown and Company, Boston. 490 pp.

Peterson, Roger Tory & Margaret McKenny 1968. A Field Guide to the Wildflowers of Northeastern and North-Central North America. Houghton Mifflin Company, Boston. 420 pp.

Petrides, George A. 1972. A Field Guide to Trees and Shrubs 2nd edition. Houghton Mifflin Company, Boston. 428 pp.

Ricklefs, Robert E. 1990. Ecology. W.H. Freeman and Co, New York. 896 pp.

WETLAND RESTORATION IN PUERTO RICO WITH EXOTIC EQUIVALENTS OF EXTINCT KEYSTONE VERTEBRATES.

By Francisco Watlington, Ph.D.
Department of Geography
University of Puerto Rico

ABSTRACT

Restoration of viable freshwater wetlands in Puerto Rico is obstructed by faunal impoverishment and the dogmatic opposition of some conservationists to the naturalization of exotic species. However, paleobiogeographic evidence of extinct keystone vertebrates provides an enlightening frame of reference in identifying appropriate species for vacant ecological niches in depauperate island ecosystems.

In the early seventies an awakening public concern with surface and ground water pollution was decisive in the establishment of the Puerto Rico Department of Natural and Environmental Resources (DNER). An early consultant publicly lamented the absence in local freshwater wetlands of a long-lived top-of-the-food-chain carnivore which might serve to monitor biomagnification of toxic wastes. However, at that very time, recently introduced caimans (*Caiman crocodilus*) were quietly proliferating in the marshes around Lake Tortuguero on the north-central coast of Puerto Rico.

Discovery of the caimans led to an ill-advised and futile policy of eradication in the mid-eighties. A random analysis of caiman viscera in the DNER lab revealed a high concentration of mercury. The finding was suppressed along with the ecological observation that the caimans were feeding mainly on shoals of introduced Tilapia that had virtually displaced the native fishes. While the underlying reasons for instatement and maintenance of a "control" attitude toward the caiman are unclear, official justification invokes a nativist rationale couched in the tenets of the equilibrium theory of island biogeography.

The theory holds that insular species and ecosystems are intrinsically fragile and vulnerable to onslaught by alien invasions. Thus, hapless endemics must be protected from adaptively superior exotics. Island equilibrium theory has been challenged on diverse grounds, including incongruity with historical biogeography. In the caiman case, fossil remains conjoin paleogeographic and regional biogeographic evidence in support of the conclusion that crocodilians were abundant in prehistoric Puerto Rico.

An overview of the extinct endemic caviomorph rodents that once inhabited the Greater Antilles further suggests that there were also wetland herbivores related to modern coypu and capybara.

INTRODUCTION

Puerto Rico, the smallest of the Greater Antilles (GA), is the most densely settled and urbanized polity of the archipelago, with a population of over 3.5 million persons compressed into 8,900 k^2 for a general density approaching 400 inhabitants per k^2. Except for fragmentary areas in upland rainforests and karst topography, extensive freshwater swamps and their brackish littoral borders comprise roughly 3% to 5% of the coastal plains and inland valleys (about two thirds of the island, or some 590,000 ha). This is also the most heavily urbanized portion of the island, where competition for available geographic space is most intense.

The urban revolution that ended the demand for fuelwood and dependence on subsistence agriculture has fostered spontaneous reforestation of the mountainous third of the island, over 300,000 ha (Birdsey and Weaver 1987). The lowlands, on the other hand, have been transformed into a dynamic patchwork of urban areas, parklands, grasslands, croplands, interstitial hedgerows and landfills. Wetlands take up the only major expanses of virtually uncommitted buffer space that remain in the metropolitan continuum that laces the island.

In recent years diverse interests have converged to impel a growing grassroots movement in pursuit of ecosystem restoration in Puerto Rico, with clear emphasis on wetlands. The unsustainable indebtness of the old (agricultural) land authority (Autoridad de Tierras) that owns or controls most of the marsh-prone areas and runs the heavily subsidized drainage programs is a key factor. However, the extensive holdings are viewed by the Commonwealth as a hedge against the impending collapse of the welfare state. On the other hand, rising public awareness of the potential importance of wetlands in attainment of desirable environmental objectives has inspired a number of academic and community-based groups that are pressing the government to support ambitious restoration projects.

Although freshwater marshes are surely the most degraded natural ecosystems in Puerto Rico, they are probably also the most indestructible. Some are the periodically inundated overflow basins of major rivers. Others are coastal impoundments uneasily resisting saltwater intrusion: permanent lakes, canal systems and bogs, spring-fed from perched aquifers. Most have been artificially reduced to a fraction of their former extent by so-called "land reclamation" projects featuring engineered works such as drainage channels, dikes and pumping stations, and remain contained only at exorbitant cost. Despite all investments the marshes are restored, if only for a short time, whenever pumps break down or unusually heavy rains occur.

Long-term restoration is obstructed by, among other things, a depauperate aquatic vegetation and lack of key vertebrate fauna, both herbivores and predators, compounded by opposition from entrenched professional conservationists to recognition of naturalized exotics as ecologically valid ecosystem components. The theoretical rationale behind the antagonism is a questionable aggregate of ecological theories and supportive anecdotal "just so" stories that purport to prove that endemic island wildlife, having evolved in overprotective isolation, are intrinsically vulnerable to competitive and zoonotic onslaught by exotics, especially those evolved on continents amid vastly greater numbers of competitors, predators and parasites.

Thus, hapless endemic species must be protected from adaptively superior foreigners by eradicating the latter. The conservation paradigm that justifies a prescriptive bioxenophobia has been referred to in the literature as the equilibrium theory of island biogeography. In all fairness, deep and diverse roots converge in the famous nomothetic synthesis by MacArthur and Wilson (1967). In Puerto Rico, island equilibrium theory (IET) has been institutionalized in statutory regulations and administrative policies of the Department of Natural and Environmental Resources (DNER), that are supported by the U.S. Fish and Wildlife Service (FWS).

MATERIALS AND METHODS

IET and its derivative bioxenophobism has been challenged on diverse grounds. As early as 1969, on the heels of MacArthur and Wilson's (1967) definitive statement, Berkeley biogeographer Jonathan Sauer published an incisive dissection that would remain unappreciated for many years. His critique appeared long before sufficient empirical tests would weaken the model's credibility. Furthermore Sauer's assertion that its spatial determinism overlooked the findings of traditional biogeography made both approaches incommensurable, eventually leading Michael Soulé (1985) to propose that a "new" aggressively militant conservation biology incorporate as twin supporting disciplines both "historical" and "island" biogeography.

Coincidentally Soulé proclaimed an uncompromising stand against exotics that would contribute to a rising tide of vigilantism in wildlife management, and concomitant intolerance of dissenting opinion (Brown 1989, Mann 1991, Lodge 1993, Ruesink et al. 1995, Watlington 1995). Surprisingly, IET was adopted as theoretical underpinning for the reborn "crisis discipline" of conservation biology a full decade after the paradigm entered decline on being discredited by one of its founding adherents as specious (Simberloff and Abele 1976). Subsequent critics proceeded to dismantle the model's forbidding mathematical facade and demolish its claims (Connor and McCoy 1979, Gilbert 1980, Williamson 1981).

In Puerto Rico the formative period of DNER wildlife management policy tracked the rise of IET under the aegis of northern missionaries and their local neophytes. The bioxenophobic tenets of the paradigm soon formed a politically correct syncretism with a naive if chauvinistic nativism that believed Puerto Rico had inherited the best of all possible natural worlds (i.e. free of poisonous snakes, large predators and herbivorous pests). It was only after the articles of ecological faith were securely codified that Ariel Lugo (1987, 1990, 1992; Watlington 1995) came forward to argue that the dichotomy between native and exotic species is irrelevant to the energy flow dynamics of ecosystems.

For his trouble Lugo was soundly vituperated by some of his peers (Mann 1991, Lugo 1996 pers. comm.) despite focusing on plants and avoiding reference to animals. Lugo seems to have gained notoriety by questioning orthodox "biodogma" regarding the supposed fragility and intrinsic species poverty of oceanic island ecosystems and by favoring experimentation with exotics. Incongruously, it appears to have gone unnoticed that E.O. Wilson, the doyen of IET, had early on advocated intentional translocation of species as a creative tool of ecosystem restoration (Wilson and Willis 1975). Some contemporaries of Lugo agreed with Wilson (Conant 1988, Atkinson 1989, Conway 1989).

Eventually, Lugo's position managed to achieve recognition as a respectable "minority opinion" among his mainland peers (Lugo 1994). In Puerto Rico, meanwhile, Lugo's views have met stiff resistance in the DNER and allied circles. Although his proposed incorporation of exotic tree species into forest recovery programs (Lugo 1988, 1995) are gradually being accepted, he himself has been reluctant to pursue the broader implications of the approach with regard to translocation of keystone animal species on behalf of ecosystem enrichment and rehabilitation (Lugo pers.comm. 1996).

The ghost of IET continues to haunt wildlife management in Puerto Rico where its premises persist long after having been discarded elsewhere (See: Whittaker 1995). The present exercise intends to broaden Lugo's insight concerning the usefulness of exotic vegetation for ecosystem rehabilitation by adding key exotic vertebrates, including species that have become naturalized over the years. In deference to conservative conservationism, the ensuing brief survey will be limited to ecological equivalents of animals that once existed on Puerto Rico, as surmised from paleobiogeographical evidence. The approach follows up a suggestion by Atkinson (1989).

RESULTS

When the first Amerindians arrived in Puerto Rico, perhaps eight thousand years ago (Burney et al. 1994), the ocean was fifteen to twenty five meters below present sea level (Hallam 1992). The island boasted one of the most extensive wetlands of the GA, formed as the Holocene transition sea level rise slowly inundated the East Puerto Rico Bank (ca. 10,000 k^2). The bank was previously a Pleistocene savanna (Figure I) with coastline a hundred meters or so below the current sea level until roughly fourteen thousand years ago (14 kya). The ancient coastline and lowlands were soon awash as the continental ice sheets melted away.

As the flooding advanced, the vast wetlands first increased then diminished until only the present vestiges remained. Its scattered hilltops became the outlying islands of Vieques, Culebra and the U.S. and British Virgins. As an extensive ecosystem, the Great East Puerto Rico Swamp may have lasted from about 14 kya to 6 kya, or roughly eight thousand years and covered as much as 500,000 ha, or about twice the area of Cuba's famed Zapata Swamp. The scale and duration of such an extraordinary landscape argue in favor of faunal migrations and adaptive evolution to exploit its habitats (Hallam 1994, Hedges 1996).

Unfortunately, most of the fossil and zooarchaeological evidence needed to support such a conjecture is under water. Moreover, remnant numbers of wetland animals in vestigial peripheral areas are likely to have been hunted to extinction during the millennial intensification of human exploitation that culminated in the Neolithic population explosion during the first five centuries AD (Watlington 1998).

In view of the growing taxonomic diversity of vertebrates: crocodilians, mammals and birds, being recovered from prehistoric contexts on all of the GA, it is open to question whether the so-called "central problem" of a depauperate GA higher fauna is not really an artifact of depauperate archaeological and paleontological prospecting (MacPhee and Iturralde-Vinent 1995, MacPhee and Wyss 1990, Williams 1989, Woods 1989 a and b. Woods and Gaffney 1989).

The following sketches are intended as illustrative congruencies of extinct animals which are presumed to have existed in ancient wetland habitats of Puerto Rico, with naturalized exotics and potential introductions.

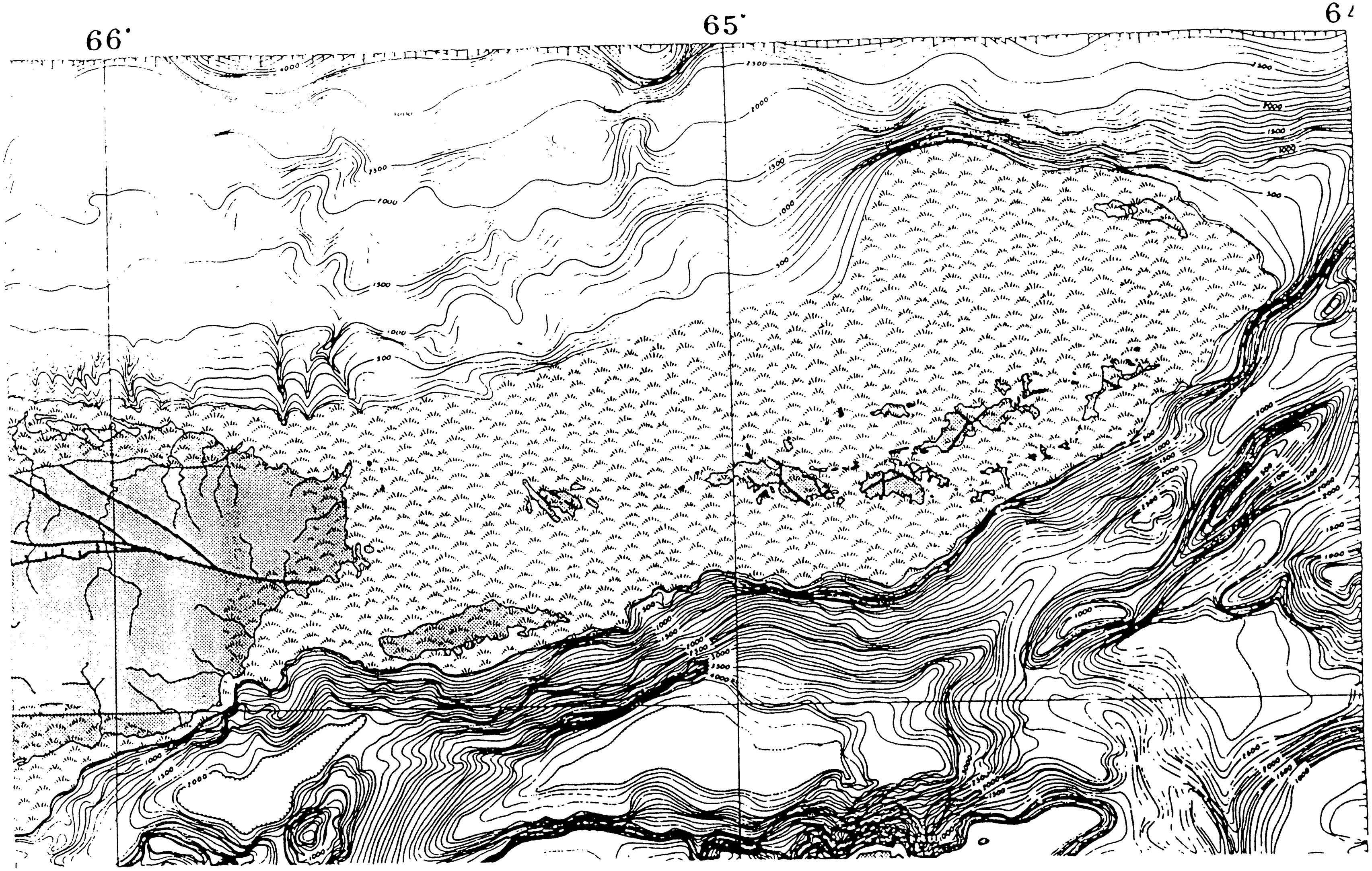


Figure I. The great East Puerto Rico Pleistocene savanna

Caimans and Crocodiles

The DNER was established in 1972 largely in response to a growing civic concern with surface and ground water contamination. Two years later, the nascent agency held its first symposium for scientists invited to present their findings concerning local natural resource problems. One consultant lamented that lack of unspecified geographical conditions in Puerto Rico did not "allow for long food chains and magnification of pesticide residues" in freshwater wetlands (Bonnefil 1974: p. 51).

Clearly, the expert's cryptic comment must have pertained to the absence of a long-lived top-of-the-food-chain carnivore which might serve to monitor bioaccumulation of toxic wastes. However, even as he spoke, recently introduced caimans (*Caiman crocodilus*) native to northern South America were quietly proliferating in the marshes around spring-fed Lake Tortuguero on the north-central coast of Puerto Rico. The karst region where the lake is located also aggregates the largest concentration of pharmaceutical and allied chemical industries on the island, and has one of the highest incidences of cancer and infant diseases (Cruz-Baez and Boswell 1987).

Within a decade the caimans had become naturalized in the 224 ha. lake and adjoining 1,500 ha plus Cibuco-Cienaga Prieta marshlands. Their eventual discovery in the early eighties led to an ill-advised and futile DNER attempt at eradication in 1985 (Santos-Reyes 1988) attended by the first of many sensationalist media campaigns against caimans and other exotics. Having failed to extirpate the caimans, the DNER has turned to a policy of containment that also appears unlikely to succeed.

While the underlying reasons for instatement and maintenance of a "control" attitude toward the caiman are unclear, the most recent public articulation of DNER anti-exotics policy restates an ambiguous dictum: "Once an exotic species has become established, the impact on native species and natural communities can be severe" (Colon-Negrón and Chabert-Llompart 1998). No DNER scientific report substantiating an ecological threat by caimans has ever been forthcoming. Nonetheless, inflammatory feature articles in the dailies cite DNER sources and press releases that allege wanton destruction of fishing nets, decimation of fish stocks, water turtles (*Trachemys*), waterfowl and even predation on rare aquatic plants.

The caimans are a convenient scapegoat for human sins such as pollution and hunting overkill that have turned Tortuguero into a checkerboard of ecological vacancies. Official slander has played on the public's biophobic dread of crocodilians. People are regularly reminded of the dire menace to life and limb that awaits those who dare enter caiman infested areas. In contrast, DNER has ignored the wealth of scientific information that has accrued in other areas of the Caribbean and Tropical America where native caimans and crocodiles are studied and usefully managed as valued keystone components of wetland ecosystems (King 1988, Ross and Garnett 1990, Thorbjarnarson 1991).

The DNER has also chosen not to publicize its own limited findings on the Tortuguero caiman. Several years ago a random analysis of caiman viscera in the agency's laboratory revealed a high concentration of mercury. The finding was suppressed along with the ecological observation that adult caimans were feeding mainly on shoals of introduced Tilapia that had virtually displaced the native fishes. Afterward, the DNER was left in the awkward position of urging hunters to kill caimans, but not to eat them --in case they might be tainted. No similar caveat has been issued for the Tilapias caimans feed on.

Systematic research on environmental chemical contamination in caimans, similar to the well-known work with alligators in Florida (Luoma 1995), has never, to my knowledge, been addressed in Puerto Rico. Lake Tortuguero is said to have been heavily seeded with lead over the many years it was used as a military firing range. There is also the possibility of more insidious, endocrine-disrupting pollutants such as DDE, PCB_s, dioxins and organochlorines, that mimic, amplify or block natural sex hormones in humans as well as other vertebrates. Injection wells were once used by some industries of the region to dispose of their toxic wastes into the same aquifer system that feeds Tortuguero and its adjoining wetlands. There are several superfund sites not far from the lake.

The DNER Bureau of Fisheries and Wildlife (BFW) that oversees the caiman control program is well aware of the Tortuguero anomaly, familiar to island hunters. Despite an apparently suitable environment for waterfowl, the area is reputedly avoided by migrant and resident species. Is the caiman to blame, as the BFW would have it? Is over- exploitation the cause? Or is the local aquatic food chain the real culprit? The bigoted policy that sees caimans only as an intrusive foreign species impedes recognition of their potential role in monitoring bioaccumulation of toxic wastes in Puerto Rico's coastal wetlands.

However, keystone predation by crocodilians was evidently not alien to the great prehistoric marshlands of Puerto Rico. Two separate fossils, a vertebra from the northern San Sebastian formation (MacPhee and Wyss 1990) and a still undescribed jaw from the southern Juana Díaz formation (MacPhee and Iturralde-Vinent 1995) confirm their presence in early Oligocene (30 to 35 mya) sediments. It is uncertain whether these early forms eventually became extinct due to environmental changes brought about by glacial cycles (Pregill and Olson 1981) or by Paleoindian blitz (Steadman, Pregill and Olson 1984).

At least one other wetland predator from the Puerto Rican Oligocene, a pelomedusid side-necked turtle may have disappeared for similar reasons (Wood and Gaffney 1989). Modern species of crocodiles survive on all the GA except Puerto Rico (Schwartz and Henderson 1991, Hedges 1996). The endemic *Crocodylus rhombifer* is a freshwater species native to Cuba and formerly to the Cayman Islands and Bahamas (Olson, 1982). Fossil remains retrieved from Abaco, the northernmost major island (Franz et al. 1995) suggests the species had a much wider range in the GA before rising sea level obliterated extensive areas of freshwater marshes.

Adult *rhombifer* attain a length of 3.5 m, but are said to have reached 15 m in times past (Ross and Garnett 1990). That is over twice the size of the largest male *Caiman crocodilus* which rarely grows to 3 m. However the smaller caiman was also introduced in Cuba's Island of Pines in mid century (Schwartz and Henderson 1991) and is said to be displacing the native species, probably because of its proven adaptability to humanized landscapes. The other GA crocodile is the mangrove or salt water *C. acutus*, a widespread Caribbean species found on the mainland coast from Florida to Venezuela and on Cuba, Jamaica, Hispaniola and Martinique.

The mangrove crocodile is often larger (over 6 m) than its related freshwater cousin with which it is known to hybridize. It appears likely that both species or their ancestral forerunner were at one time part of the native faunal assemblage in Puerto Rico. Hedges (1996) has conjectured that the ancestral species may have arrived on the North Equatorial Current from Africa as recently as 2 to 4 mya in the late Pliocene. If so, a landfall on Greater Puerto Rico is likely. In any case, GA crocodiles are fearless predators of size and disposition that make a translocation more problematical than accepting the naturalized caiman as their ecological replacement.

Hutias, Extinct and Extant

Besides Tortuguero, the only other substantial natural freshwater lake in Puerto Rico is Cartagena, in the semiarid southwest corner of the island. Lying in a small rain-shadow sub-basin of an ancient estuarine synclinal depression, the Lajas Valley, Cartagena is also a relict of the extensive (over 3,000 ha) wetlands that were drained in mid century as part of a costly and ill-fated land reclamation scheme. Its original area, 130 ha, was barely two thirds that of Tortuguero.

In contrast to the northern lake, there are no bubbling springs. Sheet flow and temporary streams fill the lake during the rainy season, September to November, and high evapotranspiration causes a March to August dry-down. The natural energy flow cycle is unusually productive. The lake and its savanna environs was long recognized as one of the island's premier habitats for resident and migratory water birds. No more. Cartagena has been virtually eutrophicated out of existence by nutrient-rich surface runoff from agriculture, livestock and residential effluent.

Open water has almost disappeared beneath a floating mat of peat up to one meter thick. Atop the peat a savanna of tall cattails (*Typha*) grows vigorously, augmenting the biomass. Mismanagement by private and public landowners worsened the problem until a 318 ha National Wildlife Refuge was entrusted to the U.S. Fish and Wildlife Service (FWS) in 1989. The FWS had actively promoted the "unprecedented opportunity to conduct a wetland restoration demonstration project of key significance to the conservation of wildlife and wetland habitats throughout the entire Neotropics" (Schaffner 1994).

Unfortunately, Cartagena is unlike any similar project in most Neotropical areas. There are no native aquatic herbivores that can be used to control the rampant *Typha* introduced in the 1950's by the FWS. Nor are there any caimans or crocodiles to help control eventual herbivores. Recommended solutions feature physical removal of the peat-cattail carpet, a gargantuan undertaking, and a kind of "by-pass surgery" to channel nutrient rich drainwater away from the lake. Water control structures would regulate inflow. Maintenance of the proposed system will have to be mechanical, and therefore costly because FWS stringently rejects the possibility of introducing exotic herbivores (Shaffner pers. comm.), in sharp contrast to former policy and practices until the early 1970's when the IET paradigm was instated.

FWS policy, like that of the DNER is predicated on the ingenuous ahistorical belief that Puerto Rico has always lacked the kinds of keystone faunal components normally found on the Neotropical mainland. A cursory examination of the pertinent biogeographical literature will show that prehistoric Puerto Rico had several kinds of herbivorous mammals closely related, and presumably ecologically equivalent, to certain forms that are widespread in the wetlands of tropical and subtropical America (MacPhee and Iturralde-Vinent 1995, MacPhee and Wyss 1990, Woods 1984, 1989 a & b, Woods and Hermanson 1985).

Only two existing Neotropical mammals are specialized foragers on submerged, emergent and floating aquatic vegetation. The coypu (*Myocastor)* and the capybara (*Hydrochaeris*) belong to the same family of cavy-like rodents, the Caviomorpha, as the domestic cavy or guinea-pig (*Cavia*) and the living and extinct hutias of the GA. There were actually three kinds of hutia: the small mohuys or spiny rats

(*Echimyidae*), the medium, 3 to 5 kg, "true" hutias (*Capromyidae*), and the giant kemis (*Heptaxodontidae*) weighing from around 10 kg up to 200 kg (Biknevicious et al. 1993).

It may surprise some wildlife managers to learn that all of the GA caviomorph rodents started out in Oligocene Puerto Rico, 30 to 35 mya. It is still uncertain whether the ancient spiny rat *Puertoricomys* represents the original ancestor of all the hutias, or whether the Puerto Rican kemi (*Elasmodontomys*) is independently ancestral to all of the giant cavies. It is also unclear whether the initial colonizers rafted from South America or Africa (George 1993). Regardless, the Puerto Rican forms evolved, adapting to available local habitats including wetlands, dispersed to nearby islands and eventually, it seems, to mainland America.

Tantalizing clues, anatomical, cladistic, paleobiogeographical and ecological, suggest that coypus are phylogenetically derived from the same ancestral stock as the capromyid hutias of Cuba, Jamaica, Hispaniola and Bahamas. Likewise, there is good reason to surmise that capybaras evolved in ancient GA wetlands from the same forerunner of the extinct giant heptaxodont cavies that appeared in Anguilla-St. Martin (*Amblyrhiza)*, Hispaniola (*Quemisia*), Jamaica (*Clidomys*) and of course Puerto Rico (*Elasmodontomys*). Elaboration of my hypothesis is the substance of work in progress.

Six of the ten recent species of Cuban hutias are from wetland habitats (Berovides-Alvarez and Comas 1994). All except the largest (*Capromys pilorides*) are rare, if not extinct. The racoon-size hutia conga, which may weigh as much as 7 kg, has survived largely as a specialized forager on red mangrove (*Rizophora mangle*) cays. Their closest living relative the coypu is somewhat larger (7 to 10 kg) and feeds on a broad range of aquatic and semi-aquatic plants in brackish and freshwater marshes (Gosling and Skinner 1984). Although native to southern South America, they have been translocated widely, including the U.S., for their excellent fur and palatable meat.

The coypu is especially effective in controlling cattails and reeds by devouring their starchy rhizomes and soft lower stems. They are so adaptable they have thrived even in extremely nutrient-rich cattle sewage lagoons (Brown 1975). Contrary to anti-exotic myth, they have not been a significant pest anywhere, except low-lying areas in England and the Netherlands where their burrows are said to weaken drainage ditch berms. A more pertinent question is whether the coypu, at home in subtropical latitudes would do as well in Puerto Rico as the more tropical and much larger (to 50 kg and more) capybara. However, capybara herds would have to be protectively managed as a semi-domestic complement to cattle (Ojasti 1991). Otherwise they would be easy game for poachers.

CONCLUSIONS

Wetland restoration in Puerto Rico is hampered by an intractable problem. It is not so much the lack of keystone herbivores, predators and other usual faunal components of normally biodiverse swamp ecosystems. Rejection by wildlife officials and agencies of species translocations that would redress the vacancies is based on the erroneous assumption that a depauperate faunal assemblage is the natural outcome of apparent geographical isolation, which also implies local endemics are frail vis á vis exotic (i.e. foreign) invaders.

Deconstruction of island equilibrium theory, the entrenched ecological paradigm that provides a "scientific" rationale in support of official bioxenophobism is beyond the scope of this presentation. However, it is a step in the right direction to point out that Puerto Rico's apparent dearth of native wetland vertebrates is an artifact of many millennia of human exploitation and consequent extinctions.

ACKNOWLEDGMENTS

Presentation of this paper is made possible by a University of Puerto Rico travel grant.

LITERATURE CITED

Atkinson, I. 1989. "Introduced animals and extinctions." pp. 54-75, in D. Western and M. Pearl (eds.) Conservation for the Twenty-First Century. Oxford U. Press, NY.

Berovides-Alvarez, V. and A. Comas, 1994. "Función ecológica de las jutías (Capromys sp) en los humedales de mangle cubanos." pp. 165-167, in Memorias II Simposio Internacional Humedales. Cienaga de Zapata, Cuba.

Biknevicious, A.R.; D.A. McFarlane and R.D.E. MacPhee. 1993 "Body size in Amblyrhiza inundata (Heptaxodontidae, Caviomorpha) an extinct megafaunal rodent from the Anguilla Bank, West Indies: estimates and implications." American Museum Novitates 3074: 1-25.

Birdsey, R.A. and P.L. Weaver, 1987. "Forest area trends in Puerto Rico." Research Note, S0-331. Southern Forest Experiment Station, New Orleans, LA.

Bonnefil, L. 1974. "The impact of pesticides on the environment of Puerto Rico". pp. 49-51 in Primer Simposio de Recursos Naturales. Department of Natural Resources, San Juan.

Brown, J.H. 1989. "Patterns, modes and extents of invasions by vertebrates." pp. 85-109, in J.A. Drake, et al. (eds.) Biological Invasions, A Global Perspective. Wiley, NY.

Brown, L.N. 1975. "The adaptable nutria". Florida Wildlife, (1): 24-26.

Burney, D.A.; L.P. Burney and R.D.E. Macphee, 1994. "Holocene charcoal stratigraphy from Laguna Tortuguero, Puerto Rico, and the timing of human arrival on the island". Jour. of Archaeological Science 21: 274.

Colon-Negrón, M. and J.L. Chabert-Llompart, 1998. "Posible impacto de las especies exóticas en Puerto Rico y medidas adoptadas por el Departamento de Recursos Naturales para minimizar el mismo." Paper presented at 23rd DNER Symposium of Natural Resources, San Juan.

Conant, S. 1988. "Saving endangered species by translocation." BioScience, 38: 254-257.

Connor, E.F. and E.D. McCoy, 1979. "The Statistics and biology of the species-area relationship". American Naturalist, 113: 791-833.

Conway, W.G. 1989. "The prospects for sustaining species and their evolution." pp. 199-209, in D. Western and M. Pearl (eds.), Conservation for the Twenty-First Century. Oxford U. Press, NY.

Cruz-Baez, A.D. and T.D. Boswell, 1997. Atlas Puerto Rico. Cuban American National Council; Miami, Fl.

Franz, R., G.S. Morgan, N. Albury and S.D. Buckner, 1995. "Fossil skeleton of a Cuban crocodile (*Crocodylus rhombifer*) from a Blue Hole on Abaco, Bahamas." Caribbean Jour. of Science, 31 (1-2).

George, W. 1993. "The strange rodents of Africa and South America." pp. 119-141, in W. George and R. Lavocat (eds.). The Africa-South America Connection. Clarendon Press, Oxford GB.

Gilbert, F.S. 1980. "The equilibrium theory of island biogeography: fact or fiction? Jour. of Biogeography 7: 209-235.

Gosling, L.M. and J.R. Skinner, 1984. "Coypu". pp. 246-251, in I.L. Mason (ed.), Evolution of Domesticated Animals. Longman, London.

Hallam, A. 1994. An Outline of Phanerozoic Biogeography. Oxford U. Press, NY.

_____. 1992 Phanerozoic Sea-Level Changes, Columbia U. Press, NY.

Hedges, S.B. 1996. "The origin of West Indian amphibians and reptiles", pp. 95-128, in R. Powell and R.W. Henderson (eds.) Contributions to West Indian Herpetology, A Tribute to Albert Schwartz. Soc. for Study of Reptiles and Amphibians; Ithaca, NY.

King, F.W. 1985. "Crocodilians: Keystone wetland species." In G.H. Dalrymple, et al (eds.) Abstracts of Proceedings, First National Park Symposium, FIU. Miami, FL.

Lodge, D.M. 1993. "Biological invasions: lessons for ecology." Trends in Ecology and Evolution, 8: 133-137.

Lugo, A.E. 1995. "Tropical forests: their future and our future", pp. 3-17, in A.E. Lugo and C. Lowe (eds.) Tropical Forests: Management and Ecology. Springer-Verlag, NY.

________. 1994. "Maintaining an open mind on exotics". pp. 218-220, in G.A. Meffe and C.R. Carroll, Principles of Conservation Biology. Sinauer; Sunderland, MA.

_____ 1992. "More on exotic species". Conservation Biology, 6 (1): 6.

_____ 1990. "Removal of exotic organisms". Conservation Biology, 4(4): 345.

______ 1988. "The future of the forest: Ecosystem rehabilitation in the tropics." Environment, 30 (7): 17-45.

_______ 1987. "Are island ecosystems different from continental ecosystems?" Acta Cientifica, 1: 48-54.

Luoma, J.R. 1995. "Havoc in the hormones". Audubon, 97 (4): 60-67.

MacArthur, R.H. and E.O.Wilson. 1967. The theory of Island Biogeography. Princeton U. Press, NJ.

MacPhee, R.D.E. and M.A. Iturralde-Vinent, 1995. "Origin of the Greater Antillean land mammal fauna 1. New Tertiary fossils from Cuba and Puerto Rico." American Museum Novitates, 3141: 1-30.

________. and A.R. Wyss, 1990. "Oligo-Miocene Vertebrates from Puerto Rico, with a catalog of localities." American Museum Novitates, 2965: 1-45.

Mann, C.C. 1991. "Extinction: Are ecologists crying wolf?" Science, 253: 736-738.

Ojasti, J. 1991. "Human exploitation of capybara." pp. 236-252, in J.G. Robinson and K.H. Redford (eds.) Neotropical Wildlife Use and Conservation. U. of Chicago Press; Chicago, IL.

Olson, S.L. (ed.) 1982. Fossil Vertebrates from the Bahamas. Smithsonian Contributions to Paleobiology 48, Wash. D.C.

Pregill, G.K. and S.L. Olson, 1981. "Zoogeography of West Indian vertebrates in relation to Pleistocene climatic cycles." Annual Review of Ecology and Systematics, 12: 75-98.

Ross, C.A. and S. Garnett, 1990. Crocodiles and Alligators. Weldon Owen, Sydney.

Ruesink, J.L., I.M. Parker, M.J. Groom and P. Kareiva, 1995. "Reducing the risks of nonindigenous species introductions: Guilty until proven innocent." BioScience, 45: 465-477.

Santos-Reyes, R. 1988. "Programa de control poblacional de caimanes (*Caiman crocodylus fuscus*) en Laguna Tortuguero." p. 123, in Proc. 14th DNER Symposium on Natural Resources, San Juan.

Sauer, J.D. 1969. "Oceanic islands and biogeographic theory: A review." Geographical Review, 59: 582-593.

Schaffner, F.C. 1994. Re-establishment of the open water wetland of Laguna Cartagena National Wildlife Refuge, Phases 1 and 2. U.S. Fish and Wildlife Service.

Schwartz, A. and R.W. Henderson, 1991. Amphibians and Reptiles of the West Indies: Descriptions, Distributions and Natural History. U. of Florida Press, Gainesville, FL.

Simberloff and Abele, 1976. "Island biogeography theory and conservation practice". Science, 191: 285-286.

Soulé, M.E. 1985. "What is conservation biology?" BioScience, 35: 727-734.

Steadman, D.W., G.K. Pregill and S.L. Olson, 1984. "Fossil vertebrates from Antigua, Lesser Antilles: Evidence for late Holocene human caused extinctions in the West Indies." Proc. Natl. Acad. Sci., 81: 4448-4451.

Thorbjarnarson, J.B. 1991. "An analysis of the spectagled caiman (*Caiman crocodilus*) harvest program in Venezuela." pp. 217-235, in J.G. Robinson and K.H. Redford (eds.) Neotropical Wildlife Use and Conservation. U. of Chicago Press.

Watlington, F. 1998. "La transición neolítica en Borikén." pp. 113-120, in J. Seguinot-Barbosa (ed.) Globalization in America: A Geographical Approach. Inst. Carib. Studies, UPR; Río Piedras.

_________. 1995. "Native vs. exotic species: The good, the bad and the ugly." Unpublished manuscript (6 p), Dept. of Geography, U.P.R.

Whittaker, R.L. 1995. "Disturbed island ecology". Trends in Ecology and Evolution, 10: 421-425.

Williams, E.E. 1989. "Old problems and new opportunities in West Indian biogeography." pp. 1-46, in C.A. Woods (ed.) Biogeography of the West Indies, Past, Present and Future. Sandhill Crane Press, Gainesville, FL.

Williamson, M. 1981. Island Populations. Oxford U. Press, NY.

Wilson, E.O. and E.O. Willis, 1975." Applied biogeography." pp. 522-534 in M.L. Cody and J.M. Diamond (eds.) Ecology and Evolution of Communities, Belknap Press, Cambridge, MA.

Woods, C.A. 1984. "Hystricognath rodents." pp. 390-446, in S. Anderson and J.K. Jones (eds.) Orders and Families of Recent Mammals of the World, John Wiley & Sons, NY.

________. 1989 a. "The biogeography of West Indian rodents." pp. 741-798, in C.A. Woods, (ed.) Biogeography of the West Indies... FL.

_______. 1989 b. "A new capromyid rodent from Haiti, The origin, evolution and extinction of West Indian rodents and their bearing on the origin of New World hystricognaths." pp. 59-89, in C.C. Black and M.R. Dawson (eds.) Papers on Fossil Rodents in Honor of Albert Elmer Wood. L.A. Natural History Museum, CA.

________. and E.S. Gaffney, 1989. "New fossil pelomedusid (side-necked) turtle remains from the Oligocene of Puerto Rico. "Jour. of Vertebrate Paleontology, 9 (suppl): 46 A.

________. and J.W. Hermanson, 1985. "Myology of hystricognath rodents: An analysis of form, function and phylogeny." pp. 515-548, in W.P. Luckett and J.L. Hartenberger (eds.) Evolutionary Relationships Among Rodents. Plenum Press, NY.